Bottom ash and APC residue management

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Abstract
The management of residues from waste incineration aims for inertisation of the bottom ashes and minimisation of the amount of hazardous fly ashes and gas cleaning residues while still meeting the emission standards. This should for economic reasons mainly be reached by in-plant measures. Strategies to produce a bottom ash with utilisation properties and to inertise other solid residues are presented. The leaching stability as most important environment related quality parameter is addressed. The costs of the existing treatment and disposal options are discussed.

1 Introduction
Some ten years ago the debate about thermal processes was mainly focused on potential risks of air emissions, especially those related to dioxins. Meanwhile the gas cleaning devices implemented in municipal solid waste incineration (MSWI) plants are among the most effective ones found in any technical process and the interest is more directed to the quality of the solid combustion residues. The aims are to produce as far as possible inertised bottom ashes and to enable their utilisation as secondary building materials. This is especially promoted in the Netherlands, Denmark and Germany, recently also in France. A further focus is the inertisation and safe disposal - or even utilisation - of the filter and boiler ashes as well as of the gas cleaning residues.

All kinds of secondary treatment processes have been developed to tailor the residue quality according to special needs. Secondary measures, however, are expensive and hence the better approach is an optimised control of the combustion process to
- guarantee an excellent burnout of carbon compounds,
- promote the volatilisation of heavy metals like Hg and Cd out of the fuel bed, and
- fixate lithophilic elements in the silicatic and oxidic matrix of the bottom ash, thus reducing their leachability.

The following discussion of the quality of residues from modern waste incineration plants will follow these objectives. Finally it will investigate which rational options exist to inertise and eventually utilise filter ashes and flue gas cleaning residues. All considerations base mainly on the results of an international perspective on the characterisation and management of waste incineration residues published by the International Ash Working Group in 1997 [IAWG 1997].
2 Mass streams in a MSWI

The basis of all discussions about waste incineration residues is the knowledge of the different mass streams in a municipal solid waste incinerator. Fig. 1 shows average ranges for these streams as found in modern mass burning systems. The air consumption is approx. 4500 m³/Mg of waste.

State-of-the-art plants produce typically between 200 and 300 kg bottom ashes per 1 Mg of burnt waste. Most published numbers include the grate siftings which are only recently and only in some countries kept separate from the bottom ash. The mass flow of siftings depends on the type of grate and its time of operation. The siftings may increase the amount of unburnt matter in the bottom ash. In view of utilisation, however, the inventory of metallic Al which drips through the grate voids is of much higher concern.

The production of boiler ash depends on the type of boiler and on the amount of dust originally released from the grate. Boiler ash should be treated together with the filter ash due to its similar level of toxic heavy metals and organics. In some countries this has already been enforced by legislative regulations. The data presented for filter ashes reflect the situation in modern plants which try to establish a more gentle combustion with dust loads down to less than 2 g/m² [Vogg 1991].

The mass flow of air pollution control (APC) residues shows actually the highest variation of all residues. The given 10 - 12 kg/Mg is a mean value for wet systems which operate close to stoichiometry. The number comprises the dry neutral sludge (2 - 3 kg/Mg) and the soluble salts (8 - 9 kg/Mg). In semi-dry or dry systems the amount is increased because of unreacted additives.

3 Bottom ashes

3.1 Disposal and utilisation regulations

Waste incineration is performed to produce an inertised residue, the bottom ash, which meets the respective disposal standards. Many countries aim for utilisation of this residue stream in order to save space on landfill sites.

Selected German standards for disposal on landfill class 1 and for utilisation of bottom ash in road construction [LAGA 1994] are compiled in Table 1. The table indicates an only small difference between the requirements.

<table>
<thead>
<tr>
<th>Table 1</th>
<th>Selected German standards for disposal and utilisation of MSWI bottom ash</th>
</tr>
</thead>
<tbody>
<tr>
<td>unit</td>
<td>landfill class 1</td>
</tr>
<tr>
<td>LOI</td>
<td>wt-%</td>
</tr>
<tr>
<td>TOC</td>
<td>wt-%</td>
</tr>
<tr>
<td>DEV S4</td>
<td></td>
</tr>
<tr>
<td>soluble fraction</td>
<td>wt-%</td>
</tr>
<tr>
<td>el. conductivity</td>
<td>mS/m</td>
</tr>
<tr>
<td>Cl</td>
<td>mg/l</td>
</tr>
<tr>
<td>Cu</td>
<td>mg/l</td>
</tr>
<tr>
<td>Zn</td>
<td>mg/l</td>
</tr>
<tr>
<td>Cd</td>
<td>mg/l</td>
</tr>
<tr>
<td>Pb</td>
<td>mg/l</td>
</tr>
</tbody>
</table>
for disposal and utilisation and the challenge is to reach the utilisation quality without further post-combustion treatment.

