Treatment methods for waste to be landfilled

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Preface

This report is the result of the study: “Treatment methods for waste to be landfilled” which was financed by the Nordic Council of Ministers through the PA (Products and Waste) Group and carried out during the period November 2008 through August 2009. The study was initiated by the Nordic Landfill Group under the PA Group to provide an overview of the treatment options available to change some of the critical properties of selected waste materials that cannot meet the appropriate waste acceptance criteria for landfilling after implementation of EU Council Decision 2003/33/EC establishing criteria and procedures for the acceptance of waste at landfills pursuant to the Landfill Directive 1999/31/EC.

The project work has been followed by a Steering Group consisting of the members of the Nordic Landfill Group:

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Faroe Islands:	Eyð Eidesgaard
Finland:	Ari Seppänen (Chairman)
Iceland:	Gudmundur Bjarki Ingvarsson
Norway:	Rita Vigdis Hansen and Anette Rognan
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The project has been carried out by DHI in cooperation with the Technical Research Centre of Finland (VTT) and the Swedish Geotechnical Institute (SGI). The following persons have been involved in the study:

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The project leader wishes to express his gratitude to Thorvald Brix Isager, H.J. Hansen Genvindingsindustri A/S, and Anders Skibdal, Stena, for commenting and providing valuable input, particularly on shredder waste.

Summary

The legal framework for landfilling of waste in most of the Nordic countries is based on EU Directive 1999/31/EC of 26 April 1999 on the landfill of waste (the Landfill Directive) and Council Decision 2003/33/EC of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 and Annex II to Directive 1999/31/EC. The implementation of EU Council Decision 2003/33/EC in the Nordic Countries means that certain types of waste will require specific treatment prior to landfilling in order to meet the new waste acceptance criteria (WAC).

In order to provide some guidance on the selection of appropriate treatment technology, this report presents a compilation and evaluation of existing information on treatment methods for selected types of “problem waste” that does not comply with the European (or national) WAC for landfilling, in particular waste types that will not be in compliance even if the so-called “factor 3 rule”, allowing up to a three-fold increase of some of the limit values, is applied. For some waste types the same treatment methods may be applied to improve their properties in relation to reuse.

Three waste streams were selected for the study: Municipal solid waste incinerator (MSWI) air pollution control (APC) residues, shredder residues and MSWI bottom ash. Only the two first waste types constitute real problems in relation to WAC for landfilling. The environmental quality problems for MSWI bottom ash are more related to WAC for reuse than for landfilling, although quality improvements will be necessary to obtain a better sustainability in MSWI bottom ash landfilling. A brief discussion on the properties and treatment options for bioashes is also included.

The report describes the landfill WAC in each of the Nordic countries, resulting from the implementation of the EU Council Decision 2003/33/EC (and in the Faroe Islands, which are not a member of the EU and have their own landfill regulation). Next, the waste properties that may have to be improved to comply with the acceptance criteria are discussed, and a general overview of available waste treatment principles is given. Finally the properties, the potential problems associated with compliance with the landfill WAC, and the treatment options are described for each of the three waste streams. The treatment methods are described and evaluated in terms of technical efficiency, limitations, economy and compliance with appropriate landfill WAC.

The conclusions of the study are described below:

MSWI APC residues

MSWI APC residues, including fly ash, are classified as hazardous waste and must be landfilled at landfills for hazardous waste or landfills for non-hazardous waste receiving stable, non-reactive hazardous waste or they must be placed in underground storage. The main problems in relation to the WAC for hazardous waste landfills and non-hazardous waste landfills receiving stable, non-reactive hazardous waste are the high contents of soluble salts and leaching of Pb and Zn from strongly alkaline residues, and sometimes also leaching of Sb. The most promising treatment methods in terms of technology development, efficiency, maturity of technology, economy and documentation of environmental properties of the treated waste (compliance with the WAC) appear to be extraction of soluble salts combined with chemical stabilisation and possibly recycling of the stabilised remnant to the incinerator grate in order to destroy persistent organic pollutants (e.g. PCDDs/PCDFs). Thermal treatment of APC residues at high temperatures (melting, glassification) may also be effective, but is very energy-consuming and requires re-capture of new APC residues. Stabilisation/solidification using hydraulic binders requires relatively large amounts of binders (cement and/or other pozzolans) and may eventually result in mechanical instability if soluble salts are not removed prior to stabilisation. The prevailing current disposal solution for APC residues in the Nordic countries is landfilling at Langøya in Norway, where the residues, which are mostly alkaline, are mixed with and neutralised with acidic waste prior to disposal in the landfill. This solution is economically very competitive, but the environmental impact, particularly in the longer term, is not very well documented.

The content of salts (mainly calcium chloride) in MSWI APC residues may be recovered and reused, but apart from that, most attempts at finding realistic and environmentally safe reuse applications for these residues have shown little or no success.

Shredder residues

Shredder residues (SR) are classified as non-hazardous waste in most of the Nordic countries. In Denmark, SR are currently classified as hazardous waste. This means that in case of landfilling, the available options are disposal at landfills for hazardous waste or landfills for non-hazardous waste receiving stable, non-reactive hazardous waste. The Danish classification of SR as hazardous waste is currently under revision. In Finland, some of the SR is classified as hazardous waste and some is classified as non-hazardous waste. The main problems in relation to the WAC for landfilling are the high content of TOC and the high amount of DOC in the eluates from the leaching tests. Both often exceed the WAC for hazardous waste landfills. Other potential problems are the leaching of some metals, in particular Pb, from the SR. The

available treatment methods for SR include mechanical separation, thermal treatment (incineration, co-incineration, gasification and pyrolysis) and combinations of mechanical separation and thermal treatment. Most of the treatment methods have been developed for the purpose of increasing the recovery of materials and energy from the shredder residues, not specifically to improve the environmental properties of the treated SR.

The optimal solution in terms of resource recovery, environmental impact and economy appear to be mechanical separation with extensive materials recovery or improved materials recovery integrated into the shredding operation followed by landfilling (if the residual waste stream is not suited for combustion) or co-incineration with MSW (if the residual waste stream is suited for combustion). Landfilling should be developed in the direction of increased sustainability and a reduced aftercare period, and the proportion of SR co-incinerated with MSW (with effective energy recovery) should probably not exceed 5 to 10 % (by mass) of the total feedstock to the incinerator. The bottom ash from the incinerator should be utilised if possible, and the air pollution control residues may require treatment prior to landfilling. If mechanically treated SR is classified as hazardous waste and destined for landfilling or if the EU WAC for landfilling otherwise apply, care should be taken to ensure that TOC (and DOC) is sufficiently low to meet the criteria.

MSWI bottom ash

MSWI bottom ash (BA) is classified as non-hazardous waste in all the Nordic countries, although it may in certain cases also be classified as a hazardous waste in Finland. The BA generally meets the WAC for non-hazardous waste landfills, and potential problems with the environmental quality of BA are more related to requirements for reuse than to the fulfilment of WAC for landfilling. However, the attainment of more sustainable landfilling solutions for MSWI bottom ash could very well require effective treatment prior to landfilling.

The main problems in relation to reuse of MSWI BA and possibly also in relation to sustainable landfilling are chloride, fluoride, Cu, Sb and DOC and sometimes Pb and Cr. Recovery of ferrous and non-ferrous metals as well as crushing and sifting and storage/ageing/carbonisation are parts of the state-of-the-art treatment of MSWI bottom ash. Chlorides may be removed by washing in the quench tank. The choice of the optimal treatment method for bottom ash depends on availability of landfill sites, national policy and criteria for landfilling and reuse in construction works. Often the selection of best available technique requires a multi-criteria assessment that considers a very wide range of drivers (e.g. water consumption and release, energy consumption, etc) in order to arrive at a balanced overall solution. The dry bottom ash treatment is most widely used and generally results in a bottom ash aggregate that can be used for engineering purposes, depending on the

national criteria for reuse. The leaching of copper, which is closely associated with the content of DOC in the leachate, often remains a problem even after bottom ash treatment. The removal of organic matter e.g. by use of an afterburner in the incinerator seems important to solve this problem. This means that the quality of the bottom ash from a dry treatment process is dependent on the quality of the original bottom ash. Thermal treatment of the MSWI BA (melting/glassification) is generally energy-consuming and expensive.

Ash from incineration of biofuels

Bioashes, e.g. from incineration of wood chips or straw, may have a large content of soluble salts, which may be utilised, e.g. in the production of fertilisers (KCl, K_2SO_4). This is done by aqueous extraction, and the remnant, which in the case of straw ash has a relatively high content of Cd, has to be stabilised prior to landfilling. Fly ashes from incineration of biofuels are in many ways similar to MSWI APC residues and may be treated similarly.

Abbreviations and acronyms

ANC	Acid neutralisation capacity
APC	Air pollution control
ASR	Automotive shredder residue
BA	Bottom ash
BTEX	Benzene, toluene, ethylbenzene and xylenes
DL	Detection limit (in chemical analysis)
DOC	Dissolved organic carbon
EC	European Commission
ELV	End-of-life vehicles
EU	European Union
EWG	European Waste Catalogue
FA	Fly ash
HZ	Hazardous
LOI	Loss on ignition
L/S	Liquid to solid ratio
LV	Limit value
MSW	Municipal solid waste
MSWI	Municipal solid waste incinerator
NHZ	Non-hazardous
PAH	Polycyclic aromatic hydrocarbons
PCB	Polychlorinated biphenyls
SDR	Semidry (APC) residue
SR	Shredder residue
TDS	Total dissolved solids
TOC	Total organic carbon
vol/vol	Volumen/volumen
WAC	Waste acceptance criteria
w/w	Weight/weight

1. Introduction

1.1 Background and objective

The legal framework for landfilling of waste¹ in most of the Nordic countries² is based on EU Directive 1999/31/EC of 26 April 1999 on the landfill of waste (the Landfill Directive) and Council Decision 2003/33/EC of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 and Annex II to Directive 1999/31/EC.

Council Decision 2003/33/EC requires waste to be characterised prior to landfilling and sets waste acceptance criteria (WAC) for some of the categories of landfills. During and after the implementation of the Council Decision it has become apparent that certain types of waste cannot meet the EU WAC for hazardous waste landfills, much less those for non-hazardous waste landfills receiving stable, non-reactive hazardous waste. Consequently, the waste cannot be landfilled without prior treatment and/or (in some cases) application of the so-called factor 3 rule³, which on a case-by-case basis and subject to a favourable risk assessment, under certain circumstances allows the WAC to be increased by up to a factor of 3. However, some waste fractions still cannot meet the WAC, even after application of the factor 3 rule. Another possibility may be underground storage, but this option is not always available or considered optimal.

For some types of waste this situation constitutes a substantial problem in several Nordic countries (as well as in other EU member states). The problems experienced with the WAC for some waste streams were discussed at the meeting of the Technical Adaptation Committee (TAC) for the Landfill Directive in Brussels on 7 November 2006. The European Commission informed the member states the possibilities of treating the waste should be considered first, rather than changing the WAC or limit values. The Commission also expressed a wish to collect information compiled on the technical options for treatment of different types of waste. Giving first

¹ Mining waste is covered by another legal framework, namely Directive 2006/21/EC of the European Parliament and of the Council of 15 March 2006 on the management of waste from the extractive industries and amending Directive 2004/35/EC. Issues related to mining waste are not discussed in the study.

² Norway, Iceland, the Faroe Islands and Greenland are not Members of the EU. However, Norway and Iceland are obliged to implement the EU regulations, whereas the Faroe Islands and Greenland are not.

³ The “factor 3 rule” refers to the first part of section 2 in the Annex to Council Decision 2003/33/EC which states that under certain circumstances Member States can allow the WAC (with certain exceptions), to be exceeded by up to three times on a case-by-case basis, provided the emissions present no additional risk to the environment according to a risk assessment.

priority to treatment of waste is actually in line with the Landfill Directive, which in Article 6 (a) states that “Member states shall ensure that only waste that has been subject to treatment is landfilled” with some exceptions for cases where treatment is not feasible or is of little or no benefit. “Treatment” is defined in the Landfill Directive, Article 2 (h) as meaning “the physical, thermal, chemical or biological processes, including sorting, that change the characteristics of the waste in order to reduce its volume or hazardous nature, facilitate its handling or enhance recovery”. It is probably safe to say that most countries have chosen to implement this clause of the Landfill Directive quite gently with emphasis on sorting.

On the above background, the Nordic Landfill Group under the Nordic Council of Ministers has felt a need to get an overview of the treatment options available to change the critical properties of waste materials that cannot meet the landfill WAC in such a manner that they subsequently do fulfil the criteria for acceptance at the appropriate category of landfills.

The objective of this study is therefore to compile and evaluate existing information on treatment methods for selected types of “problem waste” that does not comply with the European (or national) WAC for landfilling, in particular waste types that will not be in compliance even if the “factor 3 rule” is applied. The result of the study should be a report that can be used to disseminate the information to central and local authorities, waste producers, landfill operators, consultants and other stakeholders.

1.2 Approach

The first activity to be carried out was to collect information on problem wastes/waste streams and problematic properties and from those select a limited number of waste materials (two to four) for further study. This process is described in section 1.3 below.

Chapter 2 describes the environmental criteria with which the waste materials to be landfilled must comply, i.e. the EU WAC and their implementation in Denmark, Finland, Iceland, Norway and Sweden. The Faroe Islands are not a member of the EU and have their own landfill regulation.

Chapter 3 discusses the waste properties that may have to be improved to comply with the acceptance criteria for landfills in the Nordic Countries, and provides a systematic general overview and brief explanation of a number of treatment principles and unit operations that may be used in the various waste treatment processes and methods.

Chapters 4 through 6 are dedicated to each of the specific problem waste streams identified in section 1.3 (MSWI APC residues, shredder residue and MSWI bottom ash). Each chapter starts out by a description of the characteristic properties of the waste stream in question, and then discusses the potential problems associated with compliance with the WAC and the objectives of treatment. The next step is to present a list of potential treatment technologies, which are then described and discussed in terms of processes

used, technical efficiency, limitations and development needs, compliance with the EU WAC and other issues, when information is available. If possible and relevant, recommendations are given. A brief chapter 7 on ash from biomass incineration is also included.

Chapter 8 provides a limited amount of information on bioashes and presents one treatment process aimed to produce fertilisers from the residues. To a large extent, the treatment methods described for MSW APC residues in chapter 4 can also be applied to bioashes.

The main conclusions of the study are summarised in chapter 9.

1.3 Selection of wastes

The members of the Nordic Landfill Group have each provided three to four types of problem waste streams which have then been ranked according to the priorities they were given by group members. After some discussion, the resulting list of waste streams was, in order of priority, as follows:

1. MSWI APC residues, including fly ash
2. Shredder waste or residues (fines, fluff)
3. MSWI bottom ash
4. Ashes from biomass incineration (bioashes)

The first three wastes on the list have been subject to further study. In addition, a short discussion on ashes from biomass incineration is also provided. In most cases, MSWI bottom ash will actually fulfil the criteria for acceptance at landfills for non-hazardous waste or mineral waste (near coastal) landfills, and the discussion of treatment methods for MSWI bottom ash may be of more interest in the context of re-use than landfilling.

2. Required environmental properties

2.1 The waste acceptance criteria and treatment needs

In this context only problems related to the environmental properties of the waste materials will be addressed. The study does not address the general requirement to treat waste prior to landfilling; only treatment that is necessary to meet criteria for acceptance at certain categories of landfills is considered. Most of the Nordic countries have adopted the waste acceptance criteria (WAC) directly, more or less as they appear in Council Decision 2003/33/EC, including the possibility to apply the “factor 3 rule” (see footnote 2 in section 1.1). In Denmark, the situation is somewhat different. To maintain a high level of protection of the groundwater, Denmark has decided to distinguish between landfills that are located inland and landfills that are located near the (sea)coast. The WAC for inland landfills are more stringent than those for near-coastal landfills. The latter resemble those defined in Council Decision 2003/33/EC. Denmark has further defined a subcategory of landfills for non-hazardous waste, landfills for mineral waste, and set WAC for those. To a certain extent, the Danish landfills for mineral waste can be compared to the landfills for non-hazardous waste receiving stable, non-reactive hazardous waste.

An overview of the different categories of landfills defined (with or without WAC) in the Nordic countries is shown in table 2.1.

It seems reasonable to assume that treatment of waste to be landfilled will not be relevant for inert waste⁴; if inert waste is treated, the objective will most likely be make the waste suitable for utilization rather than landfilling.

Treatment may be relevant for non-hazardous waste to be placed in landfills for non-hazardous waste in those countries where WAC has been set for non-hazardous waste landfills. This could, for example, be non-hazardous waste to be placed in the non-hazardous waste landfills also receiving stable, non-reactive hazardous waste which were defined in the EU Council Decision 2003/33/EC. It could also be non-hazardous waste to be placed in the mineral waste landfills defined in Denmark. The various EU and Nordic WAC associated with landfills for non-hazardous waste are presented in section 2.2 below.

⁴ Inert waste is a special type of non-hazardous waste with low pollution potential defined in the Landfill Directive (1999/31/EC).

Table 2.1. Categories of landfills in the Nordic countries with (WAC) and without (NoW) waste acceptance criteria based on testing and/or chemical analysis. Rec means that there are recommended non-binding WAC.

Landfill categories	DK	FAR*	FIN	ISL	NO	SE
Landfills for inert waste	WAC		WAC	WAC	Rec**	WAC
Non-hazardous waste landfills (un-specified)				NoW	NoW	NoW
Non-hazardous waste landfills for mineral waste (waste with low organic content)	WAC		Rec			
Non-hazardous waste landfills for mixed waste	NoW		Rec			
Non-hazardous waste landfills receiving stable, non-reactive hazardous waste	WAC		WAC	WAC	WAC	WAC
Landfills for hazardous waste	WAC		WAC	WAC	WAC	WAC

*) The Faroe Islands are not members of the EU and have not adopted the EU regulations. Two types of landfills exist in the Faroe Islands: Environmentally approved landfills and landfills that do not require approval.

**) In Norway, the EU WAC for inert waste are recommended, but without application of the factor 3 rule.

Treatment may also be relevant for hazardous waste to be placed in landfills for hazardous waste or in landfills or landfill cells for non-hazardous waste receiving stable, non-reactive hazardous waste to ensure that the waste meets the WAC for the landfill in question. There could also be an incentive to treat hazardous waste with the objective to re-classify it to non-hazardous waste. Reclassification would require that the property or properties (H1 through H14 in 91/689/EC on hazardous waste and subsequent amendments) which caused the classification of the waste as hazardous are changed. If a waste e.g. is classified as hazardous due to its content of a substance above a certain level, the treatment must reduce the content of that substance to a level below the hazard limit value (and/or change its chemical form to a form not exhibiting hazardous properties). There is, however, not always a “natural” incentive to re-classify hazardous waste to non-hazardous waste. In Denmark, for example, hazardous waste has been exempted from landfill tax whereas non-hazardous waste has not – this has not encouraged reclassification efforts. However, this exemption will soon be phased out. The various EU and Nordic WAC associated with landfills for hazardous waste are presented in section 2.3.

2.2 Landfills for non-hazardous waste

2.2.1 The EU WAC for non-hazardous waste landfills

Council Decision 2003/33/EC does not set any general criteria for acceptance of waste at landfills for non-hazardous waste. However, it does define and set WAC for one specific sub-category of non-hazardous waste landfills, namely non-hazardous waste landfills or landfill cells receiving stable, non-reactive hazardous waste. It also specifies that gypsum-based materials can only be placed in non-hazardous waste landfills in cells where no bio-

degradable waste is accepted. Wastes landfilled together with gypsum-based materials must comply with the limit values for TOC in table 2.2 and the limit values for DOC in table 2.3. Special criteria are also set for acceptance of asbestos waste at landfills for non-hazardous waste.

The non-leaching EU WAC for granular hazardous waste to be placed in landfills for non-hazardous waste receiving stable, non-reactive hazardous waste are shown in table 2.2. Similar criteria have not been set for granular non-hazardous waste to be accepted at the same type of landfills.

Table 2.2. Non-leaching EU criteria for acceptance of hazardous waste at landfills for non-hazardous waste accepting stable, non-reactive hazardous waste (CEC, 2003).

Parameter	Value
TOC (total organic carbon)	5 %*
pH	Minimum 6
ANC (acid neutralisation capacity)	Must be evaluated

* If this value is not achieved, a higher limit value may be admitted by the competent authority, provided that the DOC value of 800 mg/kg is achieved at L/S = 10 l/kg, either at the materials own pH or at a pH value between 7.5 and 8.0.

The EU leaching limit values (LVs) for both hazardous and non-hazardous granular waste to be placed in a landfill for non-hazardous waste receiving stable, non-reactive hazardous waste are shown in table 2.3 in terms of leached amounts at a liquid to solid ratio (L/S) = 2 l/kg and L/S = 10 l/kg and eluate concentrations at L/S = 0–0.1 l/kg (C_0).

No WAC are set at EU level for monolithic waste to be placed in landfills. The Council Decision requires each Member State to set such criteria in such a manner that they provide the same level of environmental protection as the EU WAC for granular waste.

Table 2.3. EU leaching limit values (LVs) for acceptance of granular non-hazardous and hazardous waste at landfills for non-hazardous waste accepting stable, non-reactive hazardous waste (CEC, 2003).

Components	L/S = 2 l/kg	L/S = 10 l/kg	C_0
	mg/kg	mg/kg	mg/l
As	0.4	2	0.3
Ba	30	100	20
Cd	0.6	1	0.3
Cr (total)	4	10	2.5
Cu	25	50	30
Hg	0.05	0.2	0.03
Mo	5	10	3.5
Ni	5	10	3
Pb	5	10	3
Sb	0.2	0.7	0.15
Se	0.3	0.5	0.2
Zn	25	50	15
Chloride	10 000	15 000	8 500
Fluoride	60	150	40
Sulphate	10 000	20 000	7 000
DOC (*)	380	800	250
TDS (**)	40 000	60 000	-

(*) If the waste does not meet these values for DOC (dissolved organic carbon) at its own pH, it may alternatively be tested at L/S = 10 l/kg and a pH of 7.5–8.0. The waste may be considered as complying with the acceptance criteria for DOC, if the result of this determination does not exceed 800 mg/kg.

(**) The values for TDS (total dissolved solids) can be used alternatively to the values for sulphate and chloride.

2.2.2 Denmark

The Danish implementation of the Council Directive 2003/33/EC and the Landfill Directive 1999/31/EC in the form of a Statutory Order (BEK 252, 2009) has led to the definition of three sub-categories of non-hazardous waste landfills, namely landfills for mineral waste, landfills for mixed waste and non-hazardous waste landfills receiving stable, non-reactive hazardous waste. There are no WAC for non-hazardous waste to be placed in landfills for mixed waste. However, mixed waste landfills cannot receive waste that can be accepted at a landfill for mineral waste, and there is a general ban on landfilling of waste that is suitable for incineration.

Mineral waste is defined as “a sub-category of non-hazardous waste, primarily consisting of inorganic, mineral material with a total content of TOC not exceeding 50 g/kg of dry matter. Mineral waste may only to a limited extent be soluble in or react chemically with water.”

Mineral waste landfills are further subdivided into three types, based on location and the surrounding environment:

- Mineral waste landfills located inland (MA0)
- Mineral waste landfills located near the seacoast (MA1)
- Mineral waste landfills located near the seacoast (MA2)

The difference between type MA1 and MA2 is the required dilution potential of the nearby sea (lower for MA2 than for MA1) and corresponding lower leaching limit values for a few contaminants (Ba, Cr and Cu).

Non-hazardous waste to be accepted at any of the three types of mineral waste landfills must meet the limit values for solid content of organic material and substances listed in table 2.4 as well as the leaching limit values described in table 2.5. Hazardous waste to be accepted at any of the three types of mineral waste must meet the criteria in table 2.4 and table 2.5, and must in addition be stable and non-reactive and have a pH above 6. ANC must be determined and evaluated.

For basic characterisation testing of leaching, a column test (CEN/TS 14405) is normally required, and the limit values at all three L/S values must be met. For compliance testing, a batchtest EN 12457-1, performed at L/S = 2 l/kg, is normally required, and it must then also be included in the basic characterisation testing. If testing at L/S = 2 l/kg is impossible or impractical, e.g. because of the water content of the waste, then the waste can be tested at L/S = 10 l/kg, using either a batch test (EN 12457-2) or the column test. It is allowed to use the column test, CEN/TS 14405, for compliance testing at L/S = 0–2 l/kg, collecting only one fraction of eluate.

If the basic characterisation testing shows that there is a substantial difference between the results of the column and the batch test at a level which is relevant in relation to the limit values, then the column leaching data takes precedence in determining whether or not the limit values are met, and

the column test should then be used also for compliance (still collecting the eluate fraction from 0 to 2 l/kg).

Table 2.4. Danish limit values for solid content of organics in non-hazardous waste to be accepted at mineral waste landfills (after BEK 252, 2009).

Parameter	Value
TOC (total organic carbon)	5 %*
BTEX (benzene, toluene, ethylbenzene and xylenes)	15 mg/kg
PCB (polychlorinated biphenyls – the sum of PCB 28, PCB 52, PCB 101, PCB 118, PCB 153 and PCB 180)	10 mg/kg
Sum of hydrocarbons (C6–C40)	450 mg/kg**
PAH (polycyclic aromatic hydrocarbons – the sum of fluoranthene, benz(b+j+k)fluoranthene, benz(a)pyrene, dibenz(a,h)anthracene and indeno(1,2,3-c,d)pyrene)	75 mg/kg
Naphthalene	5.0 mg/kg

* If this value is not achieved, a higher limit value may be admitted by the competent authority, provided that the DOC value of 800 mg/kg is achieved at L/S = 10 l/kg, either at the materials own pH or at a pH value between 7.5 and 8.0. If it can be shown that part of the TOC determined is elementary carbon, this part may be subtracted before evaluation.

** Subject to potential revision in the near future.

Table 2.5. Danish leaching limit values (LVs) for acceptance of granular non-hazardous waste at mineral waste landfills (after BEK 252, 2009).

Components	L/S = 2 l/kg			L/S = 10 l/kg			C ₀		
	mg/kg			mg/kg			mg/l		
	MA0	MA1	MA2	MA0	MA1	MA2	MA0	MA1	MA2
As	0.082	0.40	0.40	0.37	2.0	2.0	0.040	0.30	0.30
Ba	9.5	30	10	28	100	30	5.5	20	6.0
Cd	0.072	0.60	0.60	0.11	1.0	1.0	0.060	0.30	0.30
Cr (total)	0.36	4	1.5	1.0	10	4.0	0.20	2.5	1.0
Cu	5.9	25	15	13	50	35	4.0	30	10
Hg	0.012	0.050	0.050	0.050	0.20	0.20	0.0063	0.030	0.030
Mo	0.44	5.0	5.0	0.90	10	10	0.31	3.5	3.5
Ni	0.22	5.0	5.0	0.50	10	10	0.14	3.0	3.0
Pb	0.28	5.0	5.0	0.60	10	10	0.18	3.0	3.0
Sb	0.022	0.20	0.20	0.080	0.70	0.70	0.012	0.15	0.15
Se	0.17	0.30	0.30	0.31	0.50	0.50	0.12	0.20	0.20
Zn	2.1	25	25	5.0	50	50	1.4	15	15
Chloride (*)	2000	10000	10000	2900	15000	15000	1700	8500	8500
Fluoride (*)	13	60	60	33	150	150	8	40	40
Sulphate (*)	2600	10000	10000	5200	20000	20000	1800	7000	7000
DOC (**)	80	380	380	230	800	800	50	250	250

(*) The limit values for chloride, fluoride and sulphate are only relevant for landfills where the downstream groundwater flows into a surface water body with fresh water.

(**) If the waste does not meet these values for DOC at its own pH, it may alternatively be tested at L/S = 10 l/kg and a pH of 7.5–8.0. The waste may be considered as complying with the acceptance criteria for DOC, if the result of this determination does not exceed 800 mg/kg.

Based on the results of a Nordic study, the new Danish Statutory Order on landfilling of waste does not distinguish between granular and monolithic waste, i.e. monolithic waste is crushed to the requirements of the test method and tested as a granular material. The results are compared to the limit values for granular waste materials.

2.2.3 *The Faroe Islands*

The Faroe Islands have not joined the European Union and have not adopted EU legislation on landfilling. Waste is not tested in the Faroe Islands, but is distributed to incineration, landfilling, special treatment or recycling on the basis of lists (origin and type). Two types of landfills are defined, environmentally approved landfills and landfills that do not require approval. Environmentally approved landfills may receive all normal non-combustible waste that cannot be utilised, asphalt, non-recyclable glass, impregnated wood, sludge from roadwells, oil contaminated soil, asbestos, dredging sediments, MSWI bottom ash (APC residues and fly ash is exported to Norway), batteries containing less than 0.025 % Hg. Landfills that do not require approval may receive topsoil, gravel, sand, compost (including composted organic household waste), branches, leaves, grass and other yard waste and waste concrete.

2.2.4 *Finland*

Finland has defined three sub-categories of non-hazardous waste landfills:

- Landfills for non-hazardous waste with low organic content that cannot meet the criteria for non-hazardous waste landfills receiving stable, non-reactive hazardous waste (landfills for mineral waste)
- Non-hazardous waste landfills receiving stable, non-reactive hazardous waste.
- Non-hazardous waste landfills for non-hazardous mixed waste.

Finland has implemented the EU WAC shown in section 2.2.1, including the factor 3 rule, for non-hazardous waste landfills receiving stable, non-reactive hazardous waste. In addition, Finland has formulated recommendations for non-binding WAC for non-hazardous waste landfills for mineral waste and mixed waste, respectively.

2.2.5 *Iceland*

Iceland has defined two sub-categories of non-hazardous waste landfills:

- General landfills for non-hazardous waste
- Non-hazardous waste landfills receiving stable, non-reactive hazardous waste.

Iceland has implemented the EU WAC shown in section 2.2.1, including the factor 3 rule, for non-hazardous waste landfills receiving stable, non-reactive hazardous waste. No WAC has been set up for non-hazardous waste landfills in general, and no distinction is made between non-hazardous waste landfills for mineral waste and mixed waste, respectively.

2.2.6 Norway

Norway has defined two sub-categories of non-hazardous waste landfills:

- General landfills for non-hazardous waste
- Non-hazardous waste landfills receiving stable, non-reactive hazardous waste.

Norway is enforcing the EU WAC shown in section 2.2.1 for hazardous waste to be accepted at non-hazardous waste landfills receiving stable, non-reactive hazardous waste. From July 1 2009, a limit of 10 % TOC or 20 % loss on ignition will be imposed on non-hazardous waste to be accepted at landfills for non-hazardous waste. For all three classes of landfills, the waste must be tested both at $L/S = 0.1$ l/kg and at $L/S = 10$ l/kg when testing is relevant.

2.2.7 Sweden

Sweden has defined two sub-categories of non-hazardous waste landfills:

- General landfills for non-hazardous waste
- Non-hazardous waste landfills receiving stable, non-reactive hazardous waste.

Sweden has implemented the EU WAC shown in section 2.2.1, including the factor 3 rule, for non-hazardous waste landfills or landfill cells receiving stable, non-reactive granular hazardous waste. The leaching LVs must be met for $L/S = 0.1$ l/kg and $L/S = 10$ l/kg. Waste to be accepted at landfills must – if it has a homogeneous composition - have a TOC < 10% (w/w) or – if it has a heterogeneous composition - a content of combustible material that does not exceed 10% (vol/vol). Exemptions are made for bottom ashes, fly ash and fluegas cleaning residues that can contain up to 18% (w/w) TOC. Exemptions are also made for composted sludge from treatment of sewage, green liquor sludge from recycling of boiling fluids as well as animal waste. Non-hazardous waste landfilled together with gypsum-based materials must comply with the limit values for TOC in table 2.2 and the limit values for DOC in table 2.3.

Based on the results of a Nordic study, the Swedish landfill regulations do not distinguish between granular and monolithic waste, i.e. monolithic waste is crushed to the requirements of the test method and tested as a granular material. The results are compared to the limit values for granular waste materials.

2.3 Landfills for hazardous waste

2.3.1 The EU WAC for hazardous waste landfills

The Landfill Directive (1999/31/EC) states in Article 6 (b) that “only hazardous waste that fulfils the criteria set out in accordance with Annex II is assigned to a hazardous waste landfill.” This requirement is interpreted differently e.g. in Denmark and Sweden. In Denmark, only waste that is classified as hazardous according to Directive 91/689/EC and subsequent amendments may be accepted at landfills for hazardous waste, whereas also non-hazardous waste that fulfils the acceptance criteria for hazardous waste landfills may be placed in hazardous waste landfills in Sweden.

Council Decision 2003/33/EC sets criteria for acceptance of granular hazardous waste at EU level. The non-leaching EU WAC for granular hazardous waste to be placed in landfills for hazardous waste are shown in table 2.6. The EU leaching WAC for hazardous granular waste to be placed in a landfill for hazardous waste are shown in table 2.7 in terms of leached amounts at $L/S = 2$ l/kg and $L/S = 10$ l/kg and eluate concentrations at $L/S = 0-0.1$ l/kg (C_0).

Table 2.6 Non-leaching EU criteria for acceptance of hazardous waste at landfills for hazardous waste (CEC, 2003).

Parameter	Value
LOI* (loss on ignition)	10 %
TOC* (total organic carbon)	6 %**
ANC (acid neutralization capacity)	Must be evaluated

* Either LOI or TOC must be used.

** If this value is not achieved, a higher limit value may be admitted by the competent authority, provided that the DOC value of 800 mg/kg is achieved at $L/S = 10$ l/kg, either at the materials own pH or at a pH value between 7.5 and 8.0.

Table 2.7. EU leaching WAC for acceptance of granular hazardous waste at landfills for hazardous waste (CEC, 2003).

Components	$L/S = 2$ l/kg	$L/S = 10$ l/kg	C_0
	mg/kg	mg/kg	mg/l
As	6	25	3
Ba	100	300	60
Cd	3	5	1,7
Cr (total)	25	70	15
Cu	50	100	60
Hg	0.5	2	0.3
Mo	20	30	10
Ni	20	40	12
Pb	25	50	15
Sb	2	5	1
Se	4	7	3
Zn	90	200	60
Chloride	17 000	25 000	15 000
Fluoride	200	500	120
Sulphate	25 000	50 000	17 000
DOC (*)	480	1 000	320
TDS (**)	70 000	100 000	-

(*) If the waste does not meet these values for DOC at its own pH, it may alternatively be tested at $L/S = 10$ l/kg and a pH of 7.5–8.0. The waste may be considered as complying with the acceptance criteria for DOC, if the result of this determination

does not exceed 800 mg/kg.

(**) The values for TDS can be used alternatively to the values for sulphate and chloride.

No WAC are set at EU level for monolithic waste to be placed in landfills. The Council Decision requires each Member State to set such criteria in such a manner that they provide the same level of environmental protection as the EU WAC for granular waste.

2.3.2 Denmark

The Danish implementation of the Council Directive 2003/33/EC and the Landfill Directive 1999/31/EC has introduced the definition of four sub-categories of landfills for hazardous waste according to location and the surrounding environment:

- Hazardous waste landfills located inland (FA0)
- Hazardous waste landfills located near the seacoast (FA1)
- Hazardous waste landfills located near the seacoast (FA2)
- Hazardous waste landfills located near the seacoast (FA3)

The difference between type FA1, FA2 and FA3 is the required dilution potential of the nearby sea (decreasing from FA1 to FA3) and corresponding lower leaching limit values for a few contaminants (Ba, Cr, Cu and Sb).

Hazardous waste to be accepted at any of the four types of hazardous waste landfills must meet the limit values for solid content of organic material and substances and the requirement for ANC measurement listed in table 2.8 as well as the leaching limit values described in table 2.9. The leaching WAC for FA1 is similar to the EU leaching WAC for granular hazardous waste to be accepted at landfills for hazardous waste.

The same requirements for basic characterisation testing and compliance testing as those applying to non-hazardous waste (see section 2.2.2) also apply to hazardous waste. The limit value must be met at all three L/S values in basic characterisation but is generally evaluated at L/S = 2 l/kg in compliance testing.

As for non-hazardous waste landfills, the Danish regulations do not distinguish between granular and monolithic waste, and monolithic waste is crushed to the requirements of the test method and tested as a granular material.

Table 2.8. Danish non-leaching criteria for hazardous waste to be accepted at hazardous waste landfills (after BEK 252, 2009).

Parameter	Value
ANC (acid neutralisation capacity)	To be evaluated
TOC (total organic carbon)	6 %*
PCB (polychlorinated biphenyls – the sum of PCB 28, PCB 52, PCB 101, PCB 118, PCB 153 and PCB 180)	50 mg/kg**

* If this value is not achieved, a higher limit value may be admitted by the competent authority, provided that the DOC value of 800 mg/kg is achieved at L/S = 10 l/kg, either at the materials own pH or at a pH value between 7.5 and 8.0. If it can be shown that part of the TOC determined is elementary carbon, this part may be subtracted before evaluation.

** Refer to Regulation (EC) No. 850/2004 of the European Parliament and of the Council of 29 April 2004 on persistent organic pollutants and amending Directive 79/117/EEC and subsequent amendments.

Table 2.9. Danish leaching WAC for acceptance of granular hazardous waste at hazardous waste landfills (after BEK 252, 2009).

Components	L/S = 2 l/kg				L/S = 10 l/kg				C ₀			
	mg/kg				mg/kg				mg/l			
	FA0	FA1	FA2	FA3	FA0	FA1	FA2	FA3	FA0	FA1	FA2	FA3
As	0.080	6.0	6.0	6.0	0.37	25	25	25	0.043	3.0	3.0	3.0
Ba	10	100	100	25	30	300	300	75	6.0	60	60	15
Cd	0.090	3.0	3.0	3.0	0.14	5.0	5.0	5.0	0.070	1.7	1.7	1.7
Cr (total)	0.39	25	12	3.5	1.1	70	35	10	0.23	15	7.0	2.5
Cu	6.8	50	30	25	15	100	70	55	4.4	60	20	17
Hg	0.012	0.50	0.50	0.50	0.051	2.0	2.0	2.0	0.0064	0.30	0.30	0.30
Mo	0.52	20	20	20	1.0	30	30	30	0.36	10	10	10
Ni	0.25	20	20	20	0.53	40	40	40	0.16	12	12	12
Pb	0.32	25	25	25	0.72	50	50	50	0.21	15	15	15
Sb	0.025	2.0	2.0	1.0	0.080	5.0	5.0	3.5	0.014	1.0	1.0	0.60
Se	0.21	4.0	4.0	4.0	0.39	7.0	7.0	7.0	0.15	3.0	3.0	3.0
Zn	2.4	90	90	90	5.3	200	200	200	1.6	60	60	60
Chloride (*)	2300	17000	17000	17000	3300	25000	25000	25000	1900	15000	15000	15000
Fluoride (*)	15	200	200	200	38	500	500	500	9	120	120	120
Sulphate (*)	3100	25000	25000	25000	6200	50000	50000	50000	2200	17000	17000	17000
DOC (**)	100	480	480	480	280	1000	1000	1000	60	320	320	320

(*) The limit values for chloride, fluoride and sulphate are only relevant for landfills where the downstream groundwater flows into a surface water body with fresh water.

(**) If the waste does not meet these values for DOC at its own pH, it may alternatively be tested at L/S = 10 l/kg and a pH of 7.5–8.0. The waste may be considered as complying with the acceptance criteria for DOC, if the result of this determination does not exceed 800 mg/kg.

2.3.3 The Faroe Islands

The Faroe Islands do not have any landfills specifically for hazardous waste (see section 2.2.3).

2.3.4 Iceland

Iceland has implemented the EU WAC for granular hazardous waste to be landfilled at hazardous waste landfills. However, Iceland has no hazardous waste landfills for the moment (all hazardous waste is exported).

2.3.5 Finland

Finland has implemented the EU WAC for granular hazardous waste to be landfilled at hazardous waste landfills.

2.3.6 Norway

Norway has implemented the EU WAC for granular hazardous waste to be landfilled at hazardous waste landfills.

2.3.7 Sweden

Sweden has implemented the EU WAC for granular hazardous waste to be landfilled in landfills for hazardous waste.

As already mentioned, Sweden does not distinguish between granular and monolithic waste, and monolithic waste is crushed to the requirements of the test method and tested as a granular material.

3. Treatment principles

3.1 Required improvements

The types of improvement required to meet the WAC shown in chapter 2 or to re-classify a waste material will differ from one type of waste to another. The problems requiring attention for any given waste may be one or more of the following:

- Too high content of organic material (TOC)
- Too high leachability of organic material (DOC)
- Too high content of specific organic contaminants (mainly a potential problem in Denmark where WAC have been set on the content of certain organic contaminants for acceptance at landfills for mineral and hazardous waste, see tables 2.4 and 2.8, but can also be a more general potential problem for waste with a high content of PCB)
- Too high content of soluble salts (chlorides, sulphates and possibly fluorides)
- Too high contents or too high leachability of trace elements
- Too low pH

These properties are directly related to the WAC in terms of limit values. More subtle requirements for treatment could e.g. arise from a combined evaluation of the acid neutralisation capacity (ANC) and the results of a pH-dependence test.

The appropriate treatment for each of these problems depends of course on the nature of the problem, but also on the nature of the waste material and on the type of solution sought (e.g. removal/reduction of total content of the problematic contaminant(s) or reduction of the leachability of the contaminant(s) or an increase of the ANC). A treatment method which improves one property of the waste may have a negative effect on another property. One type of treatment may for example reduce the leachability of some trace elements (e.g. heavy metals such as Cu, Pb, Zn) while at the same time mobilising other trace elements (e.g. those forming oxyanions such as As, Cr, Mo, Sb and Se) and vice versa.

A treatment process for a particular type of waste may therefore aim at solving more than one specific problem and may hence encompass more than one treatment principle. It is generally necessary to test and optimise waste treatment processes first in the laboratory and then in larger pilot scale, and finally to demonstrate its technical feasibility in full or near-full scale on a waste stream that exhibits the full range of variability normally occurring.

The best basis for evaluation of a given treatment process is of course a fully established plant operating under commercial conditions.

3.2. Overview of treatment principles

3.2.1 *Treatment principles*

The methods and processes available for treatment of waste prior to landfilling (and/or utilisation) practically all employ one or more of the following treatment principles:

- Separation
- Stabilisation and/or solidification
- Thermal treatment
- Biological treatment

Each of these treatment principles are briefly discussed below in terms of unit operations and processes.

3.2.2 *Separation*

Some of the most common separation unit operations are:

- Density and particle size based separation
- Magnetic separation
- Eddy current separation
- Washing and extraction, e.g. at various pH values
- Chemical precipitation
- Crystallisation/evaporation
- Ion exchange
- Distillation
- Electrolysis
- Electrokinetic separation

Separation, which is the simplest treatment principle, may include both physical and chemical processes and combinations of these. *Density and particle size based separation* covers a wide range of well-known unit processes such as sieving, filtration, sedimentation, flotation, removal in cyclones and hydrocyclones and air classification, used to separate solid particles from water/liquids and air/gas streams and to separate different size fractions of particles or particles of different densities or surface properties from each other. These separation methods may e.g. be combined with optical control systems such as colour finders.

Hand sorting is also employed e.g. in material recovery operations at shredding plants.

Magnetic separation is used to separate ferromagnetic metal from a waste stream (e.g. MSWI bottom ash), and *eddy current separation* is used to remove non-ferromagnetic metals such as copper and aluminium from waste streams (also used for MSWI bottom ash).

Washing and extraction are well established separation processes in waste treatment, e.g. for removal of readily soluble salts from MSWI residues such as bottom ash, APC residues and fly ash. Apart from the washed residue, the process does produce a saline wastewater stream that must be managed, e.g. by *chemical precipitation*, where undesired contaminants are precipitated, for example by inducing a change of pH or adding a precipitation/flocculation chemical, and subsequently removal by sedimentation. Soluble salts (e.g. calcium chloride, the main constituent of dry/semidry MSWI APC residues) may also be removed from solution by *evaporation* of the water and *crystallisation*, particularly if the salts are to be recovered in a relatively pure form. An alternative to evaporation and crystallisation is recovery of specific salt ions by selective *ion exchange*.

Distillation is a separation process that may be used to separate liquids with different boiling points (as long as they are not forming azeotropes). *Electrolysis* may be used to separate (and transform) ions dissolved in water, whereas *electrokinetic separation* may be used to remove various ions from (water saturated) contaminated soil or waste. The contaminants are transported electrokinetically across a membrane and dissolved in water.

3.2.3 Stabilisation and/or solidification

In contrast to separation processes, stabilisation and/or solidification processes aim to keep the contaminants in place and immobilise and retain them within the resulting matrix. The following processes may be distinguished:

- Stabilisation/solidification using hydraulic binders
- Stabilisation/solidification using pore-filling materials
- Chemical stabilisation

Treatment of washed or unwashed MSWI APC residues with cement (often in conjunction with the addition of proprietary stabilisation agents) is an example of *stabilisation/solidification using hydraulic binders*. Other waste materials with pozzolanic properties (e.g. coal fly ash) may partly or totally replace cement in some stabilisation processes. In general, the objective of this process is to obtain both physical strength/integrity and chemical stabilisation of the product through the production of a monolithic material. The physical stability will provide resistance (through increased tortuosity) against the diffusion of contaminants out of the material and hence provide a slower release rate. At the same time chemical reactions may transform a contaminant into a less soluble form and hence lower the total leachability of that contaminant (on the other hand, chemical reactions that mobilises other reactants may also occur). For contaminants that are not transformed

chemically during the stabilisation/solidification process, only the physical retention will affect the rate of release. In those countries (e.g. Denmark, Finland and Sweden) where monolithic materials are to be tested as granular materials after crushing, stabilised materials depending only on physical retention of contaminants will get little or no credit for stabilisation when tested and compared to the limit values.

Another type of *stabilisation/solidification process using pore-filling binders* is represented e.g. by the addition of bitumen (in a cold or warm process) to MSWI bottom ash. In this case, the matrix of the waste is encapsulated by the added binder and effectively prevented from contact with water. Little or no release of contaminants will hence take place (as long as the binder remains intact).

Chemical stabilisation is achieved by adding or bringing the waste into contact with reagents which causes chemical changes in the waste matrix and/or the contaminants, thereby reducing the mobility of the latter. The effect of chemical stabilisation alone does not rely on physical matrix retention of contaminants due to solidification, and hence full credit is obtained when the stabilised material is tested in a granular state. Some of the processes that occur when reducing waste materials come into contact with oxygen (oxidation) and alkaline materials come into contact with carbon dioxide (carbonisation), either through contact with atmospheric air during storage (ageing) and/or waste-produced gas after landfilling, or as part of a treatment process prior to landfilling, may be classified as chemical stabilisation if they lead to a more “stable” product.

3.2.4 Thermal treatment

Several types of thermal treatment are relevant for treatment of waste, including the following:

- Incineration
- Pyrolysis/gasification
- Co-incineration
- Sintering
- Melting
- Vitrification
- Thermal desorption

Incineration of MSW and other types of waste, including hazardous waste, is the most common waste treatment process. Incineration reduces the mass and, in particular, the volume of the waste, which is transformed into solid residues (bottom ash, boiler and economizer ash, fly ash) and gases, which are again treated by acid gas control (APC) systems to produce other solid residues (APC residues). The solid residues from incineration themselves often need treatment prior to landfilling. Most incinerators are mass burn incinerators that do not require prior treatment of the waste, whereas other

types of incinerators (e.g. those burning refuse derived fuel, RDF) do require pre-treatment of the waste. While mass burn incinerators operate under fully oxygenated conditions, other types of thermal treatment processes such as *pyrolysis or gasification* processes operate without oxygen or at low oxygen concentrations. The gas produced may be consumed at a later stage within the same process or it may be used for other purposes. The solid residues will be different from those produced by a mass burn incinerator. *Co-incineration* may refer to co-combustion of two or more specific waste types. However, the term co-incineration is often used for the addition of limited amounts of (hazardous) waste to the fuel at cement kilns, coal fired power plants or MSW incinerators.

Inorganic waste types (e.g. MSWI residues) may be treated thermally at different temperatures. *Sintering*, which typically occurs at temperatures of approximately 400 to 850 °C, may cause reorganisation of solid phases, thereby reducing the availability of various contaminants for leaching. *Melting* or fusion occurs at temperatures of 1100 to 1500 °C or higher. Simple melting often results in a mixture of crystalline phases, molten metal phases and possibly unmelted feed. *Vitrification* or glassification of waste occurs at temperatures similar to those of melting and often requires addition of glass precursor material. Vitrification produces an amorphous, single phase glass product with very low leachability of contaminants. When evaluating melting and vitrification processes it is particularly important to consider the entire mass balance, because several components may be partly or totally evaporated and must subsequently be captured and managed.

Thermal desorption is a process whereby organics may be separated from a waste material by evaporation and/or distillation. The waste may be heated indirectly up to approximately 500 °C by a heating agent running in pipe coils or by electrical heating elements. This process is e.g. used for treatment of oil-based drill cuttings.

3.2.5 Biological treatment

Various types of *biological treatment* may be applied to waste containing residual or major fractions of biologically degradable organic matter. Such treatment processes include for example composting, anaerobic digestion, landfill farming and phytoremediation. The processes occurring with MSWI bottom ash during storage or ageing probably include biological degradation of some of the residual organic matter.

4. MSWI APC residues, including fly ash

4.1 Waste characteristics

4.1.1 Characteristics of fly ash and acid gas cleaning residues

Treatment to remove particulates and acid gasses from the flue gas constitutes an integral part of any state-of-the-art municipal solid waste incinerator. This treatment results in a significant reduction of the air emissions from incinerators, but it also causes the production of a substantial amount of solid residues and/or wastewater, which needs to be managed. The major MSWI APC residue streams are fly ash and acid gas cleaning residues with or without fly ash. In this context, only fly ash and fly ash mixed with acid gas cleaning residue from the dry/semidry lime injection APC systems are addressed (Hjelmar et al., 2004).

Fly ash consists of relatively fine ash particles, which are entrained in the flue gas from the boiler and recovered in electrostatic precipitators or fabric filters. At incinerators equipped with wet scrubbing systems for removal of acid gases, the fly ash is always collected upstream of the scrubber(s) and constitutes a separate residue stream. In some cases, the separately collected fly ash may subsequently be mixed with the sludge from treatment of the wastewater from the wet scrubber as part of the residue management system prior to landfilling. At incinerators equipped with semidry or dry lime injection APC processes, the fly ash and the acid gas cleaning residues may or may not be collected separately, depending on residue management systems or local regulation. The amount of fly ash produced by MSW incinerators will normally range from 10 to 30 kg/ton of incinerated waste, depending on the properties of the waste, the incineration technology and the particulate collection system. IAWG (1997) estimates the average amount of fly ash produced at a state-of-the-art mass burn incinerator at approx. 20 kg/tonne of incinerated waste.

The amounts and properties of the *solid acid gas cleaning residues* produced depend both on the type of acid gas cleaning equipment used and on the properties of the feed waste. Traditionally, the main APC processes employed at MSW incinerators are dry/semidry and wet scrubbing processes and combinations of these.

Dry and semi-dry lime based APC systems operate by injecting dry powdered lime or finely atomized lime slurry into the hot flue gas stream. The lime reacts with the acid gasses (HCl, HF, SO₂), and the reaction products, predominantly calcium chloride or calcium hydroxychloride, are subse-

quently collected in bag filters or electrostatic precipitators together with excess lime and fly ash (if the fly ash has not been removed). Since a better reaction stoichiometry, i.e. a more complete reaction between the acid gases and the lime, is generally achieved in the semidry process than in the dry process, the content of unreacted excess lime will usually be higher in the residues from the dry APC process than in the residues from the semi-dry APC process. In some countries, e.g. in Denmark and Sweden, it is common practice not to remove the fly ash prior to the injection of lime and a mixture of reaction products, excess lime and fly ash is therefore produced. In other countries, e.g. in the Netherlands, upstream removal of the fly ash is a regulatory requirement, and in those cases the residues from semidry (and dry) APC systems occur as two separate streams of fly ash and reaction products/excess lime, respectively. Sometimes activated carbon is added to the lime powder or lime suspension before it is injected, to increase the transfer of particularly mercury from the flue gas to the residues. The added activated carbon and the captured mercury will be present in the solid residue.

In the wet scrubbing processes the fly ash is always removed by an upstream electrostatic precipitator or bag. The acid gasses are dissolved in water and subsequently neutralised, and the residues consist of the fly ash stream and a sludge resulting from treatment of the wastewater from the scrubbing process. This treatment usually consists of a pH adjustment by lime or sodium hydroxide and subsequent precipitation followed by addition of further precipitation with an organic sulfide, often tri-mercaptoptriazine (TMT), which forms compounds of particularly low solubility with divalent metal ions. The result is a treated wastewater stream with a high content of salt, particularly calcium chloride, and an organic sulphide sludge with a significant content of trace elements/heavy metals. In some cases, the fly ash and the sludge are managed as separate waste streams, and in other cases the two streams are combined prior to landfilling. In a two-stage wet scrubber process, the scrubbing liquid in the second stage is kept nearly neutral, usually by the addition of sodium or calcium hydroxide, in order to enhance SO_x removal efficiency. In the latter case, a small, separate stream of contaminated calcium sulphate (gypsum) may be produced.

Both fly ash and dry/semidry acid gas cleaning residues with or without fly ash appear as very fine grey/brown and white/grey powders. Due to the hygroscopic properties of calcium chloride, the acid gas cleaning residue will absorb water and eventually form hard lumps. Both react strongly alkaline with water, although fly ash may initially show a neutral or acidic reaction that generally changes to alkaline on prolonged contact with water.

One very interesting characteristic of dry/semidry APC residues is its content of very fine particles of elementary aluminium. In the very alkaline environment created when the residues come into contact with water, aluminium may be oxidised under the formation of hydrogen (Hjelmar et al., 2007). This has led to explosion accidents during conditioning of semidry residues with water prior to landfilling.

4.1.2 Composition of fly ash and dry/semidry APC residues

The MSWI fly ash and dry/semidry acid gas cleaning residues are characterised by high contents of soluble salts (particularly calcium chloride, but also chlorides and sulphates of sodium and potassium), significant contents of several trace elements/heavy metals (e.g. Pb, Zn, Cu, Cd, Sb and As), a high pH in contact with water and a substantial alkalinity (particularly the semidry residue, which contains excess lime). The soluble salt contents of acid gas cleaning residues are generally substantially higher in acid gas cleaning residues than in fly ash. Apart from chloride and sulphate, the major constituents of fly ash are Ca, Si, Mg, Fe, Al, K and Na. Most of the trace elements – with the exception e.g. of Hg – are associated with the fly ash and the fly ash part of the semidry APC residue. In general, the dry and semidry APC residues have a significantly higher alkalinity, a higher content of soluble salts and a higher pH than the fly ash.