In the case of utilisation as secondary building material additional standards are set for mechanical properties like density, mechanical strength, grain size distribution or freeze-thaw-stability. This aspect, however, will not be discussed here.

### 3.2 Burnout

The burnout is the key parameter for disposal as well as for utilisation of bottom ashes. The German Technical Ordinance Residential Waste sets a TOC (total organic carbon) limit of 1 wt.-% for disposal on a class I landfill. The same number is found in the LAGA memorandum for utilisation in road construction.

In modern well operated MSWI plants the TOC in bottom ashes is typically well below 1 wt.-% [Schneider 1994, Bergfeldt 2000]. Special combustion trials in the Karlsruhe test incinerator TAMARA demonstrated that an increasing heating value of the feed and the resulting higher bed temperatures improve the burnout of bottom ash (see Fig. 2) [Vehlow 1994].

The TOC of bottom ashes comprises mainly elementary carbon, but to a certain extent also organic compounds are found which cover the spectrum from short-chain compounds [Köster 1998] up to low volatile species such as PAH or PCDD/F. Typical concentrations of organic compounds in the various solid residues are compiled in Table 2.

![Fig. 2 Residual carbon in TAMARA bottom ashes versus heating value of the feed](image)

**Table 2** Concentration ranges of organic compounds in bottom, boiler, and filter ashes

<table>
<thead>
<tr>
<th>parameter</th>
<th>bottom ash ng/g</th>
<th>boiler ash ng/g</th>
<th>filter ash ng/g</th>
</tr>
</thead>
<tbody>
<tr>
<td>I-TEQ</td>
<td>$&lt;0.001 - 0.01$</td>
<td>$0.02 - 0.5$</td>
<td>$0.2 - 10$</td>
</tr>
<tr>
<td>PCB</td>
<td>$&lt;5 - 50$</td>
<td>$4 - 50$</td>
<td>$10 - 250$</td>
</tr>
<tr>
<td>PCBz</td>
<td>$&lt;2 - 20$</td>
<td>$200 - 1000$</td>
<td>$100 - 4000$</td>
</tr>
<tr>
<td>PCPh</td>
<td>$&lt;2 - 50$</td>
<td>$20 - 500$</td>
<td>$50 - 10000$</td>
</tr>
<tr>
<td>PAH</td>
<td>$&lt;5 - 10$</td>
<td>$10 - 300$</td>
<td>$50 - 2000$</td>
</tr>
</tbody>
</table>

Only data from modern facilities have been used as basis [Johnke 1995, Schneider 1994, Bergfeldt 1997]. The PCDD/F numbers are given in terms of international toxic equivalence data (I-TEQ). It is evident that the organic pollution is higher in the boiler and fly ashes than it is in the bottom ash. The I-TEQ levels detected in the bottom ashes of modern incineration plants were in the same order of magnitude as found in uncontaminated soils in Germany [Bergfeldt 2000].

### 3.3 Chemical and mineralogical characterisation

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. This is demonstrated by the concentration ranges of selected metals depicted in Fig. 3 [IAWG 1997]. Some heavy metals, e.g. As, Cd, or Hg are to a great extent volatilised...
out of the fuel bed and show eventually lower concentrations in the bottom ashes than in the waste. The graph contains the respective concentration ranges in filter ashes, too, and it is evident that - with the exception of the mainly lithophilic Cu - all other selected heavy metals are highly enriched in these materials. For comparison the concentrations in the lithosphere are enclosed, too.

Apart from the chemical analysis a geochemical and mineralogical characterisation provides useful information in view of the long term behaviour of a material. Bottom ashes can be characterised as a mixture of silicatic and oxidic phases. Some typical mineral phases found in these residues are shown in the micrographs in Fig. 4 [Pfrang-Stotz 1992]. These phases do not only tell about the structure of the bottom ash but can in special cases also supply information about the temperature, the material has been exposed to on the grate. This important number which controls mainly the fate of elements in the combustion chamber, is widely unknown in full scale plants. The knowledge of formation temperatures of single phases and the specific search for high-temperature phases are promising ways to obtain better information in this area [Pfrang-Stotz 1993].