Figure 4.1 shows acid titration curves for a fly ash and a semidry APC residue from Danish incinerators (Hjelmar et al., 2006a).

Figure 4.1 pH titration curves for a fly ash from Vestforbrænding and a semidry APC residue from Amagerforbrænding in Denmark (Hjelmar et al., 2006a).

The titration curves in figure 4.1 clearly show the difference in pH buffering capacity between the semidry APC residue and the fly ash.

The APC residues also contain trace organics such as polychlorinated dibenzo-p-dioxines and dibenzofuranes, polychlorinated benzophenoles, chlorinated benzenes and phenols, polychlorinated benzenes and biphenyls and polycyclic aromatic hydrocarbons (PAHs). Some examples of the composition of MSWI fly ash and acid gas cleaning residues are shown in Appendix 1.

4.1.3 Leaching properties of fly ash and dry/semidry APC residues

Figures 4.2a to 4.2c show the results of column tests (CEN/TS 14405) and batch leaching tests (EN 12457-2) for a fly ash from Vestforbrænding in Denmark and a semidry acid gas cleaning residue (with fly ash) from Am-

agerforbrænding in Denmark (Hjelmar et al., 2006a). The results are shown as accumulated leached amounts as a function of the liquid to solid (L/S) ratio of the substances for which leaching criteria have been set in Council Directive 2003/33/EC, plus aluminium, as a function of L/S. The horizontal lines in the figures are the EU WAC at $L/S = 10$ l/kg for waste to be landfilled at a hazardous waste landfill (HZ WAC) and for waste to be landfilled at a non-hazardous waste landfill that accepts stable, non-reactive hazardous waste (NHZ WAC).

Lines with slope = 1 have been inserted into the figures. A slope of 1 for the release curve for a substance is an indication of solubility control, i.e. the concentration of the substance is the same in consecutive eluates from the column test. Note that plotting the release of a substance with a concentration level below the detection limit as a function of L/S using the detection limit as default values would also give a slope = 1.

The two APC residues shown may not be representative of all MSWI fly ashes and semidry APC residues but they do exhibit the typical non-compliance with the EU NHZ WAC and HZ WAC for chloride and sulphate. Similar problems may be seen with e.g. Pb and Zn for other MSWI fly ashes and APC residues.

In Annex 1, the leachability of an MSWI fly ash and a semidry APC residue as a function of pH is shown for a number of parameters, including those for which EU leaching WAC have been set. The pH dependent leaching of the APC residues was determined at $L/S = 10$ l/kg using CEN/TS 14997 (Hjelmar, 2007).

Figure 4.2a Examples of results of column and batch leaching tests on fly ash and semidry (SD) APC residue (with fly ash) from Danish MSW incinerators (Hjelmar et al., 2006a). A more detailed view of the upper part of the plots for chloride and sulphate are shown in figure A.1 in Appendix 1.

Figure 4.2b Examples of results of column and batch leaching tests on fly ash and semidry (SD) APC residue (with fly ash) from Danish MSW incinerators (Hjelmar et al., 2006a).

Figure 4.2c Examples of results of column and batch leaching tests on fly ash and semidry (SD) APC residue (with fly ash) from Danish MSW incinerators (Hjelmar et al., 2006a).

4.2 Classification, challenges and treatment objectives

MSWI APC residues, including fly ash, are generally classified as hazardous waste according to the European Waste Catalogue, EWC (CD, 2000). The acid gas cleaning residues are “born” hazardous waste regardless of any threshold concentrations and listed as:

19 01 07* Solid wastes from flue-gas treatment

The MSWI fly ash has a so-called mirror entry in the EWC, and whether or not it is a hazardous waste depends on the content of dangerous substances with hazardous properties (unless an EU member state determines that it will be a hazardous waste under all circumstances in that country). The fly ash is listed as:

19 01 13* Fly ash containing dangerous substances

Due to its high content of e.g. Pb, fly ash will normally be classified as a hazardous waste.

MSWI APC residues may hence be landfilled at landfills for hazardous waste or at landfills for non-hazardous waste receiving stable, non-reactive hazardous waste, provided they comply with the appropriate WAC. From the leaching data shown in section 4.1.3 it is clear that the residues must be treated to meet the WAC prior to landfilling.

The main problems in relation to the EU leaching WAC are the high contents of soluble chlorides and sulphates. Some residues will not comply with the chloride leaching WAC, even if the “factor 3” rule (see section 2.1) is applied. In many cases leaching of Pb and Zn may also exceed the leaching WAC, particularly for very alkaline residues. For residues with lower pH, some of the oxyanions, in particular Sb, may become problematic. The high content of chloride and high content of free lime (resulting in a high

pH), particularly in dry/semidry APC residues, mobilises the amphoteric Pb and Zn through complexation. As can be seen from figure A.1 in Annex 1, a lower pH will, on the other hand, tend to mobilise several of the trace elements that form oxyanions (e.g. As and Sb) as well as e.g. Ba, Cd and Ni.

Treatment of APC residues may have different objectives. If the objective for example is to re-classify the material as a non-hazardous waste (or product), the substances rendering the residues hazardous must be removed or changed into a non-hazardous chemical form. This would generally require not only a relatively extensive treatment, but also the formation of a valuable end-product that could be sold to off-set the treatment costs which are likely to be considerable.

In this context the objective is less ambitious, namely to improve the properties of the residues to the extent that they meet the EU (and national) WAC for landfilling. In generic terms, the optimum target properties for MSWI APC residues after treatment aimed at improving the landfilling properties would probably be a well buffered material with a slight to moderately alkaline reaction with water and with a low content of soluble salts and trace elements as well as a low content of hazardous organic substances. The treatment carried out to achieve this objective may be combined with efforts to recover and recycle some of the constituents in the MSWI APC residues, e.g. the calcium chloride.

4.3 Current MSWI APC residue production and management in the Nordic countries

The current status of production and management of MSWI APC residues in the Nordic countries are summarised in table 4.1. It should be noted that the MSW incinerators in Sweden do not receive household waste exclusively, but some other types of waste are incinerated as well. This is also true for e.g. Denmark, where industrial waste types, waste from trade and commerce as well as residual waste from recycling stations are also incinerated. It should also be noted that the production of APC residues in Finland will increase significantly (to several times the current production) due to changes in the management of municipal waste. The information on MSWI APC residue production and management has been provided by the members of the Nordic Landfill Group.

Table 4.1. Overview of production and management of MSWI APC residues in the Nordic countries.

Country	APC residue production t/y (year)	Classification	Main form of disposal
Denmark	90,000 (2006)	Hazardous waste	Exported to Norway and Germany
Faroe Islands	Approx. 1,000	Hazardous waste	Exported to Langøya in Norway
Finland	10,000–15,000	Hazardous waste	Landfilling
Iceland	Approx. 800 (2007)	Hazardous waste	Stored for later disposal
Norway	30,000	Hazardous waste	Landfilled at Langøya (a little used as an additive to concrete)
Sweden	202, 920 (2008)	Hazardous waste	75,000 t were exported to Langøya & Mo in Rana, Norway. The rest was landfilled at Swedish landfills/underground. The export is expected to increase in 2009.

4.4 Overview of potential treatment methods for MSWI APC residues

This overview and the following discussion draw upon information provided in a comprehensive and excellent report published by ISWA (2008): “Management of APC residues from W-t-E plants - An overview of management options and treatment methods.” Information has also been obtained from Hjelmar et al. (2004). It should also be noted that DHI has developed a process for treatment of MSWI APC residues – the so-called VKI process – and therefore has access to more detailed information on the properties of the treated residues from this process than from the other technologies discussed.

Most of the treatment methods and technologies that have been applied to MSWI APC residues are based on one or more of the following process types:

- Separation
 - Washing and extraction
 - Electrokinetic separation
- Chemical stabilisation/transformation
- Solidification
- Thermal treatment
 - Vitrification
 - Melting
 - Sintering
 - Pyrolysis

4.5 Technical description of treatment methods

4.5.1 Washing and extraction

The simplest way to reduce the amount of soluble salts in APC residues is to subject it to an aqueous extraction. The effect of this washing process is illustrated in figure A.1 in Appendix 1 which shows the pH dependent leaching of an MSWI fly ash and a semidry APC residue which have been washed in a large scale pilot plant (Hjelmar, 2007). The leachability of chloride, sodium and potassium is strongly reduced, whereas the leachability of sulphate and most trace elements/heavy metals, which are solubility controlled, are practically unaffected. The extracted calcium chloride may be recovered and recycled (Birch et al., 1993). Although an aqueous extraction in itself may not be sufficient to achieve the treatment objectives for landfilling, it may constitute the first step of a combined treatment (e.g. with subsequent chemical stabilisation or solidification, see below).

Washing of fly ash and sludge from the wet APC process with water is e.g. applied at the incinerator in Sundsvall in Sweden based on the *T.I.L. Integrated LAB ash treatment process* (ISWA, 2008). It is reported that the washed residues which has water a content of 40 to 50 % meet the leaching criteria for landfilling, except for Cr and Mo. The plant has obtained dispensation for the WAC for these two substances based on the factor 3 rule. It should be noted that the mixing of the sludge from the wet process with fly ash also provides some chemical stabilisation that reduces the leachability of several substances – the “*Bamberg model*” (Reimann, 1990).

In the so-called *3R process*, which has been proven in large pilot scale, pre-collected fly ash from incinerators equipped with wet scrubbers is extracted with the acidic first scrubber stage effluent from which the mercury has been removed. The filter cake from the extraction is fed onto the grate, thereby destroying any organic contaminants. The extracted metals may possibly be recovered (Vehlow, 1995). The same process is commercially available (and in use) as the *FLUWA process*, marketed by von Roll. ISWA (2008) reports a treatment cost of 150 to 250 €/tonne for the FLUWA process, but does not comment on compliance with EU WAC for landfilling. A similar process, the *Symbiosis Process*, in which alkaline residues from the dry and semidry APC processes are extracted with acid scrubber effluent from the wet scrubbing process, has been studied by Rasmussen et al. (1993). No information is available on the compliance of the residues from the 3R and symbiosis processes with the current EU WAC. The *Halosep process* (currently owned by Stena Metall A/S in Denmark) is based on the same principle – extraction of APC residues with hydrochloric acid from the wet scrubbing process followed by various precipitation processes to remove/recover metals and evaporation to recover salts. The remnant may be landfilled (documentation of compliance with EU WAC not available) or fed into the Wapro reactor as part of the Watech process (see section 4.5.5). The Halosep process has been tested in pilot scale.

4.5.2 Electrokinetic separation

An electric current can be used to transport metal ions in a residue suspension towards an anode or cathode, and ion exchange membranes facilitate the separation of the metal ions from the suspension (Pedersen et al., 2003; Ottosen et al., 2003). The *electrokinetic separation process* requires the metal ions to be in an aqueous phase and has been tested in small scale only. No data on residue properties in relation to the EU leaching WAC for land-filling are available. The method will not be further discussed.

4.5.3 Chemical stabilisation

Several chemical stabilisation processes for treatment of MSWI APC residues have been developed. These processes focus on retaining most of the critical inorganic contaminants (often with the exception of the salts) and reduce their leachability rather than removing them. Some of the processes (the VKI, DRH and Ferrox processes) include extraction of the soluble salts prior to or during the stabilisation reactions, whereas other methods (e.g. Wesphix) do not. There are several advantages associated with the removal of the salts from the residues that contain 25 to 50 % of readily soluble material: It enables recovery of e.g. calcium chloride, it reduces the volume to be landfilled, and it provides for a more effective chemical stabilisation, undisturbed by salt effects.

The *VKI process* (Hjelmar & Birch, 1997; Hjelmar et al., 2001, Hjelmar & Hansen, 2006; Hjelmar et al., 2006a) which has been developed by DHI (formerly as VKI) in co-operation first with I/S KARA and FLS Miljø A/S and later with I/S Amagerforbrænding, I/S Vestforbrænding and AV Miljø, aims at the transformation of the residues into materials suitable for landfilling. The process consists of an initial (two-stage process) or simultaneous (one-stage process) removal of soluble salts by an aqueous extraction followed by chemical stabilisation of the remnant with carbon dioxide (CO₂) and/or phosphoric acid (H₃PO₄) and washing with water followed by dewatering. Both carbon dioxide and phosphoric acid have substantial pH buffering capacities. This means that they can reduce the high pH of the residues without totally removing the alkalinity, which serves as protection against future acidification. In addition, both carbonate and phosphate form compounds of low solubility with many of the metals present in the residues. The resulting product, which appears as a filter cake, is suitable for landfilling. The wastewater from the process, which corresponds to L/S = approximately 3 l/kg, must be treated to reduce the content of trace elements/heavy metals prior to recovery of the salts or discharge to a wastewater treatment plant or a recipient. The process water may be co-treated with the wastewater from e.g. the wet scrubbing APC process (pH adjustment with lime and polishing with trimercapto-triazine, TMT).

The VKI process has been tested and optimised in large pilot scale at three locations (see figure 4.3). The treated residues have been shown to comply

with the EU leaching WAC for non-hazardous waste landfills receiving stable, non-reactive hazardous waste (and for hazardous waste landfills), see figure 4.4 and Appendix 2. The best stabilisation for fly ash was found using CO_2 alone, whereas the best results for semidry residue were found using both CO_2 and H_3PO_4 . From the results in Appendix 2 it can be seen that some of the oxyanions (most notably dichromate) are mobilised by the treatment (but still in compliance with the WAC), and that e.g. the leaching of Sb increases and may exceed the WAC if pH decreases too much. It should also be noted that this treatment does not destroy any organic contaminants (POPs) present – if necessary that may possibly be accomplished by subsequent sintering of the treated residues on the incinerator grate.

Figure 4.3 The VKI process in pilot scale at Vestforbrænding in Denmark (Hjelmar et al., 2001). The same equipment was used to test the Ferrox process.

Sketch project plans have been produced for full scale application of the VKI process see Appendix 2 (Hjelmar et al., 2006a). Based on this, the treatment costs (operating costs and depreciation of investments) for a one-stage full scale continuously operating VKI process plant for treatment of 28000 tonnes of MSWI APC residues per year have been estimated at 630 to 770 DKK/tonne (corresponding to 84 to 102 €/tonne) for semidry gas cleaning residues and 520 to 700 DKK/tonne (69 to 93 €/tonne) for MSWI fly ash (2006 prices). The prices at the lower ends of the ranges do not include the possible addition of phosphoric acid and the costs of separate process waste water treatment (a conventional treatment plant for wastewater from a wet scrubber may be available) whereas this is included in the higher prices. The price of treatment of MSWI fly ash may be reduced by

40 DKK per tonne if addition of lime can be avoided. The price does not include the costs of landfilling.

Figure 4.4 Examples of stabilisation results for the VKI process or Cd and Pb (Hjelmar et al., 2001).

The *DRH process* which has been developed by Dansk Restprodukt-håndter- ingt in Denmark is similar to the VKI process, but is only using CO₂ as the stabilizing agent. The process has been tested in pilot scale in Switzerland, and should also, according to ISWA (2008) be able to comply with the WAC for non-hazardous waste landfills receiving stable, non-reactive haz- ardous waste. No economic information is reported but treatment costs should be similar to those for the VKI process.

The *Ferrox process* has been developed by Babcock & Wilcox Vølund and DTU (ISWA, 2008). It involves five steps: residues are first mixed with a FeSO₄ solution and then aerated with atmospheric air at L/S = 3 l/kg in order to oxidize Fe(II) to Fe(III) and precipitate iron oxides. This step also includes extraction of soluble salts. The pH of the suspension is maintained at 10 to 11 for about 0.5 to 1 hour to allow dissolved heavy metals to bind to the precipitated iron oxides. The fourth step involves dewatering of the treated residues and finally a washing step to exchange the remaining water and remove salts. The final stabilised product has a water content of about 50 %. ISWA (2008) does not report on compliance/non-compliance with the EU WAC for landfilling. Results resembling those from the VKI and DRH processes may be expected. However, the aeration step tends to oxide Cr (III) to Cr(VI), which may cause some formal problems with compliance with the WAC (after landfilling, most of the Cr(VI) is likely to be reduced again). It has also been suggested to reduce Cr(VI) in the product by adding elementary Al. The Ferrox process has been tested in pilot scale and is offered commercially by Babcock & Wilcox Vølund but no plants have built yet. ISWA (2008) reports treatment costs of approximately 65 €/tonne for a 20000 tonnes per year plant.

The *WesPhix process* also applies stabilisation with phosphoric acid. In this case the acid is injected into the solid residues which may have been

moistened to avoid dusting. The process has been developed and is marketed by Wheelabrator in the USA (Lyons, 2000). It is applied commercially in the USA, Japan and Taiwan (ISWA, 2008). Lyons (2000) reports a treatment cost in the USA of 15 to 25 US\$/tonne. No data on compliance/non-compliance with EU WAC for landfilling are available.

4.5.4 Solidification with or without chemical stabilisation

Solidification is probably one of the most widely applied treatments of MSWI APC residues prior to landfilling. Most of the solidification processes are based on the mixing of the residues with a hydraulic binder (cement), water and various proprietary additives and subsequent maturing with development of strength and resistance to water movement and leaching. If no chemical stabilisation of the residues takes place, the effect will be purely physical (and without effect if the material is crushed and tested for leaching as a granular material). In many cases, however, the result is a combination of the physical solidification and chemical (stabilisation) reactions. The cement or concrete matrix is strongly alkaline and may in fact mobilise certain amphoteric substances such as Pb and Zn. As long as the cement matrix is intact, the Pb and Zn will be retained, but if it disintegrates or is crushed, these metals will be highly leachable. In general, better cement stabilisation results should be obtained if the soluble salts are removed from the residues prior to solidification. Experience has shown that it is impossible to prevent a significant release of salts (particularly chlorides) from cement stabilised/solidified dry and semidry APC residues in contact with water unless the salts have been extracted prior to the solidification.

Cement stabilisation/solidification (S/S), often with proprietary additives, transforming the materials into monolithic waste, is extensively used e.g. in France. Council Directive 2003/33/EC requires that the individual EU member states set leaching WAC for landfilling of monolithic waste that provide the same level of environmental protection as the WAC for granular waste. France has chosen to disregard this requirement (see Hjelmar and Hansen, 2009), and compliance with French landfill regulation and French WAC for monolithic waste is therefore no guarantee of compliance with European legislation.

Inertec is a French company and solidification/stabilisation process based on the use of hydraulic binders that can be tailored to specific waste types. The process is applied commercially to hazardous waste, including MSWI APC residues, at several landfills in France (ISWA, 2008). No information about compliance/non-compliance with regulation is provided. A treatment and disposal price of 200 to 220 €/tonne of APC residue is reported.

Ragn-Sells in Sweden operates a stabilisation/solidification and landfilling process where MSWI APC residues are treated with cement, water and pozzolanic coal fly ash prior to landfilling. The waste is reported to comply with the EU WAC for hazardous waste landfills. The treatment price, in-

cluding landfilling, is reported at 1000 to 1600 SEK/tonne (90–146 €/tonne).

The *gypsum solidification process* that takes place at NOAH A/S on Langøya in Norway may also be categorised as solidification (ISWA, 2008). The MSWI APC residues are crushed in order to remove aggregates and then mixed with water at a density of 1.5 tonnes/m³ and stored in tanks with stirring devices. The slurry is pumped through a grinder in order to remove larger particles. It is then mixed with waste acid (e.g. sulphuric acid from TiO₂ production) and slaked lime. During the mixing a controlled pH of 4 to 5 is maintained. In this step gypsum is precipitated together with some of the heavy metals (hydroxides). In the third step, remaining metals in solution (iron as well as other heavy metals) are precipitated by adding slaked lime and raising the pH to 8–9. The final mixture is pumped to the old limestone quarry that makes up Langøya. Remaining residue solids, gypsum and precipitated metals are then deposited at the bottom of the waterfilled quarry by sedimentation and subsequently consolidated by settling. Water is recycled and used in the neutralisation process. Surplus water from the process is treated to remove heavy metals prior to discharge (ISWA, 2008). No information is available on compliance/non-compliance with the EU WAC for landfilling. The gate fee for treatment and landfilling of MSW APC residues at Langøya is reported at approximately 50 €/tonne (ISWA, 2008).

Another solidification and disposal process is *backfilling of old mines* after mixing of the MSWI APC residues with various other solid and liquid waste materials. This process is applied e.g. in Germany by Liebherr/GTS GmbH & Co. KG, and a gate fee of 70 to 100 €/tonne of APC residue is reported (ISWA, 2008). No information on the environmental properties of the resulting material is reported.

In the Netherlands, MSWI fly ash has been used as a *filler material in asphalt* (IAWG, 1997). This can be seen as an example of stabilisation with a pore-filling binder.

4.5.5 Thermal treatment processes

The main advantages of most thermal treatment processes are the formation of a very dense and stable end-product and the destruction of POPs such as dioxins and furans. The main disadvantages are the high energy demand and the production of new off-gasses that must be cleaned. *Melting and vitrification* both occurs at temperatures of 1200 to 1500 °C or higher. Vitrification requires addition of glass precursors, e.g. quartz sand, whereas melting does not. At least 30 to 40 melting and vitrification plants for treatment of MSWI bottom and/or APC residues are in operation, most of them in Japan (ISWA, 2008). It is probably not without reason that no cost estimates for melting and vitrification of APC residues have been provided for the ISWA report by e.g. *JFE Engineering* (electric resistance melting), *Daido Steel* (electric plasma arc melting) and *Takuma* (melting) who all have supplied technical information for the report. An ISWA position paper estimates the

cost of melting/vitrification to range from 100 to 500 €/tonne of residue (ISWA, 2003) and Ecke et al. (1999) reported that costs of thermal treatment of MSWI APC residues in Japan might be as high as 480 €/tonne. Due to the high costs it is unlikely that these melting or vitrification processes will be attractive for treatment of MSWI APC residues in the Nordic countries in a foreseeable future. *Sintering*, which occur at lower temperatures (see section 3.2.4) and may take place on the grate of the incinerator, probably has a lower cost and may be relevant as a supplement e.g. to some of the extraction/chemical stabilisation processes (see section 4.5.3).

A special melting process is the so-called *CTU process* developed by Conzepte Technik Umwelt AG in Switzerland. In this process the MSWI APC residues are mixed with pretreated shredder waste in a primary melting furnace heated by a gas burner to about 200 °C. Slag, dust and gases move to a secondary furnace kept at 1400 °C. Melted slag and metals are removed from the bottom of this furnace, while gasses are treated in a wet cleaning system and also used for electricity production and heat. Wastewater from gas cleaning is treated for heavy metals. Slags are quenched and can be separated from molten metals. The process thus uses another waste stream, shredder waste, as energy supply and it will destroy any POPs present in the residues to be treated (ISWA, 2008). The technology has shown promising results in pilot scale but has not yet been proven in full scale. At 100,000 tonnes per year, Cramer et al. (2006) estimates the treatment costs at approximately 65 €/tonne of *shredder waste*. Nielsen et al. (2006a) estimates a gate fee between 90 and 112 €/tonne of *shredder waste* if the produced slag has a low value. The process may treat equal amounts of APC residues and shredder waste on a mass basis (Nielsen et al., 2006a).

Another special thermal treatment process for MSWI APC residues is the Danish *Watech pyrolysis process* which also requires input of another waste stream such as shredder waste. It may be considered an add-on to the Halosep process described in section 4.5.1. The extracted remnants from the Halosep process are transferred to a pyrolysis (“Wapro”) reactor together with the shredder waste. The outputs from this step include coke, gases, acid solution, and oil/condensate. Non-soluble residue particles are expected primarily to be found in the coke fraction which is washed and used as energy input to the process together with the washed gas. The remaining ash constitutes the “treated residue” (ISWA, 2008). The process has shown promising results in laboratory and possibly pilot scale. No reliable cost estimates are reported.

4.6 Overview and discussion of treatment options for MSWI APC residues

Based on the information described in section 4.5, an overview of the options available for treatment of MSWI APC residues prior to landfilling may be given in terms of process type, technology development, commercial

availability, compatibility of the treated product with EU WAC and estimated treatment cost, see table 4.2.

Table 4.2 Overview of selected existing and potential technologies for treatment of MSWI APC residues prior to landfilling.

Technology	Process type	Technology development	Commercially available?	Product compatible with EU WAC?	Estimated treatment costs €/tonne
VKI process	Chemical stabilisation	Large scale pilot	No*	Yes	69–102
DRH process	Chemical stabilisation	Large scale pilot	Yes	Yes, probably	No information
Ferrox	Chemical stabilisation	Large scale pilot	Yes	Yes**	65
FLUWA	Washing/acid extraction	Full scale operation	Yes	Not documented	150–250
T.I.L. Integrated LAB	Washing/extraction	Full scale operation	Yes	No***	No information
WesPhix	Chemical stabilisation	Full scale operation	Yes	Probably not	15–25
Inertec	Cement based S/S	Full scale operation	Yes	Not documented	200–220
Ragn-Sells	Cement based S/S	Full scale operation	Yes	Yes (stated by R-S)	90–146
NOAH, Langøya	Gypsum solidification	Full scale operation	Yes	Not documented	~ 50
Backfilling of mines	Solidification/stabilisation	Full scale operation	Yes	Not documented	70–100
JFE Engineering	Melting (electric resistance)	Full scale operation	Yes	Probably	100–500
Daido Steel	Melting (plasma)	Full scale operation	Yes	Probably	100–500
Takuma	Melting (furnace)	Full scale operation	Yes	Probably	100–500
CTU	Melting with SR	Tested in pilot scale	No	?	65–112****
Watech	Extraction and pyrolysis	Tested in pilot scale	No	?	No information

*: Sketch project of full scale plant exists

** : A potential problem with chromium oxidation may be solved by addition of Al

***: Problems with leaching of Cr and Mo solved by means of dispensation (using the factor 3 rule)

****: Cost expressed per tonne of shredder residue (SR) co-treated with MSWI APC residue

Not all information is available for all process types and technologies, and the preconditions upon which the information is based vary widely from one process to another. Due to the lack of consistent and comparable data the comparison of the processes is to a certain extent based on assumptions.

It should be noted that the cost estimates listed in table 4.5 probably are somewhat incompatible. ISWA (2008) e.g. points out that the prices for backfilling of mines NOAH and Inertec seem to include both treatment and

final disposal. The same is probably true for Ragn-Sells, whereas most of the other costs listed do not include landfilling. In addition, the estimated costs for the different technologies do not refer to the same year, and the conditions concerning plant capacity, depreciation interest rates and periods are generally not known. So, at best, the cost ranges shown are indicative.

5. Shredder residue (SR)

5.1 Waste characteristics

5.1.1 General characteristics and composition of SR

Shredder waste or shredder residue (SR) is generated from businesses that recover metals from automobiles or end-of-life vehicles (ELV) and discarded metal (iron) containing products - often referred to as white goods (Forton et al. 2006, Boughton and Horvath, 2006), although appliances are not necessarily dominating (Isager, 2009; Skibdal, 2009). The input to the process may also include some electric and electronic (EE) waste. Shredder residue from the processing of cars alone is often referred to as automotive shredder residue (ASR) or “car fluff”. The composition and properties of SR depends on the input to the metals recovery process and the type and effectiveness of that process. The SR undergoes changes with time due to changes in the production of goods (e.g. the replacement of steel with plastic in car manufacturing) and improved materials recovery technology. Figure 5.1 shows an example of some of the main unit operations of a relatively simple shredder plant.

Figure 5.1 Diagramme showing a simple shredding facility. The SR consists of the fractions marked: “To landfill” (Forton et al., 2006). It should be noted, however, that the processes have been optimised and that the normal yield of separated non-ferrous products today is closer to 5% than the 1 to 3% shown (Isager, 2009; Skibdal, 2009).

Many shredder plants are much more sophisticated than the one shown, and apply further separation steps to the waste, in particular to extract more metal before the waste stream is discarded. At H.J. Hansen Genvindingsindustri A/S in Denmark, iron, non-ferrous metal (e.g. aluminium, copper, brass), glass and plastic fractions are recovered (Nielsen et al., 2006a, Isager, 2009).

Important drivers that influence the composition and management of SR within the EU jurisdiction are EU directives such as the Hazardous Waste Directive and the associated European Waste Catalogue, the ELV Directive (2000/53/EC), the Waste Incineration Directive (2000/76/EC), the Landfill Directive (1999/31/EC) and Council Decision 2003/33/EC). Other important drivers are the metal demand and metal prices and the development of materials separation technology.

Many of the descriptions of the composition and properties of shredder waste have focused on ASR from shredding of ELVs. GHK (2006) states that ASR comprises approximately 50 % by weight of the total SR stream (assumingly at European level), and that the composition of non-ASR does not differ substantially from from that of ASR, except that white goods may contain more copper than ELVs and that SR from white goods may contain PCBs which ASR does not (the latter is confirmed by Hjellnes Consult (2008)). GHK (2006) further states that SR from white goods would be unlikely to contain such fractions as shredder fibre. One of the larger Danish shredding companies, H.J. Hansen Genvindingsindusri A/S, states that ELVs constitutes 10 to 20% of the input to the shredding process (Nielsen et al., 2006a). Figure 5.2 shows pictures of typical SR from Denmark.

Figure 5.2 Pictures of shredder residues (SR) from three different landfills in Denmark.

As can be seen from figure 5.2, the shredder waste is quite heterogeneous. It consists of a variety of materials. Table 5.1 and table 5.2 show typical compositions of light and heavy fractions of ASR (from shredding of ELVs) reported by Francois (2003). Based on numerous sources, Zevenhoven and Saeed (2003) report the composition ranges of ASR shown in table 5.3.

Table 5.1. Typical composition of ASR light fraction (Francois, 2003).

Material	Composition and particle size	Content (wt%)
Polymers	2-10 cm	9
Foams PUR	10/10-20/20 in cm (size of PUR pieces in the residues)	8
Rubber	Long pieces (gaskets seals), no tires	3
Metals	Wires / Al plates	2.5
Cu wires	Length: 10-20 cm	1
Wood	Pieces 10-20 cm	1
Mixed PUR with textiles	Textiles, carpets, leather etc	32.5
Minerals	Stones, sand, glass, dust etc	43

Table 5.2. Typical composition of ASR heavy fraction (Francois, 2003). It should be noted, though, that the content of tyres generally is much lower today (Isager, 2009).

Material	Composition and particle size	Content (wt%)
Polymers		19
Rubber	Tires (45 %), seals and gaskets (55 %)	55
Metals	Fragmentised Fe and non ferrous metals	5
Wood		7
Textiles	Small pieces	3
Cu wires	Cable (5-15 cm)	3
Minerals	Stones, up to 100–500 g/piece	8

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Table 5.3. Typical composition data for ASR from various sources (Zevenhoven and Saeed, 2003) from the period from 1995 to 2003.

Material	Reported content in ASR (weight percentage)					
	a	b	c	d	e	f
Plastics	30–48	20	21.5	41		33
Plastics (foam)						15
Plastics (including coatings, textile)					83.1	
Elastomers (including rubber)	10–32	20	5.3	21	2.6	18
Fibres (textile, wood, paper)	4–26	25	53.7	10		10
Paints, lacquer	3–10			5		
Metals	~ 20		8.1		13.5	3
Glass, ceramics, electric materials	3–16		3–5	19		
Dust, soil, etc.	10–20		excl.			
Inert (glass, sand, grit, etc.)		35				
Other (residues)			7.9	4	0.6	21
Oils, water	15–17					

a: Keller (2003), b: Galvagno et al. (2001), c: Das et al. (1995), d: Mirabile et al. (2002), e: Lanoir et al. (1997), f: Ambrose et al. (2002).

The data in tables 5.1, 5.2 and 5.3 are some years old and may not be representative of the situation today (Isager, 2009). Isager has provided composition data from sorting of SR in 2009, see table 5.4.

Information on the chemical composition is available from Norway (Hjellness Consult, 2008a) and Denmark (Hjelmar, 2009), see table 5.5. The Danish sample was subjected to removal of most of the remaining free metal and stones, and subsequent crushing and cutting (using cryogenic techniques) prior to total chemical analysis.

Table 5.4. Typical composition of SR (10–15 % ELV). From H.J. Hansen Recycling Industry Ltd., February 2009 (Isager, 2009).

Material	Content (% w/w)	Comments
Ferrous > 6 mm	3.7	Mostly Cu wires and Al
Non-ferrous > 6 mm	3.3	
Wood	3.8	
Plastics	10.5	
Plastic foam	1.9	Textile, carpets, leather and miscellaneous (mixture of all fractions)
Rubber	3.5	
Other	13.8	
Inorganics > 6 mm	5.2	
< 6 mm	54.3	Stones, glass, ceramics
		Rust, sand, soil, glass, dust

Table 5.5. Examples of chemical composition of SR (Hjelmar, 2009; Hjeltnes Consult, 2008a).

Parameter	Unit	Denmark (mixed)	Norway (only ELVs)	Norway (mixed)
Number of samples	-	1	2	6
Si	mg/kg	93000		
Fe	mg/kg	14000		
Ti	mg/kg	3900		
Ca	mg/kg	37000		
Mg	mg/kg	6700		
Na	mg/kg	8700		
K	mg/kg	4100		
Al	mg/kg	30000		
As	mg/kg	29	4.2–6.3	5.7–36
B	mg/kg	560		
Ba	mg/kg	2100		
Br	mg/kg	230		
Cd	mg/kg	33	12	13–63
Cl	mg/kg	4300		
Co	mg/kg	520		
Cr	mg/kg	770	120–140	75–380
Cu	mg/kg	46000	14000–30000	5200–65000
Hg	mg/kg	1.8	0.56–0.91	1.1–1.9
Mn	mg/kg	1900		
Mo	mg/kg	51		
Ni	mg/kg	1800	180–230	130–800
P	mg/kg	1200		
Pb	mg/kg	3000	1500–1800	1500–5500
S	mg/kg	2600		
Sb	mg/kg	130		
Se	mg/kg	< 3		
Sn	mg/kg	380		
Sr	mg/kg	240		
V	mg/kg	39		
W	mg/kg	2400		
Zn	mg/kg	14000	8500–9300	7300–30000
TOC	g/kg	180		
LOI	g/kg		230–700	88–700

The samples shown in table 5.5 were also analysed for content of a number of organic substances and groups of substances. Some of the results are presented in table 5.6. Further information may be found in Hjellnes Consult (2008a) and Hjelmar (2009).

Table 5.6 Examples of the content of organic substances in SR (Hjelmar, 2009a; Hjellnes Consult, 2008a).

Parameter	Unit	Denmark (mixed)	Norway (only ELVs)	Norway (mixed)
Number of samples	-	1	2	6
PAH (16 US EPA)	mg/kg	7.4	9–14	13–32
BTEX	mg/kg	9.4	13–32	4.1–15
PCB (7 congeners)	mg/kg	5.6	n.d.	n.d.–3.6
Hydrocarbons				
C6–C10	mg/kg	16		
C10–C25	mg/kg	1700		
C25–C35	mg/kg	6800		
C5–C8	mg/kg		< 20	< 10
C8–C10	mg/kg		< 20	< 10–16
C10–C12	mg/kg		< 20–24	22–96
C12–C16	mg/kg		240–1100	180–740
C16–C35	mg/kg		8000–30000	5100–25000
C5–C35 (NO)	mg/kg		8200–31000	5400–26000
C6–C35 (DK)	mg/kg	8600		

n.d. = Not detected

In relation to thermal treatment of SR, the calorific value is of importance. Calorific values ranging from approx. 9 to approx. 15 MJ/kg with typical values at 12 to 13 MJ/kg have been reported for Danish shredder residues after 2000 (Force Technology, 2009).

5.1.2 Leaching characteristics of SR

Column leaching data have been obtained (using CEN/TS 14405) for two Danish shredder residues from different shredder plants (one of the samples, SR1, is the one for which composition data are reported in tables 5.4 and 5.5). The results are shown in figures 5.3a to 5.3c as accumulated leached amounts of the substances for which leaching criteria have been set in Council Directive 2003/33/EC, plus aluminium, as a function of L/S. For SR1, batch test results at L/S = 2 l/kg (EN 12457-1) and L/S = 10 l/kg (CEN/TS 14997, own pH) are also shown. Batch tests (EN 12457-2 at L/S = 10 l/kg) have also been performed on 6 of the Norwegian samples for which composition data are presented in tables 5.4 and 5.5. The results of the Norwegian batch leaching tests are also shown in figures 5.3a to 5.3c. The horizontal lines in the figures are the EU WAC at L/S = 10 l/kg for waste to be landfilled at a hazardous waste landfill (HZ) and for waste to be landfilled at a non-hazardous waste landfill that accepts stable, non-reactive hazardous waste (NHZ).

Figure 5.3a Results of column and batch leaching tests on Danish and Norwegian SR samples. Sources: Hjelmar (2009a), DHI (2009), Hjeltness (2008b).

Figure 5.3b Results of column and batch leaching tests on Danish and Norwegian SR samples. Sources: Hjelmar (2009a), DHI (2009), Hjeltness (2008b).

Figure 5.3c Results of column and batch leaching tests on Danish and Norwegian SR samples. Sources: Hjelmars (2009a), DHI (2009), Hjellness (2008b).

Lines with slope = 1 have been inserted into the figures. A slope of 1 for the release curve for a substance is an indication of solubility control, i.e. the concentration of the substance is the same in consecutive eluates from the column test. Note that plotting the release of a substance with a concentration level below the detection limit as a function of L/S using the detection limit as default values would also give a slope = 1.

Information on the pH dependent leaching of the Danish waste sample, SR1, is shown in Appendix 2 together with data on the acid neutralisation capacity of the waste.

5.2 Classification, challenges and treatment objectives

The composition and leaching data does not lead to a clear-cut classification of shredder residues in accordance with the European Waste Catalogue (EWC). The shredder residue may be found in the EWC under two mirror entries for wastes from shredding of metal-containing wastes:

- 191003* Fluff-light fraction and dust containing dangerous substances
- 191005* Other fractions containing dangerous substances

Wastes under mirror entries may or may not be hazardous waste; it depends on the content of dangerous substances with hazardous properties. The rules may be applied and interpreted slightly different from one country to another, and a waste material which is considered hazardous in one country may not necessarily have the same status in another country. Hjellness (2008) indicates that all of the Norwegian samples of shredder residue shown in table 5.4 exceed the Norwegian criteria for hazardous waste (Cu,

Pb and Zn). Four of the eight samples also exceed the hazardous waste criteria for hydrocarbons (C12–C35), also referred to as “heavy oil”, see table 5.5. In Denmark, shredder residue has been classified as hazardous waste for more than a decade. The basis for this classification has not been very well documented and a study to review the classification has recently been initiated (Force Technology, 2009). However, Isager (2009) states that the classification of SR as hazardous waste in Denmark was based on the content of heavy hydrocarbons which did not – as probably assumed – originate from heavy oil, but rather from the rubber in tyres. The classification as hazardous waste has favoured landfilling of SR in Denmark, because landfilling of hazardous waste has been exempted from the landfill tax of 375 DKK per tonne (this exemption is likely to be phased out, starting in 2010). As a result of the exemption, all shredder waste produced in Denmark is being landfilled.

Force Technology (2009) states that the classification of shredder residue as hazardous or non-hazardous is likely to depend primarily on the content of heavy metals, PCBs and oil.

If shredder residue is classified as a non-hazardous waste, it may, in principle, be landfilled in accordance with Council Decision 2003/33/EC, provided it has undergone treatment. If it is classified as a hazardous waste, it must be treated further to reduce the high content of TOC, which with a value of 18 % (w/w) (table 5.4) or higher (table 5.6) exceeds the waste acceptance criterion for TOC (6 % (w/w)) for hazardous waste to be landfilled at a landfill for hazardous waste. The leaching of DOC (approx. 1500 mg/kg at L/S = 10 l/kg for the two samples in figure 5.3c) similarly exceeds the EU WAC for waste to be accepted at a non-hazardous landfill receiving stable, non-reactive hazardous waste (800 mg/kg at L/S = 10 l/kg) and at a landfill for hazardous waste (1000 mg/kg at L/S = 10 l/kg). The Danish shredder residues shown in section 5.1.2 comply with the other WAC for the two types of landfill, whereas the leaching of Pb from one of the Norwegian shredder residue samples exceeded the WAC for acceptance of hazardous waste at a landfill for hazardous waste.

From the point of view of compliance with the EU WAC for landfilling, the requirement for treatment of SR hence depends on the classification of the waste. If the SR is classified as non-hazardous, no treatment is required, because there are no composition or leaching WAC for landfilling in non-hazardous waste landfills at EU level. In Denmark, the same is true for acceptance of waste at landfills for mixed (non-hazardous) waste, although the waste must not be suitable for incineration. If, on the other hand, the SR is classified as hazardous waste, TOC and DOC need to be reduced substantially. In addition, it may be necessary to reduce the leaching of certain heavy metals/trace elements (in particular Pb).

For SR classified as a hazardous waste, the objective of treatment might be to change the classification to non-hazardous (and hence remove the WAC requirements in relation to landfilling). The treatment target and methods will depend on the properties which have given rise to the hazard-

ous classification; based on the above, it may in this case most likely be necessary to reduce the content of heavy metals, PCBs or oil.

There might be other incentives than compliance with WAC to treat shredder residue prior to landfilling, regardless of the classification as hazardous or non-hazardous waste. The most common motive for pre-treatment of SR is probably to increase the amount of materials and resources recovered (metals, plastics, energy). Another motive could be to increase the sustainability of shredder landfilling through reduction of the aftercare period. This could possibly be achieved by reducing the content of substances that would pose a threat to the environment over a longer period of time and by giving the treated SR properties that would enhance the envisioned stabilisation processes after landfilling. Such treatment would most likely have to be combined with operational measures to be successful.

5.3 Current SR management in the Nordic countries

The current status of SR production, classification and management in the Nordic countries is summarised in table 5.8.

Table 5.8 Overview of production and management of SR in the Nordic countries (sources of information: the members of the Nordic Landfill Group).

Country	SR production, t/y (year)	Classification	Main form of disposal
Denmark	Approx. 200,000 (2008)	Hazardous waste (under revision)	Landfilling
Faroe Islands	None	Not relevant	Metal containing waste exported for shredding (mainly to DK)
Finland	70 000	Partly classified as non-hazardous waste, partly as hazardous waste	Landfilling
Iceland	< 5000	Non-hazardous waste in most cases	If classified as non-hazardous waste, it is used as a daily cover material at landfills
Norway	100,000	Non-hazardous waste	Landfilling + a small amount for incineration
Sweden	Approx. 160,000 (2008) (50% "fluff" and 50% "fines" (= particles < 7 to 10 mm))	Non-hazardous waste	"Fines" are mainly used for construction purposes at landfills, etc. "Fluff" is disposed of through 50% landfilling and 50% incineration.*

*: The "fluff" has previously been disposed of exclusively by landfilling. An increase in available incineration capacity has resulted in more incineration of this fraction. It is, however, uncertain how this situation will develop in the future.

5.4 Overview of potential treatment methods for SR

This overview and evaluation of treatment methods for shredder waste is based on a number of studies and reviews conducted during the past eight to ten years. Some of the more comprehensive and useful studies upon which this work is based are reported by Cramer et al. (2006), Nielsen et al. (2006a), GHK (2006), Zevenhoven and Saeed (2003) and Jody and Daniels (2006). Cossu & Gadia (2007) has produced a short review based primarily on GHK (2006) and Zevenhoven and Saeed (2003).

Most – if not all – treatment methods for SR for which information is available have been developed not specifically to improve the properties of the material to be landfilled but rather to increase the recovery of resources from the shredding process (and at the same time reduce the amount of treated shredder waste to be landfilled). The main focus has therefore often been on the recovered products and resources, and the information available on the environmental properties of the waste streams going to landfilling is generally poor. The uncertainty of the economic evaluations is substantial due to differences in pre-conditions and fluctuations in energy and product prices. In addition, the technology has developed considerably during and after the period covered by the studies. This must be born in mind when evaluating the performance of the different treatment methods.

Most of the treatment methods that have been applied to shredder residue belong to one of the following types (see also chapter 3):

- Mechanical separation
 - Thermal treatment
 - Incineration
 - Co-incineration
 - Gasification
 - Pyrolysis
- Combinations of mechanical and thermal treatment

Gasification and pyrolysis based methods are often combined with melting/vitrification of the slag.

5.5 Technical description of treatment methods

5.5.1 Mechanical separation

Typical unit operations

Mechanical separation may in many respects be viewed as a natural extension of the shredding process that produces the SR, or it may actually be integrated into and become part of the main shredding process. Many of the unit operations applied are the same. They include cutting, crushing, drying,

screening, sifting, magnetic and eddy current separation, air classification, sink/float separation. Some of these processes are exploiting density differences between the various materials present in SR. Figure 5.4 shows some typical specific densities of a number of materials found in SR. A separation based on density differences allows for separating the organic (plastics) and inorganic (metals, glass) fractions in SR (Zevenhoven & Saeed, 2003).

The separation of various fractions of SR may in some cases be controlled by advanced identification techniques including infrared separators, color sorters, X-ray and UV fluorescence sorters. These and the other processes are also applied “in-house” by many shredder companies.

Mechanical processing can give recovered fractions that may be recycled, or can be used to improve the quality of SR by removing harmful substances, or increase its value as a waste/refuse derived fuel (RDF). In the following, a number of mechanical separation technologies aiming primarily at recycling and recovery of materials are presented.

Figure 5.4 Densities of various materials found in SR (Sattler & Laage, 2000). PUR = polyurethane, PA = polyamides, ABS = acrylonitrile-butadiene-styrene plastic, PP = polypropylene.

The WESA-SLF process

Density separation plays an important role in the WESA-SLF process which is shown schematically in figure 5.5

Figure 5.5 The WESA-SLF process for mechanical separation of SR (Sattler & Laage, 2000).

According to Zevenhoven & Saeed (2003), a 4 t/h pilot plant is or has been operating at Eppingen in Germany. It treats automotive shredder residue (ASR) which is reduced to a size less than 7 mm to ensure that insulation is stripped off copper wires. A fraction < 1.2 mm, consisting mainly of sand, is screened off as the first step. The following products are subsequently obtained from the process (Sattler & Laage, 2000):

- A magnetic fraction with approximately 95 % iron,
- Copper granules/chaff
- A mixture of mineral materials with some metals and
- An organic fraction, with the following contents and properties:
 - Metal, each < 0.5 %
 - Carbon ~ 50 %
 - Hydrogen ~ 6 %
 - Oxygen ~ 12 %
 - Chlorine 1–2.5 %
 - Ash content 20–28 %
 - Calorific value 23 MJ/kg

The two first product streams were sold to the metal industry, and the third stream was further processed to recover copper and aluminium. For the fourth stream use as a carbon source in steel plants or use for methanol production has been suggested. Sattler & Laage (2000) report processing costs of 61 to 76 €/t for the 4t/h or 16000 t/year facility with an investment of 2.04 million €. No information on leaching properties of the waste streams is available.

The VW-SiCon process

The VW-SiCon process involves multistage shredding of SR in combination with sorting and segregation on the basis of physical criteria, such as density, particle shape, magnetic properties, conductivity and optical characteristics. In the sink/float section of the process, the fluid and the plastics mixture are fed to the mixing tank separately by using a metering pump and a metering screw. The tank is equipped with an agitator that keeps the plastics suspended in solution. The mixed slurry then enters the separation unit, in which some plastics sink and some float. The fluid used in the separation process is recovered and reused (Jody & Daniels, 2006). The output streams from the process are shredder ferrous and non-ferrous metals, granules (plastics), shredder fibres which may be sold, and sand, sludge and dust which must be disposed of by landfilling and/or incineration. A mass balance for the process is shown in figure 5.6.

Figure 5.6 Mass balance flows for the VW-SiCon process (GHK, 2006).

Figure 5.7 shows the estimated costs (negative) and incomes (positive) for each input and output stream. The figure is based on a treatment plant with an annual capacity to treat 100,000 t of SR. Investment costs are estimated to be between 6 million € (without separation of plastics and with minimised features) and 12 million € (including all features and with maximum security) in GHK (2006). According to SiCon GmbH (2009), a plant for treatment of up to 20,000 t/year of SR has been in operation in Belgium since 2008. No information of the leaching properties of waste streams is available.

Figure 5.7 VW SiCon process cost estimates per tonne of SR (GHK, 2006).

The Galloo process

Galloo Recycling is a multi-step shredding and SR treatment company that operates in France and Belgium. The Galloo Plastics stage can treat a plastic concentrate from the shredder treatment and separate polyethylene and polypropylene from polystyrenics. The outline of the process is shown in figure 5.8 indicates that 48 % of the treated SR must be landfilled (GHK, 2006).

Figure 5.8 Mass balance flows for the Galloo process (GHK, 2006).

It is difficult to find reliable cost estimates for the Galloo process, and no information on the leaching properties of waste streams has been found.

The R-Plus process

The R-Plus process treats SR mechanically by sifting and density separation, see figure 5.9. It produces three fractions: Metals (ferrous, non-ferrous, copper) which are sold, a mineral fraction (sand, glass, etc.) which is used in the construction industry and an organic fraction which is used as feedstock for for energy recovery or chemical processes.

R-Plus incurs costs for processing the SR and for disposal of the mineral (transport costs only) and organic fractions. Total costs are 120 €/t of SR, the income from sale of metals is 30 €/t of SR, hence net costs for the treatment are 90 €/t of SR. this information is reported by GHK (2006) who further reports that R-Plus Recycling GmbH runs one post-shredder plant in Germany. A simple mass balance flow diagramme is shown in figure 5.9. No information can be found on the leaching proerties of the minerals stream.

Figure 5.9 Mass balance flows for the R-Plus process (GHK, 2006).

Other mechanical separation processes

In addition to the above mentionen processes, several other technologies based on mechanical separation exist. These processes include e.g. the Saly process for separation of mixed plastics (Jody & Daniels, 2006) and the Sult process, which is also based on sifting and density separation. GHK, (2006) reports that the Sult process produces steel (7%), copper (3%), minerals (15–20 %), water vapour (20 to 25 %) and an organic fraction (50 %) that can be incinerated or partly recovered as plastics. A gate fee of 100 €/tonne of SR residue is estimated (GHK, 2006).

Overview of mechanical processes

An overview of some of the mechanical separation processes described above is given in table 5.8. There is no information available on the leach-

ing properties of the waste streams after mechanical separation and materials recovery.

Table 5.8 Overview of some post shredder technologies based on mechanical separation (after GHK (2006)).

Name of technology	Level of technology development (2006)	Approximate outputs from the process	Indicative gate fee € per tonne of SR
WESA-SLF	4t/hr pilot plant	Ferrous stream, copper granules, mineral waste, organic fraction	61–76
VW-Sicon	1 trial plant (8000 t) + 2 under construction	Shredder granules 36%, shredder fibres 31%, metals 8%, wastes 26%	20 –50
Galloo	Operating plants	Recycled plastics 9%, metals 30%, RDF 13%, wastes 48%	Not available
Sult	Operating plant in Japan	Organic (plastic) 50%, minerals 20%, metals 10%, water 20%	100
R-Plus	Operating plants	Organic fraction 60%, metals 5%, minerals 35%	90

5.5.2 Thermal treatment: Incineration and co-incineration

Incineration/combustion is defined as the complete oxidative conversion of a fuel in order to produce heat (and/or to reduce the volume of a waste material). The stoichiometric factor for oxidation (the air factor) is higher than 1 (Zevenhoven & Saeed, 2003). Typical combustion reactors for low-grade fuels and wastes are based on grate firing, kiln firing and fluidised bed combustion (FBC). The latter often requires pre-treatment of the waste to refuse derived fuels (RDF) in the form of pellets.

Incineration of pure SR or large proportions of SR mixed with other wastes does not seem attractive due to the chlorine which may prevent the production of high temperature steam and the relatively high content of trace elements such as Zn, Pb, Ni, Cd, Sb, As and Sn which will be transferred to the incineration residues (APC residues and bottom ash) and possibly increase the leaching of contaminants from these residues (Zevenhoven & Saeed, 2003), thereby causing problems in relation to landfilling and/or utilisation. In a study of fly ash from three Danish MSW incinerators (Hjelmar, 1993), it was shown that the leachability of Cd, Cr, Pb and Zn from the fly ash from one of the incinerators which during a period received 30 % SR in the feedstock was considerably higher than it was from the fly ash from the other MSW incinerators which did not receive any shredder waste (see table 5.9).

Table 5.9 The influence of incineration of a large proportion of SR on the leaching of certain contaminants from MSWI fly ash (Hjelmar, 1993).

Element	Unit	Amount leached at L/S = 25 l/kg (combined column and batch test)		
		Incinerator 1 (70% MSW + 30% SR)	Incinerator 2 100% MSW	Incinerator 3 100% MSW
Cd	mg/kg	35	0.051	7.3
Cr	mg/kg	0.12	0.053	0.081
Pb	mg/kg	370	19	< 1
Zn	mg/kg	580	< 2	< 3

Co-combustion or co-incineration of SR, most importantly with MSW, is an option that has been investigated at several locations in Europe. Zevenhoven & Saeed (2003) reports that co-incineration with at least 95 % MSW in Swiss and German incinerators has been the “short term” solution to ASR management in Switzerland since the mid 1990s. The objectives of co-incineration are the same as for MSW incineration: mineralization and reduction of the mobility of the inorganic fractions and destruction of the organic fraction with production of heat and electricity.

DTU in Denmark has performed full-scale testing of co-incineration of shredder waste with MSW at a Danish MSW incinerator and tested the influence on the stack gas, the fly ash and the bottom ash. Over a period of 10 to 12 hours, a mixture of 14 % of SR and 86 % of MSW was fed to the incinerator and samples of stack gas fly ash and bottom ash were collected after a certain period of time when it was ensured that they were representative of the SR induced conditions (Astrup, 2009a). The changes in waste input significantly changed the composition and leaching characteristics (at L/S = 2 l/kg) of the bottom ash (Astrup, 2009b). The most pronounced increases in content of the bottom ash as compared to the baseload situation were seen for Ba, Cu, Mo, Ni, Sb, Sn and Zn (Hycks, 2009). As could be expected, there was no direct relationship between bottom ash composition and leaching. For fresh bottom ash, the effects of SR on leaching could be both positive and negative, depending on the substance. In general, ageing/carbonation would reduce the leachability of several substances, and the difference in leaching between aged/carbonated bottom ash from the SR incineration and the reference bottom ash appear to be limited. Based on the results, Astrup (2009c) expects co-incineration of 5 % SR with MSW to have an insignificant effect on the leaching properties of the bottom ash. This is in agreement with information provided by Zevenhoven & Saeed (2003) that co-firing of 6 % SR with MSW at Bazenheid in Switzerland was “without severe problems”.