### 3.4 Leaching stability

#### 3.4.1 Leaching fundamentals

The chemical composition of a product does in principle not allow to evaluate its environmental impact. This is far more depending on the leaching stability of the material in question. Even if the matrix and the speciation of single elements were known, a reliable theoretical prediction of the short- and long-term behaviour is more or less impossible. The most important parameters influencing the leaching stability of a material are enumerated below:

- its chemical composition,
- its chemical/geochemical/mineralogical speciation,
- the fraction of a species available for leaching,
- the particle morphology,
- the properties of the leachant, especially its pH or the presence of complexing constituents,
- the liquid-solid ratio (LS) in the leaching system.

<table>
<thead>
<tr>
<th>Element</th>
<th>Concentration in mg/kg</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cu</td>
<td>0.01 - 10</td>
</tr>
<tr>
<td>Zn</td>
<td>0.01 - 10</td>
</tr>
<tr>
<td>As</td>
<td>0.01 - 10</td>
</tr>
<tr>
<td>Cd</td>
<td>0.01 - 10</td>
</tr>
<tr>
<td>Sb</td>
<td>0.01 - 10</td>
</tr>
<tr>
<td>Hg</td>
<td>0.01 - 10</td>
</tr>
<tr>
<td>Pb</td>
<td>0.01 - 10</td>
</tr>
</tbody>
</table>

**Fig. 3** Concentration ranges of selected elements in various materials.

**Fig. 4** Micrographs of minerals in bottom ashes: glass formed during combustion (left), gehlenite $\text{Ca}_2\text{Al}_2\text{Si}_3\text{O}_{12}$ in glassy matrix (centre), magnetite, $\text{Fe}_3\text{O}_4$ (right) [Pfrang-Stotz 1992]
It seems evident, that no single - and on top of that - simple test procedure will deliver results that allow a sound evaluation of impacts on the environment. In fact a great number of different tests has been developed to get detailed information about the leaching properties of residues from waste incineration.

There are two categories of test procedures: extraction tests and dynamic tests. The most common principles of these test categories are comprised in Fig. 5.

Extraction tests allow the determination of leaching equilibria. If the leachate is analysed in time increments before the equilibrium is reached, information on the kinetics of the system can be obtained. An example of such tests is the Dutch tank leaching test [NEN 7345] for stabilised materials. This test gives also indication of the major parameters controlling the leaching process, e.g. diffusion or solubility. Sequential tests in different leachants of increasing chemical strength are often used to investigate the chemical bond of specific elements in the matrix.

Dynamic tests are applied to reveal the kinetics of the leaching process. A rather common one is the Dutch column leaching test [NEN 7341] for granular material. This test is typically performed up to an accumulated LS of 10 and enables the modelling of contaminant release during 50 – 100 years.

In practice a material in question is subject to a number of different tests which are selected to model as close as possible the envisaged disposal or utilisation scenario. After the fundamental properties have been acquired, an indicator test - in most cases a standardised test procedure - is chosen to control the quality of an actual sample in short time and with limited effort.

Almost all regulations for the disposal or utilisation of waste products are based on standardised leaching tests, unfortunately different ones in different countries. Hence the testing is done under country specific conditions and the interpretation of the results of various tests has to take such differences into account.

The most important parameter influencing the results of a leaching test is the pH of the leachant. Fig. 6 gives a schematic overview of the influence of the pH of the leachant upon the solubility of metal cations and anions in aqueous solutions. It is well known that most heavy metals show rather low solubility in the weak alkaline range. Their solubility increases with decreasing pH. In the alkaline region different metals behave differently: some (e.g. Cd)
stay insoluble with increasing pH. Others, the amphoteric ones, are more or less solubilised if the pH is elevated. The amphoteric metal of highest interest in waste incineration is Pb.

Metals which tend to form anions in aquatic solutions like V, Cr, or Mo, have their highest solubility close to the neutral point.

The graph in Fig. 6 indicates the ranges of pH which establish in selected leaching tests of bottom ashes. The German DEV S4 [DIN 38 414] (LS=10, 24 h) shows numbers between 10 and > 12. Almost the same procedure is used in France with the X31-210 AFNOR leach test [Normalisation française 1988]. These varying conditions have severe impacts especially on the test results of Pb.