Zevenhoven and Saeed (2003) reports on earlier test runs at a German MSW incinerator where mixtures of MSW/ASR of 76%/24% (by weight), of 69%/31% and a 69%/31% of a coarse ASR fraction were fed to the incinerator over one day. Concentrations of dioxins/furans and trace elements such as Cd, As, Pb and Zn were increased by up to a factor of 6 in the gas but caused no problems for the gas cleaning equipment – the cleaned gas complied with regulations. The bottom ash showed higher concentrations of

especially Zn, but also Cu, Sb, Ni, Pb and Zn when compared to firing with pure MSW, but the leachability of contaminants complied with the German LAGA criteria for reuse, which were in force in the 1990s (LAGA; 1993).

The typical calorific values of 12 to 13 MJ/kg for SR (see section 5.1.1) is not very different from the normal calorific value for MSW. Pretreatment to remove inorganic constituents (e.g. metals, glass and sand) will tend to increase this value, whereas removal of organic constituents (e.g. plastics, rubber) will have the opposite effect. Both may potentially cause problems if the SR is (co-)incinerated, particularly if it constitutes a significant proportion of the waste feed.

The cost of co-incineration of shredder residue with MSW – possibly after improved mechanical removal of some of the inorganic fractions from the SR – should be of the same order of magnitude as incineration of MSW with combined production of electricity and heating (approximately 27 €/tonne under Danish conditions, excluding taxes, according to RenoSam (2005)).

The CTU process where shredder residue is combusted in admixture with MSWI APC residues has been discussed in section 4.5.5.

5.5.3 Thermal treatment: Gasification and pyrolysis

Gasification is defined as a thermo-chemical conversion of a carbon-containing material using a gaseous compound such as water, air, oxygen and mixtures of these, producing a gaseous product. The stoichiometric factor for oxidation (the air factor) is less than 1. Three main types of reactors are being used: 1): moving beds (updraft or downdraft), 2): fluidised beds and 3): entrained flow reactors. Particle sizes are of the order of 10 mm, 1 mm and 0.1 mm, respectively, for these types of reactors. Temperatures vary from 400–500°C to more than 1500°C, and residence times can vary from less than a second to more than an hour. The gasification process is exothermal (can in some cases be adiabatic) and produces a combustible product gas (fuel gas) and ashes. It is advantageous for treatment of shredder residue that no dioxins/furans are produced because of the starved air conditions, that metals may be recovered from the residues partly non-oxidised, that valuable gases are produced (may be used/sold externally or may be used to melt/vitrify the ashes), and that the volume of the product gas is small compared to the volume of flue gas from an incinerator (Zevenhoven & Saeed, 2003).

Pyrolysis is defined as a thermo-chemical conversion of carbon-containing materials under the absence of a reactive gaseous compound. The stoichiometric factor for oxidation (the air factor) is 0. Two types of pyrolysis processes can be distinguished: 1): Low temperature processes operating at 400–600°C (in Germany called “Schwelprozess”) and high temperature processes operating at 500–800°C. Advantages of pyrolysis processes are the relatively low temperatures, the production of fuels that can be used to drive the process and no formation of dioxins/furans or NO_x

due to the oxygen deficiency. Drawbacks of SR pyrolysis may be a risk of high PCB levels in the oil or char/solid products which may then have to be classified as hazardous waste, and, typical for pyrolysis, the formation of tars (Zevenhoven & Saeed, 2003).

The Watech pyrolysis process where shredder residue is subjected to pyrolysis together with MSWI APC residues has been discussed in section 4.5.5.

In a comprehensive Danish study, Cramer et al. (2006) conclude that Ebara/TwinRec and Takuma are the only two documented well functioning thermal and mechanical processes that are able to treat shredder waste. Both Ebara and Takuma have three and four years of production from one commercial plant (in Japan), and offer the plants with guarantees worldwide. The Ebara process consists of fluid bed gasification and recycling of metals. The Takuma process consists of pyrolysis in a rotary kiln, mechanical/magnetic separation of ferrous and non-ferrous metals and melting of minearals into a glassy slag. Cramer et al. (2006) compare and evaluate the two processes as follows:

- Both processes produce iron, copper and aluminium in mixtures from the SR, and both processes produce a mineral granulate that can probably meet the Danish criteria for utilisation. Most of the Pb and Zn end up in the flue gas cleaning product which must be disposed of.
- The treatment cost of both processes are expected to be approximately 120 €/tonne of SR.
- Both technologies can, without any problems, meet the Danish requirements on working environment and emissions.
- Both processes can treat other types of waste as well and can therefore be flexible in terms of capacity adjustment.

GHK (2006) reports that the TwinRec process (Ebara) produces metals (8%), glass granulates (25%), recovers energy (52%) and produces 15% of waste (mainly to be landfilled – within the EU that waste stream would most likely require pre-treatment prior to landfilling). The gate fee is reported at 120 to 200 €/tonne of SR and includes investment costs, processing costs, average profit, final disposal costs as well as income from sale of recovered metal fractions and energy.

5.6 Overview and discussion of treatment options for SR

The choice of management option for SR from a particular shredding operation may depend on several factors, including the feedstock to the shredding operation, the effectiveness of the shredding operation and the composition of the SR, the price level for recovered metals, plastics, glass and minerals, the availability of incineration and landfilling capacity as well as national waste management policies and waste taxation levels.

Some of the options that may help improve the materials and energy recovery and reduce landfilling and/or the long term effects of landfilling may be described as follows:

- Mechanical separation with extensive materials recovery or improved materials recovery integrated into the shredding operation followed by landfilling (if the residual waste stream is not suited for combustion) or co-incineration with MSW (if the residual waste stream is suited for combustion). Landfilling should be developed in the direction of increased sustainability and a reduced aftercare period, and the proportion of SR co-incinerated with MSW (with effective energy recovery) should probably not exceed 5 to 10 % (by mass) of the total feedstock to the incinerator. The bottom ash from the incinerator should be utilised if possible, and the air pollution control residues may require treatment prior to landfilling. If mechanically treated SR is classified as hazardous waste and destined for landfilling, care should be taken to ensure that TOC (and DOC) is sufficiently low to meet the WAC for landfilling.
- Co-incineration of SR with MSW (with effective energy recovery) without dedicated pre-treatment of SR (but possibly with improved materials recovery integrated into the shredding operation producing the SR). The proportion of SR co-incinerated with MSW should probably not exceed 5 to 10 % (by mass) of the total feedstock to the incinerator. The bottom ash from the incinerator should be utilised if possible, and the air pollution control residues may require treatment prior to landfilling (see chapter 4 on APC residue treatment). Co-incineration will reduce the TOC (and DOC) to a sufficiently low level to meet the WAC for landfilling.
- Treatment by gasification or pyrolysis including recovery of materials and energy, production of a granulate that may be utilised for construction purposes and pre-treatment and landfilling of air pollution control residues. The first part of the Ebara gasification process takes place in a fluid bed reactor which requires extensive pre-treatment of the SR, including size reduction. It should be noted, however, that much of the energy recovered generally is used for the melting of the slag both in the gasification and pyrolysis processes.

If materials and energy recovery from SR does not appear to be technically and/or economically feasible due to insufficient development of technology or low materials prices, it might be worthwhile to store or landfill the SR under conditions that would preserve the properties of the waste and facilitate future recovery (landfill mining) at a time when processing has become possible and/or more attractive.

It should be noted that some countries, e.g. Switzerland and Japan have banned landfilling of SR which must be mechanically and/or thermally treated (GHK, 2006). This has undoubtedly been a major driving force in

the installation of trial and commercial units for thermal treatment of SR in these two countries.

The approximate costs of the various treatment options discussed in section 5.5 are summarised in table 5.10 together with some estimated prices of landfilling within the European Union. The cost of landfilling varies substantially from country to country. Examples of low cost EU member states with respect to landfilling are Poland, the Czech Republic and Hungary, whereas e.g. UK, France, Belgium, Spain and Italy are examples of medium cost member states. Denmark, Sweden and Germany are high cost member states as far as landfilling is concerned. Within Denmark there are, however, substantial regional differences in landfill cost. The Netherlands falls between the medium and high cost EU member states (GHK, 2006).

Table 5.10 Estimates of costs (gate fees) for various treatment and management options for SR.

Type of treatment/operation	Costs (in €/tonnes of SR)	Source
Mechanical treatment	20 to 100	GHK (2006)
Co-incineration, with electricity and heat prod.	27* , 100**	RenoSam (2005), GHK (2006)
Gasification (starved air)	120 to 200	GHK (2006)
Pyrolysis (no air)	120 to 150	Cramer et al. (2003)
Landfilling, low cost EU member states	30 to 40 (midpoint = 35)	GHK (2006)
Landfilling, medium cost EU member states	50 to 80 (midpoint = 65)	GHK (2006)
Landfilling, high cost EU member states	90 to 140 (midpoint = 115)	GHK (2006)

* Estimated cost in Denmark, exclusive of waste tax.

** Estimated cost at new German incineration plants.

6. MSWI bottom ash

6.1 Waste characteristics

6.1.1 General characteristics

Bottom ash is a heterogeneous mixture of slag, containing varying proportions of glass and metal residues (see figure 6.1). The main chemical contents of bottom ash are oxides of silicon and aluminium. Other constituents are alkaline and alkaline earth compounds, chlorides, sulfates, ferrous and non-ferrous metals and their compounds and unburned organics. As a result of a rapid cooling process, vitreous material is formed.

Figure 6.1 Example of bottom ash after removal of coarse metal scrap.

The relative partitioning of elements into bottom ash depends mainly on the composition of the MSW fed to the incinerator, the volatility of the elements it contains, the type of incinerator and grate system applied and the operation of the combustion system. The mass and volume reduction of waste incineration causes an enrichment of a number of heavy metals in the bottom ashes compared to their concentration in the waste feed.

State-of-the-art MSWI plants typically produce between 200 and 250 kg of bottom ash per tonne of waste treated. Most published numbers include the grate siftings, which only recently (and only in some countries) have been kept separate from the bottom ash. Requirements concerning the quality of the residues from the incineration process are included in European incineration legislation. Directive 2000/76/EC (Art. 6.1) includes an opera-

tional condition requiring that incineration plants achieve a level of incineration such that, in slag and bottom ashes, the loss on ignition is $\leq 5\%$ or the TOC is $\leq 3\%$. In modern well-operated MSWI plants the TOC in bottom ashes can be below 1 wt %. Combustion trials have demonstrated that an increase in heating value of the waste feed and resulting higher bed temperatures improve the burnout of bottom ash (Vehlow, 2002).

In modern MSWI plants, the water content of produced bottom ash ranges from 15% to 25% due to the quenching. This moisture content is an important factor for dust control and has a major effect on the physical processing of bottom ash. Bottom ash particles are distributed over a range of sizes. The particle size distribution is important because many processes, such as metals recovery, depend very much on the particle size. In general, up to 40% of bottom ash by weight is smaller than 2 mm and up till 80% by weight is smaller than 10 mm.

Bottom ash is a very porous lightweight aggregate with high specific surface areas. It has dry bulk density of approximately 1500 kg/m^3 ($1500\text{--}2000 \text{ kg/m}^3$ for the fine fraction ($<100\text{mm}$) and $1800\text{--}2400 \text{ kg/m}^3$ for the coarse fraction ($>100\text{mm}$)) (IAWG, 1997, Wiles 1996). A common feature for all particles is gas bubbles, most likely formed during boiling of the melt products. Bubbles comprise between 10–25% of the volume of the particles, therefore providing for a significant measure of particle porosity (Kinnunen 2006).

6.1.2 Chemical composition

Comprehensive data on the chemical composition of MSWI bottom ash are given in Appendix 4.

The elements present in a proportion higher than 1% in the ash are: Si, Fe, Ca, Al, Na, C and O, which is the most predominant element (ca. 40%). Approximately 80–90% by weight of the bottom ash consists of these major elements. Many of the elements are present as oxides and oxygen is therefore also a major element for all the residues. Minor constituents (0.1–1 %) include Ti, Cl, Mn, Ba, Cu, Mg, K, Zn, Cr, Pb, Mn and Ni. Some of the minor elements and many of the trace elements (e.g., Pb, Cu, Zn, Cd and Hg) are enriched in the bottom ash (IAWG, 2007).

The trace elements/heavy metals can be classified according to their degree of volatility. During incineration process the elements with high boiling temperature remain in the bottom ash, while more volatile elements with low boiling temperature are end up in the air pollution control residues. During the incineration Fe, Cu, Cr, and Al and Si remain mainly in bottom ash. Iron and aluminum are mostly fed to incinerators in metallic form and their oxides are very stable and no volatilisation occurs during incineration of MSW. The behavior of copper on incineration of MSW is similar to that of iron. Also Cr compounds are not considered thermally mobile during incineration and therefore they remain mainly in bottom ash. Mg and Al have the same tendency. Ni, a less volatile heavy metal, is more concen-

trated in the bottom ashes. However, two thirds of Pb, Zn and As remain in the bottom ash despite their high volatility (Kinnunen 2006).

An elevation of the combustion temperature, together with the fuel bed temperature, is reported to cause increased formation of CaO in the bottom ash. This causes an increase in the pH value of the bottom ash. The pH value of fresh bottom ashes often exceeds 12 due to calcite content and is therefore subject to carbonisation (Vehlow, 2002).

6.1.3 Leaching properties

The leachability of constituents from bottom ashes can vary significantly depending on the type of waste and the combustion technology. As compared to stony or inert materials, the following constituents may be considered critical for MSW bottom ash: Cu, Zn, Sb, Mo, chloride, and sulphate. Treatment techniques aim to reduce the leachability of these critical compounds (BREF, 2005), particularly in relation to utilisation.

In table 6.1, the leaching of bottom ashes at L/S = 10 l/kg (Lapa et al. 2002, Pfrang-Stotz et al. 2000, Kaartinen 2004, Zijlstra et al. 1994, Laine-Ylijoki et al. 2005, Flyhammer 2006) are compared to EU waste acceptance limits for non-hazardous waste landfill. Examples of critical constituents that may exceed the EU leaching WAC are chloride, fluoride, copper, antimony and DOC. Similar information on a large number of Danish bottom ashes leached at L/S = 2 l/kg are presented in table 6.2. With the exception of DOC for a few samples they all comply with the EU WAC for non-hazardous waste landfills receiving stable, non-reactive hazardous waste.

Figures 6.2a to 6.2c shows the results of column leaching tests performed on 6 Danish bottom ashes (after 1 to 3 months of storage) from three different MSW incinerators (Hjelmar et al., 2006). The results are shown as accumulated leached amounts as a function of L/S. The horizontal lines indicate EU leaching WAC for hazardous waste landfills and for non-hazardous waste landfills receiving stable, non-reactive hazardous waste. Lines with slope = 1 have been inserted into the figures. A slope of 1 for the release curve for a substance is an indication of solubility control, i.e. the concentration of the substance is the same in consecutive eluates from the column test. Note that plotting the release of a substance with a concentration level below the detection limit as a function of L/S using the detection limit as default values would also give a slope = 1.

Table 6.1 Comparison of some reported leaching results at L/S = 10 l/kg for MSWI bottom ashes to EU WAC for landfills for non-hazardous waste. DM = dry matter.

Parameter	Unit	European bottom ashes	German bottom ash	Swedish bottom ash	EU WAC criteria for landfill for non-hazardous waste
No. of samples		7	26–28	21–42	
Test		EN12457–2 (<4 mm)	DIN 38414 (DEVS4) (<10 mm)	not known	EN 12457–2,3, 4
L/S	l/kg DM	10	10	10	10
pH	-	8,9–12,5	9,5–11,4	-	
As	mg/kg DM	<0,02–0,34	0,0003–0,01	not determined	2
Ba	mg/kg DM	not determined	not determined	0,35–5,1	100
Cd	mg/kg DM	<0,02	0,001–0,037	0,001–0,02	1
Cr	mg/kg DM	<0,20–3,20	0,3–0,88	0,01–0,78	10
Cu	mg/kg DM	0,12–14,85	0,1–1	0,39–16	50
Hg	mg/kg DM	<0,012–0,230	0,0003–0,002	not determined	0,2
Mo	mg/kg DM	not determined	0,07–9,7*	0,32–1,4	10
Ni	mg/kg DM	<0,2	0,01–0,22	0,02–0,25	10
Pb	mg/kg DM	<0,5–10,2	0,005–0,37	0,003–0,22	10
Sb	mg/kg DM	not determined	not determined	0,20–0,80	0,7
Se	mg/kg DM	not determined	not determined	0,01–0,26	0,5
Zn	mg/kg DM	<0,50–14,32	0,09–2	0,02–1,2	50
Cl-	mg/kg DM	900–42 500	440–1 700	440–5 200	15000
F-	mg/kg DM	2,0–1 300	not determined	not determined	150
SO42-	mg/kg DM	66–1 573	2 300–7 690	2 900–14 000	20000
DOC	mg/kg DM	192–1 517	not determined	not determined	800

*Column test (NEN 7343) in L/S-ratio 10 l/kg.

Table 6.2: Results of leaching tests on Danish MSWI BA samples in terms of leached amounts at L/S = 2 l/kg (EN 12457–1). Source: Hjelmar and Hansen (2009).

Parameter	Minimum mg/kg	Maximum mg/kg	Average mg/kg	Median mg/kg	EU WAC criteria for landfill for non- hazardous waste	N	n be- low DL
As	0.0002	0.0088	0.0031	0.0026	0.4	70	10
Ba	0.047	0.12	0.079	0.074	30	14	0
Cd	0.0001	0.014	0.00074	0.00040	0.6	70	29
Cr	0.0042	0.24	0.029	0.019	4	70	0
Cu	0.099	9.6	1.5	0.98	25	70	0
Hg	0.00004	0.00021	0.000067	0.000040	0.05	7	3
Mo	0.19	0.93	0.52	0.56	5	17	0
Ni	0.001	0.11	0.013	0.0087	5	70	3
Pb	0.00064	0.037	0.0043	0.0028	5	70	0
Sb	0.056	0.083	0.070	0.070	0.2	6	0
Se	0.007	0.0097	0.0080	0.0081	0.3	6	0
Zn	0.00048	0.11	0.016	0.010	25	70	6
Chloride	500	1900	1100	950	10 000	70	0
Fluoride	1.5	30	11	10	60	6	3
Sulphate	450	2800	1500	1500	10 000	70	0
DOC	1	490	150	140	380	65	0
pH	9	11				70	

N: number of MSWI BA samples tested n below DL: number of results below the detection limit – these results were included in the statistical analysis using the detection limit as the test result

Figure 6.2a Results of column and batch leaching tests on 6 Danish MSWI bottom ashes from three different incinerators (Hjelmar et al., 2006).

Figure 6.2b Results of column and batch leaching tests on 6 Danish MSWI bottom ashes from three different incinerators (Hjelmar et al., 2006).

Figure 6.2c Results of column and batch leaching tests on 6 Danish MSWI bottom ashes from three different incinerators (Hjelmar et al., 2006).

Examples of the effect of pH on leaching of selected substances from bottom ash (Kaartinen 2004) are presented in table 6.3.

Table 6.3 Examples of the influence of pH on leachability of selected trace elements.

Parameter	Leaching (µg/l)	pH-area with maximum leaching	pH-area with minimum leaching	pH-area with the strongest changes
As	0,1–10	-	-	-
Cd	0,1–10 000	0–4	>10	4–9
Cr	1–10 000	<2	>8	0–6
Cu	100–100 000	<4, >11	7–11	0–4, >11
Mo	10–1 000	>7	<7	4–7
Ni	1–10 000	Low	High	Linear [horizontal?]
Pb	1–100 000	<8, >10	8–10	2–8, >10
Zn	10–1 000 000	0–4, >11	9–11	5–9, >11

6.2 Classification, challenges and treatment objectives

The classification of bottom ash as hazardous or non-hazardous is usually based on total content of hazardous compounds (e.g. metal). Also the ecotoxicity properties of the waste can be used as the base for the evaluation according to the Hazardous property H14. However, only a few ecotoxicity studies have been published (Lapa, 2002). The toxicity depends on the test battery and also on the dilution of the test medium used in the ecotoxicity testing. In most European countries, bottom ash is classified as a non-hazardous waste.

Bottom ash generally fulfills the waste acceptance criteria for waste to be disposed of at a landfill for non-hazardous waste. As mentioned in section 6.1.3, only chloride, fluoride, copper, antimony and DOC may in some cases appear as critical (under very alkaline circumstances, Pb may test high in the leaching test, and under oxidising conditions the same may happen for Cr). Bottom ash in Europe is generally acceptable for landfilling at non-

hazardous waste landfills for inorganic waste. Due to the enhanced leachability of especially copper in the presence of dissolved organic carbon, it is recommended to landfill bottom ash at landfills for inorganic waste. Prior to landfilling, valuable metals (e.g. iron scrap, aluminum) are usually to be recovered.

In many European countries, bottom ash is seen as an economically attractive alternative to natural materials due to its geotechnical properties. However, leaching of chloride, sulphates and oxides of alkali and heavy metal components can cause environmental problems. It is therefore generally necessary to upgrade bottom ash through physical and/or chemical treatment prior to utilisation. The treatment methods may in some cases generate a fine residue which does not fulfill the waste acceptance criteria for non-hazardous landfill. No literature reports were, however, found on this.

6.3 Current MSWI BA management in the Nordic countries

The current status of production and management of MSWI bottom ash in the Nordic countries is described in table 6.4.

Table 6.4: Current MSWI BA production and management in the Nordic countries.

Country	BA production t/a (year)	Utilisation, t/a	Landfilling, t/a
Denmark	498,000 (2006)	496,000	2000
Faroe Islands	Approx. 3000	0	3000
Finland	60,000*	Classified partly as HW and partly as Non-HW. Planned to be used for construction purposes at landfills	Most landfilled
Iceland	4,700 (2007)	2300	2400
Norway	150,000 to 200,000 (2006)**	0	Most landfilled
Sweden	693,149 (2008)	Approx. 589,000***	Approx. 104,000***

*: In Finland, the MSWI bottom ash production will increase significantly (to several times the current production) due to changes in the management of municipal waste.

** : The production of MSWI bottom ash in Norway will increase to approx. 250,000 tonnes in 2013.

***: In Sweden, approximately 15% of the MSWI bottom ash produced is landfilled, 7% is sorted out as metal and 78% is reused for road construction (mainly within waste treatment facilities).

6.4 Overview of potential treatment methods for MSWI BA

The simplest way of improving the bottom ash quality is to optimise the combustion parameters in order to bring the fixed carbon combustion to completion. This can be done using one or more of the measures such as longer exposure of the waste to elevated temperatures in the combustion chamber, higher bed temperatures and physical agitation of the waste all combine to ensure that ashes produced are lower in organic species (BREF 2005).

The organic content of the bottom ash is expressed as TOC or LOI (loss on ignition). These are key parameters for both the disposal and the use of bottom ash. Acceptance criteria for landfills generally state a maximal TOC level; and utilisation criteria generally state either a maximum TOC level or specific limit values for organic compounds. Improving the burnout will lower the residual carbon content and thus the TOC. The TOC is also related to the mobility of heavy metals in the ash. For instance, copper leaches in the form of organo-copper complexes. Improving the burnout will, therefore, also reduce copper leaching (BREF 2005).

Removal of ferrous and non-ferrous metals produces a better quality bottom ash. The common physical techniques used in recycling industry are size separation, density separation, magnetic separation and eddy current separation. Magnetic separation has been used for recovery of ferrous metals from bottom ash for a long time already, recently also eddy current separation has been applied in bottom ash processing for recovery of non-ferrous metals (Xing 2004 & 2006, Lo 2005).

Physical processing generates four streams (Pretz *et al* 2000):

1. Ferrous metal product,
2. Non-ferrous metal product,
3. Mineral residue e.g. glass, ceramic, porcelain, (so called granulate material), and
4. Wastewater, if the process is wet.

The value of the product is based on the metal content and on the world market price. For the nonferrous metal product, if the product contains excessive quantity of impurities, the price paid is reduced by the cost of separation of the impurities and disposal of these materials. In addition to aiming for the maximum possible nonferrous metals yield, measurements must also be implemented to ensure acceptable purity of the product.

In the Netherlands, AVR has carried out a research study together with Heros and TNO in order to find the right combination of treatment techniques to improve the environmental quality of the bottom ash. (especially leaching of copper, antimony, molybdenum and salts). The research study covered the following (Rutten *et al.*, 2006):

- improved iron and nonferrous metals removal
- accelerated ageing
- separation of the bottom ash into different particle size fractions
- washing/removal of salts
- stabilisation/solidification

The following techniques were considered to improve the quality significantly:

- ageing and washing
- ageing, fractionation and stabilisation
- ageing, fractionation, partly washing and stabilisation

The main goal of ageing is the decrease of the pH, caused by carbonation of oxides and hydroxides. This will lead to a decrease of the pH, caused by carbonation of oxides and hydroxides. This will also decrease the release of several metals. This is not only an effect of pH-decrease, but also of mineralisation leading to a better fixation of the heavy metals in the ash matrix.

The ageing experiments showed that the humidity of the ash influences the processability of the ash. A low humidity has a positive effect on the processing of the ash. The humidity of fresh bottom ash is about 20 %. After the aeration and ageing the humidity has dropped to below 15 %.

The following effects are important:

- a high moisture content of the ash causes higher treatment costs for the removal of fluff, because wet ash particles stick to not burned organic material (fluff). A volume reduction of 50 % can be realised by lowering the humidity
- a high moisture content leads to a lower grade of the nonferrous metals
- a high moisture content of the ash interferes with the segmentation of the ash particles, which makes the separation of iron and nonferrous metals and the non burned particles more difficult.

Thermal treatment is used for the treatment of bottom ashes, as well as for combinations of bottom ash and ashes from flue gas treatment. Thermal treatment is used extensively in a few countries, mainly to reduce the volume and to improve their leaching properties. High temperature treatments use heat in order to melt waste and initiate vitrification and ceramisation processes. Thermal treatments can be grouped into three categories: vitrification, smelting and sintering (see further explanation on the differences of the methods in 3.2.4). The differences between these processes chiefly relate to the characteristics and properties of the final material (BREF 2006). Table 6.5 summarises the most commonly used treatment techniques applied to MSWI bottom ash.

Table 6.5 Overview of different treatment methods for MSWI bottom ash used in Europe.

Method	Purpose	Techniques (example of potential techniques)
Chemical		
Storage/ageing	To allow carbonation and thereby lower the pH and reduce adverse leaching effects.	Accelerated ageing e.g. in a bunker: A ventilation system and an installed water circulation system
Washing	To reduce the easily water soluble fraction of the bottom ash	Typically batch process (e.g. ash quench tank at furnace exit), minimum L/S ratio 0.35 l/kg Note! The large amount of water needed is a drawback and means that the process is only economically feasible if washing water is available against low prices and the effluent can be discharged into brackish or salt water, i.e. removal of salts is not necessary.
Physical		
Size reduction	To increase of surface available for chemical reaction	Crusher (jaw, impact, and roll crusher)
Screening	Separation of particle population in two or more size fractions for removal of fine materials, optimisation of processability or separation of specific fractions (glass, metals)	Screens ranging from 300 mm down to 40 µm vibration screens (e.g. circular motion screens)
Separation based on density	Separation of valuable and non-valuable materials based on density.	Jig separator, shaking table and spirals separators. Two types: A relative separator splits the feed into a part of higher density and lower density (Here the cut density will depend on the feed). An absolute separator has a fixed cut density determined by constant density of medium
Magnetic separation	Recovery of ferrous metals based on the magnetic properties of materials	In recycling four main types of magnetic separators are frequently used for the separation of ferrous metals; cross belt, line belt, pulley and drum separators
Eddy Current separation	Recovery of non-ferrous metals (e.g. Al, Cu) from a mixture and for the separation of different non-ferrous metals from each other	Fraction typically > 5 mm: traditional eddy current separator, fraction typically < 10 mm: Magnus separator, wet eddy current separator,

6.5 Technical description of methods

6.5.1 Physical separation of metals

Both ferrous and non-ferrous metals may be separated physically from bottom ash. Ferrous metals separation is performed using a magnet. The ash is spread out on a moving belt or vibrating conveyor and all magnetic particles are attracted by a suspended magnet. This ferrous metals separation may be performed on the raw ash after leaving the ash extractor. Efficient ferrous

metals separation requires a multi-step treatment with intermediate size reduction and screening.

Non-ferrous metal separation is performed using an eddy current separator. A rapidly rotating coil induces a magnetic field in non-ferrous particles, which causes them to be ejected from the material flow. The technique is effective for particle sizes of 4–30 mm and requires a good spreading of the material on the moving belt. The separation is performed after ferrous metals segregation, particle size reduction and screening.

The amount of recovered metals depends on the composition of the waste input. For ferrous metals, data from Belgium suggest a recovery rate of 55–60 % (mass of metal recovered/mass of metal input) (Vrancken, 2001). This recovery rate is supported by data from IAWG (1997), which gives a residual ferrous content of 1.3 % to 25.8 % in bottom ash that is considered for utilisation or disposal.

For non-ferrous metals, using eddy current separation after size reduction and screening, allows a 50 % recovery rate (mass recovered/mass input). The actual value is dependent on the operational conditions of the furnace. Non-ferrous metals, such as Pb and Zn, are found in the boiler ash and the flue-gas cleaning residue whereas Al, Cu, Cr, Ni preferentially stay in the bottom ash. Oxidation of these metals (e.g. Al to Al_2O_3) during combustion will hamper the effective separation by Eddy Current separators. The separated non-ferrous fraction shows the following composition: 60 % Al, 25 % other metals, 15 % residue. The other metals are mainly copper, molybdenum, zinc and stainless steel (Vrancken, 2001, BREF 2005).

Ferrous metals separation is performed at most European incinerators either on-site (mainly post combustion) or at external bottom ash treatment plants. Non-ferrous metals separation exists at various bottom ash treatment plants in Denmark, the Netherlands, Germany, France, and Belgium.

6.5.2 Screening and crushing

The typical mechanical treatment operations for bottom ash include the one or more of the following unit operations:

- granulometric separation by screening
- size reduction by crushing large elements or otherwise breaking them up
- air-stream separation to eliminate light unburned fractions.

Three types of screens are encountered:

- rotary or drum screens
- flat screens (vibrating or not)
- star screens: screening is achieved by movement over a series of rollers equipped with starshaped arms on each axis.

Primary screens used to prepare bottom ash aggregate are in most cases equipped with a mesh sized 40 mm in diameter. A crusher can be installed in the treatment line to break up large chunks, generally at the exit from the first screening. Half of the facilities are equipped with crushing apparatus, some use equipment at the site (shovel, loader, rock crusher, etc) to smash blocks. Separation of light unburned fractions or air stream separation is achieved by blowing or by aspiration (BREF 2005).

6.5.3 Ageing

During weathering, the carbonation reaction stabilizes bottom ash by changing its mineralogical characteristics. It involves the CO_2 dissolution in pore water at initially alkaline conditions. This causes pH to decrease and calcite to precipitate until the material is in equilibrium with CO_2 . This phenomenon is slow and can not be achieved in the regulated weathering period which often lasts between three and twelve months. To study carbonation in bottom ash, the carbonation process can be artificially accelerated. Several methods have been described in the literature (Rendek et al., 2006).

After metals separation, bottom ash is usually stored in the open air or in specific covered buildings for several weeks. The storage is generally performed in stockpiles on a concrete floor. Drainage and run-off water are collected for treatment. The stockpiles may be wetted, if required, using a sprinkler or hose system in order to prevent dust formation and emissions and to favour the leaching of salts and the carbonisation if the bottom ashes are not sufficiently wet. The stockpiles may be turned regularly to ensure homogeneity of the processes that occur during the ageing process (uptake of CO_2 from the air due to the moisture, draining of excess water, oxidation, etc.) and to reduce the residence time of every batch of bottom ash in the dedicated facilities. In practice an ageing period of 6 to 20 weeks is commonly observed (or prescribed) for treated bottom ash before utilisation as a construction material or in some cases before landfilling (BREF 2005 2005, Astrup & Christensen 2003, Steketee et al. 1997).

Fresh bottom ash is not a chemically inert material. Ageing is performed to reduce both the residual reactivity and the leachability of metals. CO_2 from the air and water from humidity, rain or water spraying are the main activities. Aluminium in the bottom ash will react with $\text{Ca}(\text{OH})_2$ and water to form aluminium hydroxide and hydrogen gas. The main problem of formation of aluminium hydroxide is the volume increase as this causes inflation of the material. The gas production will cause technical problems if fresh bottom ash is used directly for construction purposes. Thus, ageing is needed to allow utilisation of the bottom ash. The impact of storage and ageing on leaching can be classified as:

- lowering of the pH to around pH range 8–9 due to uptake of CO_2 from the air or biological activity

- establishing of anoxic, reducing conditions due to biodegradation of residual organic matter
- local reducing conditions due to hydrogen evolution
- hydration and other changes in mineral phases causing particle cohesion (IAWG, 1997)

A well-known technology is to accelerate the aging of the bottom ash by increasing the CO₂-uptake. This can be done by bringing bottom ash in contact with a gas with a higher CO₂-percentage than atmospheric air. By CO₂-enrichment the required period of aging will decrease significantly from 6-20 weeks to 1-2 weeks (Baun 2007). Accelerated aging is used in various bottom ash treatment plants in the Netherlands, Germany, France, and Belgium.

The ageing reduces the leachability of e.g. copper and causes a stabilisation of the bottom ash. The influence of ageing on the organic compounds is not clear, but it is suspected that the ageing will influence the degradation of organic compounds due to microbiological activity in the ash (Steketee et al. 1997).

6.5.4 Washing

Washing of bottom ash is generally seen as a highly efficient method for upgrading the quality of bottom ash. Washing is primarily used to remove highly water soluble constituents like salts and thereby reduce the leaching of both soluble salts and heavy metals. Furthermore washing may remove residual organic matter. The washing can be performed in different ways. The bottom ash may simply be washed in the ash quench tank or a similar tank just after the grate. By this method the content of soluble salts such as chlorides in the bottom ash can be reduced by more than 50 %. Simple washing will generally not reduce the leaching of sulphate, which is solubility controlled (by calcium sulphate, gypsum). If the smallest size fraction (e.g. 0-2 mm) is removed by washing, the leaching properties may be further improved (Baun et al. 2007).

Results of a Danish experiment with washing of bottom ash in the quench tank and different times of subsequent aging are shown in table 6.6 (Mogensen 2002 in Baun et al. 2007).

Table 6.6 Results of washing in the quench tank.

Parameter	Unit	No washing		Washing at L/S 1 l/kg	
Aging	months	2–3	7–8	2–3	7–8
pH		10	8.7	9.8	8.8
Chloride	mg/l	900	920	490	280
Sulphate	mg/l	1200	1800	920	1800
Ca	mg/l	460	710	270	710
Na	mg/l	560	540	330	200
As	µg/l	2.3	2.2	1	1.0
Cd	µg/l	0.6	0.3	0.7	0.2
Cr	µg/l	39	5	310	18
Cu	µg/l	1900	580	340	110
Ni	µg/l	9.7	11	< 3.3	92
Pb	µg/l	9.5	3	11	2
Zn	µg/l	<10	10	<47	<10
DOC	mg/l	100	31	23	11

Poorly soluble components e.g. heavy metals can only be partially removed by washing. Extraction of other trace metals requires a significant addition of acid to reduce the pH to acidic levels. The amount of water applied in the washing treatment is the major factor influencing process costs. For economical and environmental reasons the volume of washing water should be kept to a minimum, at the same time the keeping the solubility of the contaminants at a maximum. Although there are benefits from washing process, it tends to generate a problematic wastewater in which metals and salts are dissolved. The generated wastewater must be treated to reduce the concentration of contaminants prior to discharge (Kinnunen 2006).

6.5.5 Thermal systems

Various techniques have been adapted from the glass manufacturing and nuclear waste treatment industries for the thermal treatment of ashes. The temperatures applied are in the range of 1100 to 2000 °C. Much higher temperatures are sometimes employed for plasma systems. Plasma systems are used for the vitrification and melting of a variety of inorganic waste streams including bottom and fly ash – temperatures used for plasma arc vitrification are generally in the range of 1400 to 1500 °C, with the power supplied electrically. The molten products (i.e. slag and metal) may be continuously overflowed or intermittently tapped as required.

Plasma furnaces operate with power densities of 0.25 to 0.5 MW/m² and have melting rates of 300 kg/hr/m². Process footprints are usually small. Tolerance to feedstock variation is reported. Electrode consumption with DC plasma is reported at 2 kg/tonne of treated ash. FGT is required for the process off gases.

Many examples of thermal systems exist in Japan (estimated 30–40 plants). There is also some experience in Europe, e.g. France (BREF 2005).

6.6 MSWI bottom ash treatment installations

6.6.1 Dry bottom ash treatment installations

Dry bottom ash treatment installations combine the techniques of ferrous metals separation, size reduction and screening, non-ferrous metals separation, and ageing of the treated bottom ash. The product is a dry aggregate with controlled grain size (e.g. 0–4 mm, 0–10 mm, 4–10mm), which may be used as a secondary construction material.

The process consists of the following subsequent steps:

- cooling down the temperature of the bottom ash in air
- ferrous metals separation
- sieving
- crushing of coarse fraction
- sieving
- ferrous separation
- non-ferrous separation
- ageing.
- Several plants exist in Denmark, the Netherlands, Germany, Belgium, and France.

6.6.2 Wet bottom ash treatment installations

In wet bottom ash treatment systems bottom ashes are treated by size reduction, sieving, washing and metals separation. The main feature of the treatment is the wet separation of a 0–2 mm fraction. As the majority of leachable components and organic compounds remain with the fine fraction, this results in a reduced leachability of the remaining product fraction (>2 mm). The simplest way to do washing is to carry it out in the ash quench tank, as is already performed in some German plants. (BREF 2005)

Wet bottom ash treatment aims to remove trace elements, in order to reduce both the trace element content and leaching. Other constituents of concern are soluble salts, mainly alkali and earth-alkali chlorides and sulphates. Approx. 50 % of the chloride content can be reduced by washing the ashes. The sulphate solubility is controlled by the solubility equilibrium of the predominant earth-alkali sulphates. Stabilisation or removal is then difficult (Vehlow, 2002).

In general wet concentration yields better product qualities. For finer particles (<1mm) in bottom ash, wet treatment is practically the only effective treatment method. Disadvantages are often higher costs due to additional maintaining a water circuit, corrosion, and especially fines treatment and its related water cleaning, filtering, drying and sludge disposal. Dry treatment techniques have also poorer performance than the wet concentration methods (Kinnunen 2006).

Figure 6.3 schematically shows the installation for the wet treatment of bottom ash consisting of various mechanical parts with different functions: washing, sieving and separating.

Figure 6.3 Example of wet bottom ash installation in Indaver, Belgium (Vandecasteele et al., 2007).

In a first step a robust bar sieve removes large parts of metal and stone (not shown in figure 6.3). Water is then added and the material >50 mm is removed by a screening and washing installation. Ferrous metals are removed magnetically from this fraction and the rest of the material >50 mm is partly sent back to the grate incinerator. The finer material (<50 mm) passes through a washer barrel to remove the light organic material, which is also sent back to the grate incinerator. Another screening and washing installation separates the particles in three different fractions: 6–50 mm, 2–6 mm and <2 mm. Ferrous separators retrieve the iron from the two larger fractions. Non-ferrous separators, based on eddy current, are foreseen for the two larger fractions to separate mainly aluminium and copper. In practice they are always in use for the 6–50 mm fraction, and are optional (only used to improve the quality of the end-product, when it is applied in building materials) for the 2–6 mm fraction, where the yield of non-ferrous metal is low. Finally the fraction <2 mm is separated in a sludge fraction (<0.1 mm) and a sand fraction (0.1–2 mm). The sludge is landfilled without stabilisation.

6.6.3 Examples of bottom ash treatment installations

Two different bottom ash processing flow sheets are reviewed in this section, one industrial flow sheet and one pilot plant test flow sheet. Both processes are based on the same principle: first sizing into suitable size fractions

and sorting out unburned fractions, magnetic separation to remove ferro-magnetic metal scrap and the eddy current separator for separation of non-ferrous metals.

Schmelzer (1995) described in his pioneering article a process for metal recovery from bottom ash. In the pilot operation (figure 6.4), the bottom ash was first dried and then screened into two size fractions: 0–4mm and 4–45mm. The both size fractions were treated separately with magnetic separator and eddy current separator. The nonmetallic product of 4–45mm was returned to the feed. The pilot plant test result was that from 62.6w-% of processed ash, 35.5 w-% magnetic materials and 1.9 w-% non-ferrous materials was recovered from the bottom ash. The magnetic material contained 20–30w-% Fe. After further processing with impact crushing, screening and wet magnetic separation, fine scrap with 90–95% Fe and iron concentrate with 50–55% Fe were produced. The non-ferrous material mainly comprised of Al, Cu and Zn. By further wet gravity separation, light product containing 95% Al and heavy product with Cu 52.1–61.2%, Zn 25.2–36.5%, Pb 1.2–3.5%, Sn 0.67–1.46% and Ag 0–2.0% were obtained (Schmelzer, 1995).

Figure 6.4 Recovery of metals from incineration residue (Schmelzer, 1995).

Figure 6.5 shows the MSWI processing plant at Rugenberger Damm in Hamburg. The flowsheet is very similar to the one in figure 6.4, but it includes an additional fines removal stage. The fines and the coarse particles are combined and the whole fraction is screened with 10mm screen. The overflow of the screen is fed to a wind shifter and the light weight material is discharged into bunker. The underflow of 10mm screen is combined with the heavy product of windshifter and the combination is stored for three months, which is necessary to assure a better leachate quality. The material in storage is either fed back to the treatment process or sent back to the customer.

In this process by-products of bottom ash processing that are recovered for re-use include metals, mainly ferrous metal which is extracted by magnets, and non-ferrous metals such as aluminum, brass, copper and stainless steel, which are recovered with the assistance of eddy current separators.

The amount of steel recovered is about 2.5% of the waste input. Non-ferrous metals are recovered at a rate of about 0.25% of waste input of which about 40% is stainless steel (ferrous). After incineration the metal is sterile, and once the metal has been cleaned from adhering bottom ash in a separate process the metal can easily be sold to scrap dealers (Zwarhr 2004).

Figure 6.5 Treatment of bottom ash in MSWI plant at Rugenberger Damm in Hamburg, Germany (Zwahr, 2004).

In the Netherlands and German MSWI plants ferrous and non-ferrous metals and coarse materials ($>40\text{mm}$) are removed from bottom ash. However, the current process does not yet produce products that meet the environmental requirements for reuse of the remaining bottom ash fraction because the major non-ferrous metals present in fine fractions ($<10\text{mm}$) still remain. Also, the soluble salts are not removed by this dry process (Xing 2004).

6.7 Environmental evaluation

Untreated bottom ash may contain leachable chloride, fluoride, copper, antimony and DOC exceeding the EU leaching WAC for non-hazardous waste landfills receiving stable, non-reactive hazardous waste.

Run-off water from aging and storage of bottom ash may contain salts and metals and will need treatment. The water can be re-circulated or used in the incinerator as process water. Odour and dust controls may be required. Vehicle and machinery noise may be an issue in some locations.

Anti-explosive devices at indoor ageing facilities may be required (BREF 2005).

In some cases the entire process is performed inside a closed building. This assists with dust, odour, noise (from machinery and vehicles), and leachate control. In other cases, the entire process is totally or partially performed outdoors. This generally allows more space to easily handle bottom ash, and can give more air circulation for bottom ash to mature, may avoid the release of explosive hydrogen in combination with aluminium during the ageing process (BREF 2005).

The main environmental benefit of installing a mechanical treatment process is a reduction of the volume of rejects and wastes, and therefore, a higher global recovery rate. Energy consumption, and potential for noise and dust emissions are the most notable crossmedia effects (BREF 2005).

In thermal treatment very high energy consumption is reported 0.7–2 kWh/kg of ash treated (IAWG, 1997). Power requirements are typically; AC submerged arc furnace 650–1000 kWh/t-ash; DC plasma furnace 600–800 kWh/t-ash. In France, a concern is the outlet of the vitrified residue, due to lack of clear regulation. It is not permitted to use it and it must be removed for landfill. The available (permitted) outlet is based on ash origin and not its properties (BREF 2005).

The flue gas issued from thermal treatment of solid residues can itself emit high level of pollutants such as NO_x, TOC, SO_x, dust, heavy metals etc. Therefore flue-gas treatment is also required to remove pollutants from the ash treatment stage of gases (or to treat the flue-gas in the FGT of the nearby incinerator plant). The process is reported to be complex and the availability may be critical (BREF 2005).

The wet treatment results in the production of a fine fraction (0–2 mm) for disposal or recovery. Additionally a waste water fraction is produced. This waste water may be fed back into the incinerator as process water, if the quality is suitable with the process. An example of the effects of different techniques on the fine fraction of bottom ash (<6 mm) is presented in table 6.7. The results show that the leachability of especially copper is significantly decreased by the ageing process but not further affected by the washing process. However, the additional washing step has an effect on release of salts. The leachability of antimony is not reduced by any of the techniques (Rutten et al., 2006).

Table 6.7 Effects of treatment techniques on the leaching of MSWI bottom ash fraction < 6 mm (Rutten et al., 2006). Test method: EN 12457–3.

Parameter	Unit	Raw bottom ash	Processing	
			Ageing	Ageing + washing
<i>Salts</i>				
Bromide	mg/kg	15	8,1	<0,8
Chloride	mg/kg	2650	2320	550
Fluoride	mg/kg	1,4	4	<1,1
Sulphate	mg/kg	3750	7890	3150
<i>Trace elements</i>				
Antimony	mg/kg	0,36	0,32	0,37
Arsenic	mg/kg	0,02	0,01	0,009
Barium	mg/kg	0,53	0,61	0,79
Cadmium	mg/kg	0,001	0,001	0,0008
Chromium	mg/kg	0,02	0,02	0,02
Cobalt	mg/kg	0,03	0,004	0,004
Copper	mg/kg	5,25	0,99	0,79
Lead	mg/kg	0,05	0,01	0,009
Molybdenum	mg/kg	0,5	0,41	0,25
Nickel	mg/kg	0,07	0,06	0,07
Selenium	mg/kg	0,03	0,02	0,014
Tin	mg/kg	0,07	0,004	0,003
Vadenum	mg/kg	0,07	0,07	0,01
Zinc	mg/kg	0,05	0,11	0,12

6.8 Technical development stage

6.8.1 Unit operations

The separation of metals is a necessary step to allow recycling of the various ash compounds. The ferrous fraction can be recycled, generally after separation of impurities (e.g. dust), as steel scrap for blast furnaces. The non-ferrous metals are processed externally by further separation according to metal type, which may then be re-melted. The resulting ash fraction has a lower metal content and may be processed to yield an inert secondary construction material. Ferrous metals separation uses a small amount of energy. Non-ferrous separation requires size separation, size reduction and spreading of the material. This involves electrical energy use (BREF, 2005).

Magnetic separation of ferrous metals is applicable in all new and existing installations. Nonferrous metals separation requires space and sufficient throughput and may be performed by an external (centralised) bottom ash processing installation. The applicability of the technique is strongly related to the metal content of the waste fed to the furnace. This, in turn, is highly influenced by the collection regime and pretreatment that the waste has undergone before being fed to the furnace. For example, areas with extensive and well adhered to segregation schemes for municipal wastes, may remove significant quantities of metals. Pretreatment of MSW to create

RDF will have a similar effect. At some hazardous waste plants shredded drums are removed using magnets prior to combustion (BREF, 2005).

The mechanical techniques (screening and crushing) are, in principle, applicable to all incineration installations producing an ash requiring treatment before it can be used, or where such treatment may allow increased use (BREF, 2005).

Aging can be applied to all new and existing installations producing bottom ashes. It is mainly used in practice for MSWI to improve its properties prior to utilisation. For some waste streams the ash content may not improve sufficiently from treatment to permit its beneficial use – in such cases the driver for use of the technique may be simply to improve disposal characteristics.

Although used in Japan, thermal systems have had little penetration into other markets owing principally to their high cost, and perceived lack of benefit when existing systems already produce a product of sufficient quality. Plasma treatment is applied to the treatment of combined incinerator and fly ashes. If the chemical residues of FGT are added increased FGT is required (BREF 2005).

6.6.2 Treatment processes

The dry bottom ash treatment process is applicable to new and existing installations. In order to be economically viable, a minimal throughput is needed. For small-scale installations an external (centralised) bottom ash treatment may be used (BREF 2005).

6.9. Operational data

Examples of operational data for three installations are collated below.

Dry treatment system (BREF, 2005):

- Raw slag kept in dry storage for 4– 6 weeks
- Preliminary sieving of particles >150mm
- Removal of ferrous from <150mm fraction
- Further sieve separation (<22mm, 22– 32mm, >32mm)
- <22mm fraction marketed as sand substitute
- >32 mm to hand picking and separators to remove non-incinerable and ferrous fractions, crushing and recirculation
- 22–32 mm fraction air separation of light fractions and ferrous removal
- Separated metal fractions undergo sieving, cleaning and storage before complete re-pass through the process separately from the slag

Wet treatment system (Vrancken, 2001):

- Residue for disposal (0–2 mm): 47 %
- Product for re-use (2–60 mm): 34 %
- Ferrous metals: 12 %
- Non-ferrous metals: 2 %
- Unburned back to incinerator: 5 %

A Swedish dry treatment process (Grönholm, 2008):

- A picture of a Swedish plant dry treatment of bottom ash is shown in figure 6.6.

Figure 6.6 Swedish dry treatment plant for bottom ash.

- Supplier: Heidemann Recycling GmhH
- Construction year: 2003, 2006
- Capacity: 100 000 t/year; max 80 t/h
- Installation: 2 front loaders, 1 mechanic bobcat, 3 magnetic separators, 5 sieves, 4 N/E separators
- Process: wind separator, sorting by fractioning, magnetic separation, non-magnetic separation by Eddy Current, aging
- Sorted fractions:
 - 150 mm: 4 %
 - ferrous metal: 10 %
 - non-ferrous metal: 1 %
 - unburned back to incinerator: 0.25 %
 - Inert material (stainless steel): 5 %
 - residue for disposal: 2 %
 - product for re-use: 80 %

6.10 Economic considerations and efficiency

6.10.1 Unit operations

The main driving force for the physical separation of metals is the interest of scrap market (i.e. market prices) in the produced fractions. Also increased marketability of cleaned bottom ash can be regarded as an important driver. For example, non-ferrous metals may cause damage when re-used in road works, such as swelling. The metal fractions can be sold to scrap dealers. Prices depend on the purity (ferrous) and composition (non-ferrous) of the material. Prices for ferrous scrap are determined by stock exchange. Market price for this material is generally in the range of EUR 0.01–0.05/kg. Non-ferrous scrap needs further processing into pre metal fractions. Prices for non-ferrous scrap depend on the amount impurities (i.e. amount of processing needed) and the composition (i.e. price for final products). The content of aluminium and the market price of secondary aluminium are the main determining factors. The price of the non-ferrous fraction from MSW bottom ash treatment is in the range EUR 0.10–0.60/kg (Vrancken, 2001).

The cost-effectiveness of installing a system for mechanical technologies (screening and crushing) up heavy rejects is to be evaluated on the basis of projected quantities and disposal costs. It is estimated that the payback period for a crusher is on the order of two years for 5 % of rejects to be crushed, for 40000 t/yr of bottom ash, and seven years for 20000 t/yr (BREF, 2005).

The cost of ageing is low as compared to the rest of the treatment installation. Table 6.8 shows an overview of experiences and cost estimates for various unit operations.

Table 6.8 Experience and cost estimates for unit operations.

Unit operation and data Source	Experiences and economic considerations
Carbonisation in storage silo and washing (Steketee et al. 1997)	Leaching decreased: By storage: Mo 64 % and Cu, Zn ja Pb 80–90 %, salts 0 % By washing: Mo 20 % By combined storage and washing: Mo 70–90 % Costs: Storage: EUR 2–5 /t Washing: EUR 3–5 /t Combination: EUR 5–9 /t
Sintering, 900 °C (Selinger et al., 1997)	Only a very small decrease in Pb, Cu, Zn and salt leaching
Melting, 1400 °C (Sakai & Hiraoka 1997)	Used in 24 Japanese plants generally low leaching

Compared to other techniques, external thermal treatment costs are reported to be high (see table 6.9). Vittrification costs are highly sensitive to the unit cost of electricity. Treatment costs are reported to be in the range EUR 100 to 600/t of ash. Investment costs can be up to EUR 20 million for a plant of capacity 1–1.5 t/hr. (BREF 2005).

Table 6.9 Cost estimates for landfilling and thermal treatment processes.

Treatment technique used	Cost (EUR/t of bottom ash)	Cost (EUR/t of MSW)
Direct landfill	35	12
Pretreatment for utilisation	20	7
Fusion (fossil fuel, no pretreatment)	100	30
Fusion (fossil fuel, scrap removal)	130	45
Fusion (electric heating)	120	40
Fusion processes in Japan	100	30
Fusion processes estimates (IAWG, 1997)	180	60

6.10.2 Treatment processes

The main benefit of the dry bottom ash treatment process is to avoid disposal costs; apart from that, the economy of the bottom ash treatment operation depends on the market price of the produced fractions. Treated bottom ash will generally be sold, transport included, at zero cost. Revenue is created by the quality of nonferrous and ferrous metals fractions. Aluminium is the product with the highest market value. The amount and purity of the produced aluminium is an important factor in the overall economics of the installation. (BREF 2005).

Table 6.10 presents some economic information and considerations for existing treatment plants for MSWI bottom ash.

Table 6.10 Economic information on existing treatment plants.