The Swiss TVA test [Schweizerischer Bundesrat 1990] (2 tests at LS = 10, 24 h each) is characterised by a rather constant pH of 5.5 - 6 due to the gaseous CO₂ bubbling through the test solution.

Constant pH values are used for the Dutch total availability test [NEN 7341] which gives information about the leaching potential under assumed ‘worst’ environmental conditions. The cation solubility is tested at a pH of 4, that of the anions at a pH of 7. The sample has to be finely ground in order to exclude any inhibition of the leaching by diffusion and the liquid-solid ratio is kept at 100 to avoid saturation effects in the solution.

The standardisation committee of the EU has recently proposed the leaching procedure prEN 12457 for crushed bottom ashes. A 6 h test at LS = 2 is followed by a second leaching for 18 h at LS = 8 [European Committee 1999]. The first part of this test has been adopted by the Danish authorities for quality control of bottom ashes. The test is not pH controlled. For the time being only limited knowledge exists how results from this test compare to other tests.

### 3.4.2 Effect of aging

In order to optimise the total burnout the combustion temperature and with this also the fuel bed temperature has been elevated in MSWI plants during the last decade. As an effect of such operation changes a higher formation of CaO can be seen. The pH value of fresh bottom ashes is often exceeding 12. According to the German LAGA memorandum bottom ashes have to be stored for 12 weeks prior to utilisation in road construction. During this time the uptake of CO₂ from the air converts the earth-alkali oxides into carbonates and neutralises part of the alkalinity. Hence aged bottom ashes establish a pH of about 10 - 11 in the DEV S4 test.

Data from a test program in a German full scale waste incineration plant illustrate the effect which aging has on the pH of bottom ashes and on the test results obtained by the DEV S4 method [Bergfeldt 1997]. Fig. 7 documents that the pH of the fresh bottom ashes in the DEV S4 test is typically exceeding 12 and drops down by about two units during the aging process. As can be seen in Fig. 8, this pH change has 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 2 orders of magnitude.
This strong interdependency is responsible for the strange situation, that due to German regulations fresh bottom ashes from some plants do not comply with the landfill standards while after aging they are excellent secondary building materials.

### 3.5 Potential for utilisation

As mentioned above, a number of countries has or is going to set standards for the utilisation of bottom ashes. The major application area is road construction where ashes are used in the support layers mainly under water tight capping. The requirements for leaching stability are more or less of equal stringency in all countries all over the world. The German guideline regulating utilisation in road construction is the above mentioned LAGA memorandum.

Fig. 9 demonstrates for 26 samples taken routinely during one year on an industrial ash treatment site, that the test results for the environmentally interesting heavy metals were always well below the respective standards [Pfrang-Stotz 1995]. The only component exceeding the limit in few cases was sulphate. This limit has been set to protect concrete structures from corrosion attack. Hence it can be stated, that bottom ashes from modern and well operated MSWI plants do easily meet the LAGA limits for utilisation.

Other constituents of concern are soluble salts, mainly alkali and earth-alkali chlorides and sulphates. Chlorides can be reduced by washing of the ashes [Schneider 1994]. The simplest way is a washing in the quench tank which is already performed in some German plants. The sulphate solubility is controlled by the solubility equilibrium of the predominant earth-alkali sulphates. A stabilisation or removal is hence difficult.
The compliance with standards fulfils the legislative requirements but does not necessarily tell about the acceptability of the environmental impact. To get a clue about this aspect the DEV S4 test was also applied to samples of concrete from a demolished highway bridge. The test results of four metals in terms of concentrations are displayed in Fig. 10 together with those of the 26 bottom ash samples. The bar chart gives evidence that the leaching stability of aged high quality bottom ashes can be kept in the same order of magnitude as that of conventional building materials. Hence there is no reason not to utilise - after careful testing - bottom ashes from modern waste incineration plants.

This is common practice in countries which have geological conditions hampering the siting of landfills like The Netherlands or Denmark. These countries utilise up to 90 % of the bottom ashes [Sakai 1996]. The respective number for Germany is approx. 60 % [Johnke 1995]. Some other countries like France are nowadays as well encouraging bottom ash utilisation.

A different strategy is followed by the Swiss authorities. According to their regulations, bottom ashes are categorised as reactive residues. Only stone-like materials are accepted as in building materials and stone-like refers to the concentration and not to the mobility of a single constituent. Since the bottom ashes contain higher amounts of heavy metals than the lithosphere (compare Fig. 3), almost no utilisation is practised. Bottom ashes have to be disposed of or they have to be converted into real stone-like materials by adequate measures.