Data Source (BREF 2005)	Economic considerations
A MSWI plant in Sweden using various dry ash treatments and ageing	The tax for land filling the ash is approximately EUR 40/t The stone and gravel which is separated at the earlier stages of the treatment process has a commercial value of around EUR 6/t The commercial value of the treated bottom ash residue as a construction material is around EUR 2.5/t.
A MSWI plant in Portugal using ferrous removal and 1–2 days storage prior to landfill	The cost of depositing the residues in the self owned landfill is approximately EUR 2/t. This site is geographically near the incineration plant and thus transport costs are low. The cost of depositing the residues in the site owned by the external company is EUR 2/t transport and EUR 9/t disposal – total EUR 11/t.
A MSWI in Portugal using ferrous and non-ferrous removal followed by 3 months stabilisation prior to final landfill:	The commercial value of the ferrous metal is around EUR 102.13/t. Landfill costs: EUR 28/t Bottom ash transport to landfill: EUR 3/t

6.10.3 Experiences

Data from a test programme in a full scale German waste incineration plant illustrate the effect which 12 weeks ageing has on the pH of bottom ashes and on the test results obtained by the DEV S4 method. The data showed that the pH of the fresh bottom ashes in the DEV S4 test typically exceeds 12 and drops down by about two units during the ageing process. This pH change had no effect on the leaching properties of Mo, which is present

mainly as molybdate. The leaching stability of Cu and Zn is moderately improved in the aged material whereas the leaching of Pb is reduced by almost two orders of magnitude. (BREF 2005)

The French Bureau of Mines conducted a study during 18 months about the ageing and its effect on leaching of a 400 tonnes stockpile of bottom ashes and concluded similarly to this German study. If longer ageing periods (e.g. >20 weeks) are used for ferrous free bottom ash without turning, the aged bottom ash will become increasingly solidified (BREF, 2005).

Washing can be done on-site by heap leaching during some weeks (L/S 0,5-1). Washing experiments have shown that some salts (Cl, Br, F) can be washed out properly. For washing out sulphate a more intensive washing process is necessary requiring more water (L/S 5-10) and/or additives. The leaching of trace elements like Cu, Mo and Sb is hardly influenced by washing. Because of the foregoing ageing process the metals are already fixated. Washing can be more effective when the process is performed in 2 steps at different pH values (5 for copper and 7 for molybdenum) at an L/S value of 10, without ageing the ash before the washing step. A two-step washing process can decrease the leaching of Cu, Sb and Mo by a factor of 10, 3 and 2 respectively (Rutten et al., 2006). The costs of a two-step washing process, including water treatment facilities are exceptional high (50-100 euro/ton) and therefore this option is not realistic (Rutten et al., 2006).

In a Dutch study the ash is sieved at 150 mm and the minus 150 mm fraction is stored for some weeks in a depot. Thereafter the accelerated ageing is started by aerating the ash with air with increased CO₂ content. The pH values of the ash drops from 11 to 8 or 9 and the leaching of some metals will decrease. By measuring the pH of the ash the process can be monitored. After ageing the ash can be processed in already existing ash treatment installation, consisting of sieving and treating the sieve fractions by separation fluff, iron and nonferrous metals, using sifting, magnet and Eddy-Current techniques. After the treatment the fractions are combined. Accelerated ageing in depots or in a bunker is a robust technique, proofed on a practical scale and can be easily implemented in the current treatment installation. The costs of this route are in the Netherlands estimated at 13-19 euro/ton (7-9 euro/ton treatment costs and 6-10 euro/ton for transport and sale) (Rutten et al., 2006).

In Denmark there has been study on the effect of combining ash washing in lab and pilot scale with removal of small grain sizes below 100-125 µm. A positive effect on leaching could be observed for sulfate, Na, As, Cd, Cr, Pb, Zn, Mo and organic matter. Using additives in washing process mainly affected sulfate and Mo leaching. Also technical properties of the ashes were improved (Astrup 2006). Similar results were obtained from the Amsterdam pilot plant experiment (Xing 2004).

The choice of the optimal treatment method for bottom ash is dependent on availability of disposal sites (also produced waste volumes), national policy and criteria for disposal and reuse in earth constructions. Often the selection of best available technique requires a multi-criteria assessment that

considers a very wide range of drivers (e.g. water consumption and release, energy consumption, etc) in order to arrive at a balanced overall solution. It is considered as BAT practice to recover ferrous and non-ferrous fractions of bottom ash. The dry bottom ash treatment is most widely used and generally results in a bottom ash aggregate that in many countries can be used in earth construction depending on the national criteria for re-use. The leaching of copper often remains as a problem even after bottom ash treatment. The removal of organic matter e.g. by use of an afterburner in the incineration plant seems important to solve this problem. This means that the quality of the bottom ash from a dry treatment process is dependent on the original bottom ash (Nielsen et al, 2006b).

7. Bioashes

7.1 Origin and properties of ash from biomass incineration

Biomass ashes are generated by the incineration of various types of biomass fuels such as wood chips, straw, bark, logging waste, construction wood and peat. The properties of the bioash depend on the type and origin of the biomass and the incineration and ash removal technology. The bioash may occur as bottom ash and fly ash (with or without acid gas cleaning residues), similarly to MSWI residues, or they may occur as a mixture.

Table 7.1 shows composition and leaching properties of fly ashes from production of energy from two types of biomass, straw and wood chips, in Denmark. The two ashes both have high contents of soluble salts, particularly chloride, sulphate and potassium (K). They differ in pH and alkalinity, where the straw fly ash has a very low alkalinity (addition of 0.5 mol equivalents of acid/kg of residue will change the pH in a slurry at L/S = 20 l/kg from the original value of 9.0 to 4.0) whereas the strongly alkaline wood chip ash has a much higher alkalinity (it will take 9.5 mol equivalents of acid/kg of residue to change the pH in a similar slurry from the original value of 12.2 to 4.0). Note that both ashes are more than 50 % soluble!

Table 7.1 Total content and leached amounts at L/S = 2 l/kg and 10 l/kg (EN 12457-3) of a number of substances in two Danish bioashes (Hjelmar et al., 2006a).

Parameter	Unit	Fly ash from incineration of straw			Fly ash from incineration of wood chips		
		Total content	Leaching L/S=2 l/kg	Leaching L/S=10 l/kg	Total content	Leaching L/S=2 l/kg	Leaching L/S=10 l/kg
Chloride	g/kg	200	160	160	110	79	83
Sulphate	g/kg	170	29	160	160	110	220
Na	g/kg		5.9	5.9		43	53
K	g/kg		180	280		99	130
Ca	g/kg	27	0.3	1.4	150	0.16	6.0
Si	g/kg	40			5.9		
Al	mg/kg	2300	< 0.2	< 1	2000	0.4	< 1
As	mg/kg	7.0	0.29	0.69	1.0	0.027	< 0.02
Ba	mg/kg	210	0.30	1.0	730	0.74	2.1
Cd	mg/kg	10	0.56	0.47	8.0	0.0022	< 0.004
Cr	mg/kg	97	0.22	0.32	18	3.4	4.1
Cu	mg/kg	120	0.077	0.10	150	0.020	0.10
Hg	mg/kg	< 0.2			< 0.2		
Mn	mg/kg	550	0.24	< 0.3	11000	< 0.02	< 0.1
Ni	mg/kg	49	0.03	< 0.1	92	< 0.02	< 0.1
Pb	mg/kg	24	0.026	0.14	42	0.070	0.20
Zn	mg/kg	620	0.04	0.2	500	0.03	0.10
TOC/DOC	g/kg	6.5	0.036	0.079	47	0.16	0.22
TDS	g/kg		370	570		330	520
pH	-		8.8	10.1		12.8	12.3

Both fly ashes shown in table 7.1 will exceed the leaching EU WAC for chloride and sulphate, also if the factor 3 rule is applied.

Appendix 5 shows the composition of a number of Swedish bottom ashes and fly ashes from incineration of biomass and the leaching properties of a few of these. It is a little surprising that the contents of soluble salts in the Swedish fly ashes are very low, e.g. compared to the two Danish fly ashes shown in table 7.1.

7.2 Treatment options

In many respects bioashes such as those shown in table 7.1 are similar to APC residues from MSW incinerators. For bioashes resembling those shown in table 7.1, some of the treatment processes described for MSWI APC residues, particularly those processes based on initial extraction of soluble salts, will also be applicable.

It can be seen that particularly straw fly ash has a very high content of potassium (K) which is a valuable fertiliser component. The annual production of straw fly ash in Denmark is approximately 9,000 tonnes (Vestervang, 2005). Kommunekemi A/S in Denmark has developed a process for production of a 22 % solution of potassium chloride (KCl), which is sold to a fertiliser company. The process is based on an aqueous extraction of KCl from the straw fly ash followed by pH adjustment and selective precipitation of heavy metals/trace elements, particularly Cd, which is present and leached from the raw fly ash in substantial amounts (see table 7.1). In addition to the KCl solution product, the process generates a filtercake containing the precipitated heavy metals. No leaching data are publicly available on the filtercake, which is landfilled at a landfill for hazardous waste. In 2006, Kommunekemi A/S treated approximately 2000 tonnes of straw fly ash with this process. There are plans to develop the process further to be able to produce dry potassium sulphate (K_2SO_4) which is a more valuable fertiliser product than potassium chloride (Vestervang, 2005).

8. Conclusions

The implementation of EU Council Decision 2003/33/EC establishing criteria and procedures for the acceptance of waste at landfills pursuant to the Landfill Directive 1999/31/EC in the Nordic Countries means that certain types of waste will require specific treatment prior to landfilling in order to meet the new waste acceptance criteria (WAC). Based on available information, a study to evaluate the technical, economic and environmental applicability of various methods for treatment and improvement of three selected waste streams, MSWI APC residues, shredder residues and MSWI bottom ash, has been carried out. The main conclusions of the study are presented below.

MSWI APC residues

MSWI APC residues, including fly ash, are classified as hazardous waste and must be landfilled at landfills for hazardous waste or landfills for non-hazardous waste receiving stable, non-reactive hazardous waste or they must be placed in underground storage. The main problems in relation to the WAC for hazardous waste landfills and non-hazardous waste landfills receiving stable, non-reactive hazardous waste are the high contents of soluble salts and leaching of Pb and Zn from strongly alkaline residues, and sometimes also leaching of Sb. The most promising treatment methods in terms of technology development, efficiency, maturity of technology, economy and documentation of environmental properties of the treated waste (compliance with the WAC) appear to be extraction of soluble salts combined with chemical stabilisation and possibly recycling of the stabilised remnant to the incinerator grate in order to destroy persistent organic pollutants (e.g. PCDDs/PCDFs). Thermal treatment of APC residues at high temperatures (melting, glassification) may also be effective, but is very energy-consuming and requires re-capture of new APC residues. Stabilisation/solidification using hydraulic binders requires relatively large amounts of binders (cement and/or other pozzolans) and may eventually result in mechanical instability if soluble salts are not removed prior to stabilisation. The prevailing current disposal solution for APC residues in the Nordic countries is landfilling at Langøya in Norway, where the residues, which are mostly alkaline, are mixed with and neutralised with acidic waste prior to disposal in the landfill. This solution is economically very competitive, but the environmental impact, particularly in the longer term, is not very well documented.

The content of salts (mainly calcium chloride) in MSWI APC residues may be recovered and reused, but apart from that, most attempts at finding

realistic and environmentally safe reuse applications for these residues have shown little or no success.

Shredder residues

Shredder residues (SR) are classified as non-hazardous waste in most of the Nordic countries. In Denmark, SR are currently classified as hazardous waste. This means that in case of landfilling, the available options are disposal at landfills for hazardous waste or landfills for non-hazardous waste receiving stable, non-reactive hazardous waste. The Danish classification of SR as hazardous waste is currently under revision. In Finland, some of the SR is classified as hazardous waste and some is classified as non-hazardous waste. The main problems in relation to the WAC for landfilling are the high content of TOC and the high amount of DOC in the eluates from the leaching tests. Both often exceed the WAC for hazardous waste landfills. Other potential problems are the leaching of some metals, in particular Pb, from the SR. The available treatment methods for SR include mechanical separation, thermal treatment (incineration, co-incineration, gasification and pyrolysis) and combinations of mechanical separation and thermal treatment. Most of the treatment methods have been developed for the purpose of increasing the recovery of materials and energy from the shredder residues, not specifically to improve the environmental properties of the treated SR.

The optimal solution in terms of resource recovery, environmental impact and economy appear to be mechanical separation with extensive materials recovery or improved materials recovery integrated into the shredding operation followed by landfilling (if the residual waste stream is not suited for combustion) or co-incineration with MSW (if the residual waste stream is suited for combustion). Landfilling should be developed in the direction of increased sustainability and a reduced aftercare period, and the proportion of SR co-incinerated with MSW (with effective energy recovery) should probably not exceed 5 to 10 % (by mass) of the total feedstock to the incinerator. The bottom ash from the incinerator should be utilised if possible, and the air pollution control residues may require treatment prior to landfilling. If mechanically treated SR is classified as hazardous waste and destined for landfilling or if the EU WAC for landfilling otherwise apply, care should be taken to ensure that TOC (and DOC) is sufficiently low to meet the criteria.

MSWI bottom ash

MSWI bottom ash (BA) is classified as non-hazardous waste in all the Nordic countries, although it may in certain cases also be classified as a hazardous waste in Finland. The BA generally meets the WAC for non-hazardous waste landfills, and potential problems with the environmental quality of BA are more related to requirements for reuse than to the fulfilment of WAC for landfilling. However, the attainment of more sustainable landfill-

ing solutions for MSWI bottom ash could very well require effective treatment prior to landfilling.

The main problems in relation to reuse of MSWI BA and possibly also in relation to sustainable landfilling are chloride, fluoride, Cu, Sb and DOC and sometimes Pb and Cr. Recovery of ferrous and non-ferrous metals as well as crushing and sifting and storage/ageing/carbonisation are parts of the state-of-the-art treatment of MSWI bottom ash. Chlorides may be removed by washing in the quench tank. The choice of the optimal treatment method for bottom ash depends on availability of landfill sites, national policy and criteria for landfilling and reuse in construction works. Often the selection of best available technique requires a multi-criteria assessment that considers a very wide range of drivers (e.g. water consumption and release, energy consumption, etc) in order to arrive at a balanced overall solution. The dry bottom ash treatment is most widely used and generally results in a bottom ash aggregate that can be used for engineering purposes, depending on the national criteria for reuse. The leaching of copper, which is closely associated with the content of DOC in the leachate, often remains a problem even after bottom ash treatment. The removal of organic matter e.g. by use of an afterburner in the incinerator seems important to solve this problem. This means that the quality of the bottom ash from a dry treatment process is dependent on the quality of the original bottom ash. Thermal treatment of the MSWI BA (melting/glassification) is generally energy-consuming and expensive.

Ash from incineration of biofuels

Bioashes, e.g. from incineration of wood chips or straw, may have a large content of soluble salts, which may be utilised, e.g. in the production of fertilisers (KCl, K_2SO_4). This is done by aqueous extraction, and the remnant, which in the case of straw ash has a relatively high content of Cd, has to be stabilised prior to landfilling. Fly ashes from incineration of biofuels are in many ways similar to MSWI APC residues and may be treated similarly.

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Dansk sammenfatning

Det lovmæssige grundlag for affaldsdeponering i de fleste nordiske lande er baseret på EU Direktivet 1999/31/EF af 26. april 1999 om deponering af affald (deponeringsdirektivet) og Rådsbeslutning 2003/33/EF af 19. december 2002 om opstilling af kriterier og procedurer for modtagelse af affald på deponeringsanlæg i henhold til artikel 16 og bilag II i direktiv 1999/31/EF. Implementeringen af Rådsbeslutning 2003/33/EF i de nordiske lande betyder, at visse typer affald vil skulle underkastes en speciel forbehandling for at kunne overholde de nye modtagekriterier for deponering.

Med henblik på at tilvejebringe en vejledning i valg af passende behandlingsteknologier til dette formål, præsenteres i denne rapport en sammenstilling og evaluering af eksisterende og tilgængelige informationer om behandlingsmetoder for udvalgte typer "problemaffald", som ikke kan overholde EU-kriterierne eller de nationale kriterier for modtagelse af affald til deponering, herunder specielt sådanne affaldstyper, som ikke kan overholde kriterierne selv om den såkaldte faktor 3-regel tages i brug. Denne regel tillader en forøgelse af visse grænseværdier med op til en faktor 3. For nogle affaldstyper vil de samme behandlingsmetoder kunne anvendes til at forbedre egenskaberne i forhold til nyttiggørelse.

Tre affaldsstrømme er udvalgt til dette projekt: Røggasrensningsprodukter, inklusive flyveaske, fra affaldsforbrænding, shredderaffald og slagter fra affaldsforbrænding. Af disse udgør kun de to første affaldstyper et egentligt problem i forhold til overholdelse af kriterierne for deponering. De miljømæssige kvalitetsproblemer for forbrændingsslaggen er i højere grad relateret til kriterier for nyttiggørelse end til modtagekriterier ved deponering, om end kvalitetsforbedringer vil være påkrævede for forbrændingsslaggen, hvis man vil opnå en større bæredygtighed i deponeringen. En kort diskussion af egenskaber og behandlingsmuligheder for aske fra forbrænding af biobrændsel indgår også i rapporten.

Rapporten beskriver kriterierne for modtagelse af affald til deponering i de nordiske lande. For de fleste lande stammer disse fra implementeringen af EU's Rådsbeslutning, dog har Færøerne sine egne regler, da de ikke er medlem af EU. Derefter diskuteres de egenskaber ved affaldet, som kan tænkes at skulle forbedres gennem behandling, og der gives en general oversigt over principperne i en række enhedsoperationer og behandlingsformer, som eventuelt vil kunne komme på tale. Endelig beskrives egenskaberne, de potentielle problemer i forhold til overholdelse af kriterierne for modtagelse på deponeringsanlæg, og mulighederne for behandling for hver af de tre affaldsstrømme. Behandlingsmetoderne er beskrevet og evalueret i forhold til teknisk funktion og udviklingsstatus,

begrænsninger, økonomi og effektivitet i forhold overholdelse af de relevante deponeringskriterier efter behandling.

På dette grundlag er der udtaget følgende konklusioner:

Røggasrensningsprodukter fra affaldsforbrænding

Røggasrensningsprodukter fra affaldsforbrænding, inklusive flyveaske, er klassificeret som farligt affald og skal derfor i tilfælde af deponering placeres på deponeringsanlæg for farligt affald eller på deponeringsanlæg for ikke-farligt affald, som modtager stabilt, ikke-reaktivt farligt affald (eller de kan deponeres underjordisk, hvis muligheden er til stede). Hovedproblemerne i forhold til modtagekriterierne for deponering på anlæg for farligt affald eller anlæg for ikke-farligt affald, som modtager stabilt, ikke-reaktivt farligt affald, er de høje indhold af opløselige salte og udvaskning af Pb og Zn fra stærkt alkaliske restprodukter og nogle gange også udvaskning af Sb. De mest lovende behandlingsmetoder for så vidt angår teknologiuudvikling, effektivitet, økonomi og dokumentation af de miljømæssige egenskaber af de behandlede affaldsprodukter med hensyn til overholdelse af deponeringskriterierne synes at være ekstraktion af de opløselige salte efterfulgt af kemisk stabilisering af remanensen, eventuelt kombineret med tilbageførelse af produktet til risten på et forbrændingsanlæg med henblik på termisk destruktion af POP'er (for eksempel PCDD'er og PCDF'er). Termisk behandling af røggasrensningsprodukter ved høje temperaturer (smeltning, glassificering) kan også være meget effektiv, men er samtidig meget energikrævende og forudsætter genopsamling af nye røggasrensningsprodukter. Stabilisering/solidificering under anvendelse af hydrauliske bindemidler kan forventes at kræve anvendelse af forholdsvis store mængder bindemiddel (cement og/eller andre puzzolaner) og kan eventuelt resultere i mekanisk ustabilitet, hvis de opløselige salte ikke ekstraheres før stabiliseringen. Den dominerende bortskaffelsesmetode for røggasrensningsprodukter i de nordiske lande er pt. deponering på Langøya i Oslofjorden, hvor restprodukterne, der for det meste er stærkt alkaliske, blandes med og neutraliseres af syreholdigt affald i forbindelse med deponeringen. Denne løsning er økonomisk stærkt konkurrencedygtig, men de miljømæssige konsekvenser af deponeringen er ikke særlig godt dokumenterede i det materiale, som er tilgængeligt.

Indholdet af salte (primært kalciumklorid) i røggasrensningsprodukter fra affaldsforbrænding kan indvindes og nyttiggøres, men bortset herfra, har forsøg på at finde realistiske og miljømæssigt forsvarlige nyttiggørelsesmuligheder for disse restprodukter ikke haft nogen større succes.

Shredderaffald

Shredderaffald er klassificeret som ikke-farligt affald i de fleste af de nordiske lande. I Danmark er shredderaffald pt. klassificeret som farligt affald og skal derfor i tilfælde af deponering tilføres deponeringsanlæg for farligt

affald eller deponeringsanlæg for ikke-farligt affald, som modtager stabilt, ikke-reaktivt farligt affald. Den danske klassificeringen af SR som farligt affald er pt. under revision. I Finland klassificeres noget shredderaffald som farligt affald og andet som ikke-farligt affald. Hovedproblemerne i forhold til overholdelse af kriterierne for modtagelse til deponering er det høje indhold af TOC og de høje værdier af DOC i eluaterne fra udvaskningstests. Begge overskrider oftest kriterierne for modtagelse af affald på deponeringsanlæg for farligt affald. Andre potentielle problemer er udvaskning af visse metaller, specielt Pb, fra shredderaffaldet. De tilgængelige behandlingsmetoder for shredderaffald omfatter mekanisk separation, termisk behandling (forbrænding, medforbrænding, gasificering og pyrolyse) og kombinationer af mekanisk separation og termisk behandling. De fleste behandlingsmetoder for shredderaffald er blevet udviklet med henblik på øget genvinding af materialer og energi og ikke specielt med fokus på forbedring af affaldets miljømæssige egenskaber.

Den optimale løsning med henblik på genvinding af ressourcer, minimering af miljøbelastningen og rimelig økonomi synes at være mekanisk separation med ekstensiv genvinding af materialer eller forbedret materialeindvinding integreret i shredderoperationen efterfulgt af deponering (hvis den residuale affaldsstrøm ikke er egnet til forbrænding eller forbrændingskapacitet ikke er tilgængelig) eller samforbrænding med dagrenovation (hvis en residuale affaldsstrøm er egnet til forbrænding). Deponering af shredderaffald bør udvikles i retning af mere bæredygtighed og en reduceret efterbehandlingsperiode, og andelen af shredderaffald, der forbrændes sammen med dagrenovation (med effektiv energiudnyttelse) bør formentlig ikke overstige 5 til 10 % af den samlede masse af affald til forbrændingsanlægget. Slaggen fra forbrændingsanlægget bør så vidt muligt nyttiggøres, og røggasrensningsprodukterne vil sandsynligvis skulle behandles forud for deponering. Hvis mekanisk behandlet shredderaffald klassificeres som farligt affald og skal deponeres, eller hvis EU's kriterier for deponering i øvrigt er gældende, skal det sikres, at TOC og DOC er tilstrækkeligt lave, så de overholder kriterierne.

Slagger fra affaldsforbrænding

Slagger fra affaldsforbrænding er klassificeret som ikke-farligt affald i alle de nordiske lande, selv om det i nogle tilfælde kan klassificeres som farligt affald i Finland. Slaggen overholder som regel kriterierne for modtagelse på deponeringsenheder for ikke-farligt affald, og potentielle problemer med de miljømæssige egenskaber er i højere grad relateret til kriterier for nyttiggørelse end til kriterier for deponering. Det må dog antages, at opnåelse af en højere grad af bæredygtighed i deponeringen også vil forudsætte en effektiv behandling forud for deponering.

Hovedproblemet i forhold til nyttiggørelse af forbrændingsslagger og sandsynligvis også bæredygtig deponering er udvaskning af klorid, fluorid, Cu, Sb og DOC og sommetider også sulfat, Pb og Cr. Genvinding af

ferromagnetiske og ikke ferromagnetiske metaller samt knusning, sigtning og ældning/oplagring/karbonatisering indgår i state-of-the-art håndtering af forbrændingsslagger. Klorider (men ikke sulfater) kan udvaskes i quencheren. Valget af den optimale behandlingsmetode for forbrændingsslagger afhænger af tilgængeligheden af deponeringskapacitet, national og lokal affaldspolitik og kriterierne for deponering og anvendelse til bygge- og anlægsformål. Valget af den bedste tilgængelige teknologi (BAT) kræver ofte multikriterie-vurderinger, som tager en lang række forhold (f.eks. vand- og energiforbrug) i betragtning for at nå frem til en afbalanceret løsning. Den tørre slaggebehandlingsmetode er den mest udbredte og resulterer generelt i et slaggeagregat, som kan anvendes til entreprenørmæssige formål, afhængigt af de nationale betingelser for nyttiggørelse. Udvaskningen af Cu, som er stærkt korreleret med indholdet af DOC i perkolatet, forbliver ofte et problem også efter behandling af slaggen. Fjernelse af organisk materiale fra slaggen, f.eks. ved hjælp af efterbrændere ved risten på forbrændingsanlægget kan muligvis bidrage til at løse dette problem. Det betyder, at kvaliteten af slagge, som er behandlet ved en tør proces afhænger af slaggens oprindelige kvalitet. Termisk behandling af forbrændingsslagge (smeltning/glassificering) er generelt meget energi- og omkostningskrævende.

Bioasker

Bioasker, f.eks. fra forbrænding af træflis og halmstrå, kan have høje indhold af opløselige salte, som kan oparbejdes og nyttiggøres, f.eks. til produktion af gødningsprodukter (KCl, K₂SO₄). Dette gøres gennem en vandig ekstraktion, og remanensen, som hvis der er tale om halmstrå, ofte har et højt indhold af Cd, skal stabiliseres forud for deponering. Flyveasker fra forbrænding af biobrændsler har mange lighedspunkter med røggasrensingsprodukter fra affaldsforbrænding og vil kunne behandles nogenlunde på samme måde.

Appendix 1

Data on MSWI APC residues

Table A1.1 Total contents of major and minor elements in fly ash and acid gas cleaning residues from several MSW incinerators in Europe and North America. n = number of different residue samples included (after IAWG, 1997).

Residue	Element	Average (mg/kg)	25%-75% fractile (mg/kg)	n
Fly ash	Ca	107.000	95.000-120.000	20
	Cl	74.000	40.000-102.000	24
	Si	160.000	130.000-180.000	14
	Mg	15.000	14.000-17.000	15
	Fe	25.000	18.000-33.000	20
	Al	71.000	59.000-81.000	18
	K	36.000	30.000-41.000	19
	Na	31.000	23.000-38.000	17
	Zn	28.000	16.000-35.000	26
	S	26.000	21.000-33.000	20
	Pb	11.000	6300-15.000	25
	Ti	8700	7500-9400	17
	Mn	1300	1000-1600	19
	Ba	1700	940-2600	18
	Sn	1400	860-1800	15
	Cu	1200	930-1300	25
Fly ash containing APC residues from the dry and semidry processes	Ca	230.000	180.000-280.000	19
	Cl	180.000	91.000-222.000	23
	Si	69.000	51.000-92.000	12
	Mg	9400	7400-12.000	16
	Fe	12.000	63.000-11.000	19
	Al	26.000	15.000-29.000	27
	K	23.000	15.000-31.000	18
	Na	17.000	12.000-20.000	16
	Zn	15.000	12.000-18.000	28
	S	15.000	8200-21.000	18
	Pb	5400	4100-6300	27
	Ti	3300	2600-4400	17
	Mn	480	280-680	19
	Ba	540	320-660	18
	Sn	890	770-1000	15
	Cu	710	490-860	25
Sludge from wet scrubber process (without fly ash)	Ca	150.000	87.000-200.000	3
	Cl	36.000	26.000-47.000	4
	Si	78.000	-	1
	Mg	75.000	19.000-170.000	3
	Fe	54.000	20.000-97.000	3
	Al	28.000	21.000-39.000	3
	K	3900	810-8600	3
	Na	1900	720-3400	3
	Zn	31.000	15.000-45.000	12
	S	4400	2700 6000	2
	Pb	11.000	4400-19.000	12
	Ti	2600	1400-4300	3
	Mn	9100	5400-12.000	3
	Ba	460	87-670	11
	Sn	400	340-450	2
	Cu	1200	760-1700	12

Table A1.2 Total contents of trace elements in fly ash and acid gas cleaning residues from several MSW incinerators in Europe and North America. n = number of different residue samples included (after IAWG, 1997).

Residue	Element	Average (mg/kg)	25%-75% fractile (mg/kg)	n
Fly ash	Hg	8,0	2,3-10	17
	Cd	390	240-480	26
	Sb	530	340-690	12
	Cr	650	430-840	26
	Sr	280	140-400	12
	Ni	140	91-110	25
	As	130	49-200	17
	V	51	32-63	15
	Ag	55	33-75	10
	Co	51	30-69	17
	Mo	40	25-37	13
	Se	14	11-18	12
Fly ash containing APC residues from the dry and semi-dry processes	Hg	15	8,4-18	28
	Cd	300	190-360	28
	Sb	790	630-940	13
	Cr	180	110-220	28
	Sr	460	400-500	6
	Ni	94	23-60	25
	As	170	120-210	26
	V	33	19-50	6
	Ag	22	1,1-49	11
	Co	9,6	6,0-15	11
	Mo	15	11-20	11
	Se	8,2	4,8-11	23
Sludge from wet scrubber process (without fly ash)	Hg	650	240-790	12
	Cd	630	290-880	12
	Sb	140	80-200	2
	Cr	240	130-340	12
	Sr	104	52-130	10
	Ni	62	26-53	12
	As	89	49-110	12
	V	47	25-86	3
	Co	9,8	4,8-16	12
	Mo	12	3,0-24	11

Table A1.3 Examples of contents of specific groups of organics in fly ash and acid gas cleaning residues from MSW incinerators in $\mu\text{g kg}^{-1}$.

Parameter	FA	DP/SP	WP	WP+FA
TCDD/TCDF (Eadon)	1,2-2,5 ^a	0,8-2,0 ^f	-	2,8 ^f
PCDD/PCDF (I-TEQ)	0,2-10 ^b			
PCB	6-16 ^c <20 ^d 10-250 ^b <20 ^d	-	<20 ^d	-
PCBz	890 ^e 800 -1900 ^c 100-4000 ^b 88-280 ^d	0,03 -0,4 ^g	<4-7,4 ^d	-
PCPh	1800 ^e 1500 -3200 ^c 50-10.000 ^b 110-210 ^d	50-200 ^g	42-50 ^d	-
PAH	110 ^e 450-1100 ^c 50-2000 ^b 28-82 ^d	-	20-35 ^d	-
Phtalater	<500-4300 ^d	-	1600-10.000 ^d	-

FA: Fly ash from ESP, DP/SP: Dryt/semidry product with fly ash, WP: Sludge from wet scrubber process. WP+FA: Sludge from wet scrubber process mixed with fly ash.

a: Hjelmar et al. (1994), b: Vehlou (1995), c: Sawell og Constable (1988) refereret i IAWG (1997), d: Kjølholt et al. (1994), e: Klicius et al. (1988) - refereret i IAWG (1997), f: Hjelmar (1992), g: Sawell et al. (1990b) - refereret i IAWG (1997).

Table A1.4 The effect of washing of semidry APC residue and MSWI fly ash in a large scale pilot plant (Hjelmar et al., 2006a). The results are the amounts leached in the batch leaching test EN 12457-2 (L/S = 10 l/kg). For comparison, some European criteria for acceptance of waste at landfills are also shown. Test results that do not meet the criteria for acceptance at a hazardous waste landfill are shown in bold.

Parameter	Unit	Semidry APC residue		MSWI fly ash		EU landfill WAC ¹	
		Untreated	Washed	Untreated	Washed	Non-hazardous ²	Hazardous ³
Chloride	mg/kg	110000	2200	92000	1700	15000	25000
Sulphate	mg/kg	20000	14000	28000	14000	20000	50000
Fluoride	mg/kg	54	45	52	34	150	500
Ca	mg/kg	45000	19000	24000	12000	-	-
Na	mg/kg	24000	520	32000	700	-	-
K	mg/kg	25000	570	41000	860	-	-
As	mg/kg	0.021	0.0093	0.096	0.069	2	25
Ba	mg/kg	0.72	0.044	0.036	1.1	100	300
Cd	mg/kg	0.0023	< 0005	0.18	0.014	1	5
Cr	mg/kg	2.9	0.77	8.1	12	10	70
Cu	mg/kg	2.3	0.92	0.31	0.078	50	100
Hg	mg/kg	0.093	0.0025	0.011	0.0081	0.2	2
Mo	mg/kg	2.2	1.8	6.1	2.9	10	30
Ni	mg/kg	0.15	0.047	0.061	0.025	10	40
Pb	mg/kg	560	93	420	120	10	50
Sb	mg/kg	0.0086	0.0038	0.023	0.010	0.7	5
Se	mg/kg	0.20	0.20	0.15	0.34	0.5	7
Zn	mg/kg	46	37	34	13	50	200
DOC	mg/kg	62	15	10	10	800	1000
pH	-	12.2	12.5	12.3	12.2	≥ 6	-

1): Criteria for acceptance of waste at landfills in accordance with the EU Landfill Directive (CEC, 1999) and the associated Council Decision) on waste acceptance procedures and criteria (CEC, 2003).

2): Leaching criteria for acceptance of waste at landfills for non-hazardous waste receiving stable, non-reactive hazardous waste (L/S = 10 l/kg).

3): Leaching criteria for acceptance of waste at landfills for hazardous waste (L/S = 10 l/kg).

Figure A.1-A detailed view of the upper parts of the plots in figure 4.2a in section 4.1.3 of column test results on semidry APC residues and fly ash from MSW incinerators for chloride and sulphate.

Figure A.2-a pH dependence of the leaching (CEN/TS 14997 at L/S = 10 l/kg) of semidry APC residue and MSWI fly ash before and after washing with water (Hjelmar, 2007).

Figure A.2-b pH dependence of the leaching (CEN/TS 14997 at $L/S = 10$ l/kg) of semidry APC residue and MSWI fly ash before and after washing with water (Hjelmar, 2007).

Figure A.2-c pH dependence of the leaching (CEN/TS 14997 at $L/S = 10$ l/kg) of semidry APC residue and MSWI fly ash before and after washing with water (Hjelmar, 2007).

Figure A.2-d pH dependence of the leaching (CEN/TS 14997 at $L/S = 10$ l/kg) of semidry APC residue and MSWI fly ash before and after washing with water (Hjelmar, 2007).

Appendix 2

Data on the VKI process

Figure A2.1 Layout of one-stage VKI-process for continues operation

Figure A2.2 Layout of two-stage VKI-process for continuous operation.

Figure A2.3a: Results of pH dependence test (CEN/TS 14997) performed on MSWI fly ash – untreated and treated by the VKI process (Hjelmar et al., 2006a).

Figure A2.3b: Results of pH dependence test (CEN/TS 14997) performed on semidry MSWI APC residue – untreated and treated by the VKI process (Hjelmar et al., 2006a).

Figure A2.4a: Results of column leaching test (CEN/TS 14405) performed on MSWI fly ash – untreated and treated by the VKI process (Hjelmar et al., 2006a).

Figure A2.4b: Results of column leaching test (CEN/TS 14405) performed on semidry MSWI APC residue – untreated and treated by the VKI process (Hjelmar et al., 2006a).

Appendix 3

Data on shredder waste

Figure A3.1a pH dependent leaching of shredder waste at $L/S = 10$ l/kg, determined with CEN/TS 14997 (Hjelmar, 2009).

Figure A3.1b pH dependent leaching of shredder waste at $L/S = 10$ l/kg, determined with CEN/TS 14997 (Hjelmar, 2009).

Figure A3.1c pH dependent leaching of shredder waste at $L/S = 10$ l/kg, determined with CEN/TS 14997 (Hjelmar, 2009).

Figure A3.2 Acid/base neutralisation capacity of shredder residue SR1 (Hjelmar, 2009).

Appendix 4

Data on MSWI bottom ash

Table A4.1 Composition of MSWI bottom ash and the lithosphere based on total analysis (Hjelmar et al. 2004/2009).

Element	Unit	Bottom ash from 7 Danish MSW incinerators 1991 ^a		Bottom ash from 3 Danish incinerators 2002 ^b	MSWI bottom ash from several countries ^c	Lithosphere ^d
		Range	Average	Average	Range	Average
O	g/kg	-	-	-	400 - 500	465
Si	g/kg	240 - 300	270	210	91 - 310	276
Ca	g/kg	60 - 79	71	95	60 - 120	36
Fe	g/kg	54 - 88	68	96	4.1 - 150	51
Al	g/kg	42 - 61	50	48	22 - 73	81
Na	g/kg	18 - 30	24	21	2.9 - 42	28
K	g/kg	11 - 15	12	11	0.8 - 16	26
C	g/kg	-	-	-	10 - 60	0.95
Mg	mg/kg	7700 - 9300	8500	11000	400 - 26000	21000
S	mg/kg	4000 - 7300	5700	3600	1000 - 5000	600
Ti	mg/kg	4100 - 5500	4600	6400	2600 - 9500	6000
P	mg/kg	1800 - 5100	3000	5000	1400 - 6c400	1200
Cu	mg/kg	1800 - 3600	2800	2700	190 - 8200	70
Zn	mg/kg	1900 - 3000	2300	3300	610 - 7800	80
Cl	mg/kg	1400 - 4000	2100	1700	800 - 4200	500
Pb	mg/kg	870 - 2200	1600	990	98 - 14000	16
Ba	mg/kg	1200 - 1800	1400	1500	400 - 3000	430
Mn	mg/kg	840 - 1500	1100	1600	83 - 2400	900
Ag	mg/kg	-	-	-	0.3 - 37	0.007
As	mg/kg	8.5 - 15	12	47	0.12 - 190	5
B	mg/kg	-	-	-	38 - 510	10
Br	mg/kg	-	-	-	1.4 - 150	2.5
Cd	mg/kg	0.75 - 2.2	1.4	3.7	0.3 - 71	0.2
Co	mg/kg	-	-	27	6 - 350	40
Cr	mg/kg	230 - 620	330	600	23 - 3200	200
Cs	mg/kg	-	-	-	1.0 - 2.0	3.2
F	mg/kg	-	-	-	200 - 1100	625
Hg	mg/kg	< 0.08	< 0.08	< 0.1	0.02 - 7.8	0.1
I	mg/kg	-	-	-	2 - 10	0.3
Mo	mg/kg	19 - 98	47	17	2.5 - 280	2.3
N	mg/kg	250 - 840	480	-	110 - 900	-
Ni	mg/kg	130 - 400	230	210	7 - 4300	100
Sb	mg/kg	-	-	-	10 - 430	-
Se	mg/kg	-	-	< 1	0.05 - 10	0.09
Sn	mg/kg	150 - 260	210	160	2 - 380	40
Sr	mg/kg	200 - 270	230	300	85 - 1000	150
V	mg/kg	43 - 86	67	67	20 - 120	150

a): Hjelmar *et al.* (1996)

b): Hjelmar and Holm (2003)

c): Data from Canada, Denmark, Germany, Netherlands, Sweden and USA (IAWG, 1997)

d): Lindsay (1979)

Table A4.2 Contents of various specific groups of organics in MSWI bottom ash in µg kg⁻¹ (Hjelmar et al., 2004).

Parameter	Vehlow (1995)	Kjølholt et al. (1994)
PCDD/PCDF ^a	<0,001-0,02	0,005-0,018 ^b
PCB ^c	<5-50	<20-110
PCBz ^d	<2-20	<3-22
PCPh ^e	<2-50	1,4-22
PAH ^f	<5-10	440-1700
Phtalater	-	930-5500

a: polychlorinated dibenzo-p-dioxins og polychlorinated dibenzofurans) measured as international toxic equivalents (I-TEQ).

b: normal level for Danish MSWI bottom ash (Hjelmar, 1998)

c: polychlorinated biphenyls.

d: polychlorinated benzenes.

e: polychlorinated phenols.

f: polycyclic aromatic hydrocarbons.

Table A4.3 - Results obtained by leaching of bottom ash in a column test (Hjelmar et al., 2006a).

Parameter	MSWI bottom ash leached in column tests CEN/TS 14405 (6 samples from 3 incinerators)			
	Total content	Amount leached at L/S = 0-10 l/kg	Composition of first eluate (L/S = 0-0.1 l/kg)	Composition of last eluate fraction (L/S = 5-10 l/kg)
	mg/kg	mg/kg	mg/l	mg/l
Chloride	-	690-1800	1500-8000	2.3-3.3
Sulphate	9600-13000	1700-2900	1800-3300	54-79
Ca	80000- 119000	610-950	260-640	43-57
Na	23000-28000	880-2000	1600-6100	4.6-15
K	9100-11400	300-800	250-1800	10-21
As	1300-1600	0.013-0.017	0.0025-0.020	< 0.001-0.0016
Ba	37-52	0.13-0.45	0.030-0.095	0.012-0.040
Cd	2.5-4.5	< 0.001	< 0.0001	< 0.0001
Cr	470-730	0.009-0.047	0.00035-0.0018	0.0011-0.0038
Cu	2300-4300	0.2-1.7	0.15-1.7	0.014-0.057
Hg	< 0.1	< 0.0003	< 0.004-0.19	< 0.00002
Mo	9.6-23	-	-	-
Ni	110-350	< 0.006-0.018	0.0062-0.023	< 0.0005-0.00086
Pb	860-1500	< 0.003-0.0053	0.0002-0.012	< 0.0002-0.00064
Sb		0.20-0.36	0.011-0.042	0.018-0.035
Se	< 0.5-1.0	0.0084-0.026	0.009-0.059	0.00033-0.00056
Zn	2700-3400	< 0.02-0.069	0.001-0.034	< 0.001-0.0084
TOC/DOC	7000-8000	59-110	88-330 pH = 9.8-10.8	3.6-6.4 pH = 10.3-11.6

Appendix 5

Data on compostion and leaching of bioashes

Table A5.1 Composition of Swedish bottom ash from incineration of wood (source: the Swedish database Allaska).

Data	Unit	Num-ber	Median	Avera-ge	Min	Max	StDev
Al ₂ O ₃	% DS	76	5.29	5.87	0.36	13.6	3.94
As	mg/kg DS	77	8.3	22.7	0	173	32.3
B	mg/kg DS	62	213	194	0	505	152
Ba	mg/kg DS	17	1300	1350	460	2190	452
Be	mg/kg DS	15	1.46	1.51	0.6	2.16	0.618
C	% DS	13	11.4	17.9	0.4	56.4	17.6
CaO	% DS	76	15.5	18.7	3.53	47.94	12.5
Cd	mg/kg DS	77	0.221	2.78	0	24	4.99
Cl	% DS	65	0.035	0.104	0	0.97	0.184
Co	mg/kg DS	17	8.62	11.6	3.3	52	11.2
Cr	mg/kg DS	19	144	167	8	475	133
Cu	mg/kg DS	77	125	312	9.2	6440	785
Content of DS	% DS	19	100	88.2	42.7	100.8	18.6
Fe ₂ O ₃	% DS	76	2.34	2.35	0.23	9.15	1.55
H	% DS	13	0.4	0.477	0.2	1.6	0.39
Hg	mg/kg DS	28	0.01	0.0816	0.0011	1.1	0.241
K ₂ O	% DS	76	4.76	4.72	1.21	9.76	1.91
La	mg/kg DS	14	17.1	15.7	6	24.1	5.44
LOI, 1000 °C	% DS	11	0.2	0.509	0	3.5	1.02
LOI, 550 °C	% DS	4	1.55	1.98	0.9	3.9	1.35
MgO	% DS	76	2.26	2.83	0.3	11.19	2.06
MnO	% DS	16	0.561	0.66	0.0821	1.82	0.588
MnO ₂	% DS	58	1.01	1.42	0.23	3.64	0.985
Mo	mg/kg DS	14	6	7.56	5	23.8	4.8
Na ₂ O	% DS	76	1.35	2.18	0.16	43.2	4.9
Nb	mg/kg DS	14	6	6.13	5	7.43	0.61
Ni	mg/kg DS	19	22.9	36.9	5.3	131	35.8
N-tot	% DS	13	0.2	0.238	0.2	0.5	0.087
P ₂ O ₅	% DS	76	1.61	1.61	0.239	3.86	1.03
Pb	mg/kg DS	77	50	133	0	690	180
S	mg/kg DS	76	1080	3050	100	29000	4870
Sb	mg/kg DS	5	8.94	16.5	7.19	48.8	18.1
Sc	mg/kg DS	15	4.26	4	1.93	5.15	1.06
SiO ₂	% DS	76	33	34.8	2.77	69.6	21.2
Sn	mg/kg DS	12	20	28.3	4.75	66.3	17.1
Sr	mg/kg DS	15	308	391	256	719	156
TiO ₂	% DS	18	0.303	0.324	0.1	0.595	0.149
V	mg/kg DS	32	14.5	16.2	0	42.9	15.5
W	mg/kg DS	12	60	59.2	50	60	2.89
Y	mg/kg DS	15	14.8	13.8	8.25	16.9	2.6
Zn	mg/kg DS	77	540	1380	35	14800	2090
Zr	% DS	15	151	159	110	234	39.9

DS = Dry substance

Table A5.2 Composition of Swedish fly ash from incineration of wood (source: the Swedish database Allaska).

Data	Unit	Number	Median	Average	Min	Max	St Dev
Ag	mg/kg DS	1	2.09	2.09	2.09	2.09	0
Al ₂ O ₃	% DS	106	3.16	4.92	0.12	23.4	5.22
As	mg/kg DS	95	9.6	53.1	0.372	988	161
B	mg/kg DS	76	303	311	0	740	174
Ba	mg/kg DS	29	2130	2080	216	3810	904
Be	mg/kg DS	23	0.85	1.17	0.43	3.47	0.782
Bi	mg/kg DS	2	12.4	12.4	0.9	24	16.3
C	% DS	15	10.1	20.7	1.6	65.6	19.2
CaO	% DS	108	24	23.2	1.97	45.12	10.8
Cd	mg/kg DS	97	11	12.6	0	78	11.5
Cl	% DS	83	0.51	0.646	0.0037	4.6	0.694
Co	mg/kg DS	26	9.9	12.8	1.53	38.8	8.66
Cr	mg/kg DS	35	95.4	176	10	1050	244
Cu	mg/kg DS	102	130	231	10.5	4107	492
DM	% DS	27	100	87.7	51.1	100.9	16.7
Fe	mg/kg DS	1	14000	14000	14000	14000	0
Fe ₂ O ₃	% DS	107	1.57	1.86	0.14	5.57	1.2
H	% DS	15	0.4	0.447	0.1	0.9	0.233
Hg	mg/kg DS	41	0.12	0.456	0.0038	3.38	0.798
K ₂ O	% DS	107	5.12	5.76	0.492	30.48	4.64
La	mg/kg DS	20	12.1	13.2	6	29.3	7.25
Li	mg/kg DS	1	13.5	13.5	13.5	13.5	0
LOI, 1000 °C	% DS	21	12.1	13.3	1.4	29.4	9.16
LOI, 550 °C	% DS	10	9.7	20.1	0.15	75.3	22.9
LOI, 800/850 °C	% DS	3	3	4.3	2.8	7.1	2.43
MgO	% DS	107	3.01	3.23	0.342	7.93	1.58
MnO	% DS	39	0.903	1.08	0.0915	3.12	0.757
MnO ₂	% DS	67	1.56	1.52	0.2	3.77	0.842
Mo	mg/kg DS	24	10	12	5	33	8.06
Na ₂ O	% DS	106	0.98	1.18	0.113	3.51	0.698
Nb	mg/kg DS	19	11.1	10	6	19.8	4.25
Ni	mg/kg DS	34	43.3	65.6	4.47	262	56.9
N-tot	% DS	15	0.2	0.293	0.2	0.9	0.202
P	mg/kg DS	1	12000	12000	12000	12000	0
P ₂ O ₅	% DS	107	1.91	1.9	0.2	4.09	0.949
Pb	mg/kg DS	98	131	223	0	2470	398
Rb	mg/kg DS	2	114	114	112	116	2.83
S	mg/kg DS	94	10500	13200	3	89000	13200
Sb	mg/kg DS	5	79.4	72	0.57	136	65.1
Sc	mg/kg DS	19	2.92	2.98	1	7.73	1.59
Se	mg/kg DS	5	1	1.13	0.96	1.4	0.203
SiO ₂	% DS	107	17.2	21.2	0.08	68.8	15.9
Sn	mg/kg DS	20	20	22.9	0	131	29.8
Sr	mg/kg DS	25	597	627	72.7	1170	231
Th	mg/kg DS	2	48.9	48.9	6.89	91	59.5
TiO ₂	% DS	38	0.259	0.518	0.0065	3.02	0.77
Tl	mg/kg DS	3	2.02	234	1	700	403
TOC	% DS	2	2.85	2.85	1.4	4.3	2.05
U	mg/kg DS	2	34.8	34.8	1.6	68	47
V	mg/kg DS	50	25.1	24.3	0	98.1	23
W	mg/kg DS	19	60	36.7	0	60	26
Y	mg/kg DS	23	11.2	12.1	2	62	12
Zn	mg/kg DS	102	1710	3080	36.7	48300	5960
Zr	mg/kg DS	23	108	131	2.62	439	104

DS = Dry substance

Table A5.3 Leaching data for Swedish fly ash from incineration of wood (source: the Swedish database Allaska).

Data	Unit	Number	Median	Average	Min	Max	StDev
Al	mg/kg DS	4	1.75	137	0.1726	546	272
As	mg/kg DS	2	0.0169	0.0169	0.0056	0.0282	0.016
Ba	mg/kg DS	4	1.01	1.4	0.0832	3.5	1.64
Ca	mg/kg DS	4	291	539	103	1472	641
Cd	mg/kg DS	3	0.0004	0.0012	0.0002	0.003	0.00156
Cl	mg/kg DS	5	17.5	16.9	7.4	28	8.94
Co	mg/kg DS	3	0.00024	0.00115	0.00022	0.003	0.0016
Conductivity	mS/m 25°C	3	178	376	73.2	876	437
Cr	mg/kg DS	4	0.0415	0.313	0.01	1.16	0.565
Cr (+VI)	mg/kg DS	1	0	0	0	0	0
Cu	mg/kg DS	4	0.00116	0.00238	0	0.00718	0.00338
DOC	mg/kg DS	3	4.6	4.63	4.5	4.8	0.153
Fe	mg/kg DS	3	0.009	0.0116	0.008	0.0178	0.00539
Fluorid	mg/kg DS	1	2	2	2	2	0
Hg	mg/kg DS	2	0.00152	0.00152	4.16E-05	0.003	0.00209
K	mg/kg DS	4	162	199	59.2	410	153
Mg	mg/kg DS	2	0.686	0.686	0.45	0.922	0.334
Mn	mg/kg DS	4	0.00349	0.0037	0.003	0.0048	0.000777
Mo	mg/kg DS	4	0.119	0.129	0.04	0.24	0.0868
Na	mg/kg DS	4	134	129	59.8	189.4	60.6
Ni	mg/kg DS	2	0.00225	0.00225	0.0015	0.003	0.00106
Pb	mg/kg DS	2	0.0145	0.0145	0.003	0.026	0.0163
pH	-	4	11.9	11.7	10	13.05	1.31
Phenols	mg/kg DS	1	0.022	0.022	0.022	0.022	0
Redox potential Eh	mV	2	178	178	144	212	48.1
S	mg/kg DS	4	39.5	72.7	27.4	184.4	75.2
Sb	mg/kg DS	4	0.00615	0.00733	0.0009	0.0161	0.00636
Se	mg/kg DS	4	0.00158	0.00164	0.00011	0.00332	0.00175
SO4	mg/kg DS	4	44.5	44	0	87	49.1
Sulphate	mg/kg DS	1	600	600	600	600	0
V	mg/kg DS	3	0.0046	0.0164	0.0018	0.0428	0.0229
Zn	mg/kg DS	3	0.00458	0.0389	0.0022	0.11	0.0616

Two-stage leaching – L/S=2 l/kg, EN 12457-3
DS = Dry substance

Table A5.4 Leaching data for Swedish fly ash from incineration of wood (source: the Swedish database Allaska).

Data	Unit	Number	Median	Average	Min	Max	StDev
Al	mg/kg DS	4	17.8	324	16.6	1244	613
As	mg/kg DS	3	0.027	0.05	0.02	0.103	0.046
Ba	mg/kg DS	4	2.99	4.69	0.47	12.3	5.54
Ca	mg/kg DS	4	1080	1330	445.52	2721	1050
Cd	mg/kg DS	1	0.003	0.003	0.003	0.003	0
Cl	mg/kg DS	4	50.1	44.4	15	62.34	20.6
Co	mg/kg DS	3	0.00175	0.00215	0.0017	0.003	0.000737
Conductivity	mS/m 25°C	3	94.2	124	58	219	84.5
Cr	mg/kg DS	4	0.1	0.45	0.02	1.58	0.756
Cr (+VI)	mg/kg DS	1	0	0	0	0	0
Cu	mg/kg DS	1	0.01	0.01	0.01	0.01	0
DOC	mg/kg DS	3	16.4	15.9	12	19.18	3.62
Fe	mg/kg DS	2	0.0558	0.0558	0.04	0.0716	0.0223
Fluoride	mg/kg DS						
Hg	mg/kg DS	1	0.003	0.003	0.003	0.003	0
K	mg/kg DS	4	221	253	87.4	481.3	165
Mg	mg/kg DS	3	2.36	2.23	0.9005	3.42	1.27
Mn	mg/kg DS	4	0.0174	0.0188	0.00961	0.0309	0.00961
Mo	mg/kg DS	4	0.147	0.206	0.08	0.45	0.168
Na	mg/kg DS	4	161	161	80.39	240.84	72.2
Ni	mg/kg DS	3	0.0062	0.006	0.005	0.0068	0.000917
Pb	mg/kg DS	2	0.0225	0.0225	0.0151	0.03	0.0105
pH	-	4	11.5	11.5	10	12.98	1.25
Phenols	mg/kg DS	1	0.06	0.06	0.06	0.06	0
Redox potential Eh	mV	2	189	189	167	211	31.1
S	mg/kg DS	4	166	166	33.6	298.5	137
Sb	mg/kg DS	3	0.0283	0.0332	0.014	0.0574	0.0221
Se	mg/kg DS	4	0.002	0.00225	0	0.005	0.00222
SO ₄	mg/kg DS	4	151	305	0	916	416
Sulphate	mg/kg DS	1	898	898	898.43	898.43	0
V	mg/kg DS	3	0.032	0.048	0.026	0.086	0.033
Zn	mg/kg DS	2	0.065	0.065	0.01	0.12	0.0778

Two-stage leaching – L/S=10 l/kg, EN 12457-3

DS = Dry substance