### 3.6 Quality assurance by sintering

The good leaching stability of bottom ashes presented above needs to be reached permanently and this gives reason to ask, how to guarantee such high quality. The best approach seems to establish a high temperature in the fuel bed for volatilisation of mobile metals and immobilisation of the lithophilic ones by sintering. Since sintering is a solid phase re-speciation, higher residence time improves the effect.

This strategy has been investigated in laboratory scale by sinter experiments using fresh bottom ashes from two German incineration plants [Schneider 1994]. The ashes have been annealed under air atmosphere at temperatures of 850, 1000 and at 1300 °C for 30 min each. At the latter temperature the material was melted. The resulting DEV S4 leaching data of the products of these tests are depicted in Fig. 11.

The graph indicates a significant improvement of the leaching stability of four selected metals by the treatment. At 850 and 1000 °C comparable effects were observed. The fusion, however, did not improve the elution stability significantly. This finding is supported by the comparison of test results from bottom ashes with those published for molten residues from high-temperature processes like Thermoselect or the Siemens Thermal Waste Recycling Process [Vehlow 1995].
The stabilisation by sintering could also be validated in semi-technical experiments with fresh bottom ashes from a full scale incineration plant [Bergfeldt 1997].

Based on these results it can be concluded, that a sintering at temperatures of 850 °C has a stabilising effect upon heavy metals. The energy consuming - and that means expensive - fusion, however, does not pay since no significant further fixation could be observed. Hence a simple in-plant measure to produce bottom ashes of high leaching stability can be recommended: the bed material should be kept at high temperature at the back end of the grate.

4 Filter ashes and APC residues

Filter ashes and to a certain extent boiler ashes, too, carry substantial loads of volatilised heavy metals (as has been documented above in Fig. 3) and of low volatile organic compounds (compare Table 2).

Wet and (semi-)dry gas cleaning systems produce different amounts of scrubbing residues which are different in quality, too. Their major constituents are water soluble salts derived from the removal of acid gas constituents. The main waste inventory of Hg is discharged along with these residue streams. Furthermore, contaminants like organic compounds and - depending on the quality of fly ash removal - traces of other heavy metals are found.

The filter ashes as well as the scrubbing residues are classified as hazardous waste in almost all legislations and consequently the only safe disposal is that on an adequate special disposal site, preferentially in the underground in old salt mines (as preferred in Germany). The alternative, the inertisation of these residues will be addressed in the next chapter.

5 Treatment and costs

5.1 Treatment principles

Many efforts have been made to improve the environmental quality of residues from waste incineration by secondary treatment and to recycle or utilise at least parts of specific residues. A compilation of proposed strategies and processes is shown in Fig. 12. The disposal/utilisation in salt mines is a German speciality and will briefly be discussed below.

To assess the usefulness of post-combustion treatment it is necessary, not only to consider the environmental benefits of a measure but also to set the obtained improvement into relation to the spent effort. The measure for the effort should be the cost of the process. In other words: a real eco-balance is needed.
The International Ash Working Group identified a number of principles which have to be considered when assessing the benefits but also the obstacles of a given treatment measure:

- Does the process result in a significant quality improvement?
- Does the process impose any health, environmental, or safety impacts?
- Are there secondary residues and where do they end up?
- Is there a final product of high quality?
- Is there a long-term market for that product?
- What is the cost of the process?

It is not easy to answer these questions in particular, the more so if the respective process has not been tested in full scale. This applies especially for the costs. In view of the total process costs of waste incineration an expensive treatment process might be acceptable for a small residue stream like filter ashes, for the bottom ash, however, even moderate process costs are prohibitive.

### 5.2 Bottom ashes

Especially in Japan fusion or vitrification of bottom ashes is practised in order to reduce their volume and to improve their environmental quality. In other countries like Germany such processes have been proposed, but did not enter the market for economic reason.

As has been documented above, bottom ashes from modern waste incineration plants have the potential to be utilised as secondary building material in road construction - and there is a permanent requirement for such material. It is also evident, that fusion of bottom ashes from stat-of-the-art MSWI plants does not improve the quality to an extent which would open new markets.

**Table 3** Cost estimates for land filling and treatment of bottom ashes

<table>
<thead>
<tr>
<th></th>
<th>€/Mg of bottom ash</th>
<th>€/Mg of MSW</th>
</tr>
</thead>
<tbody>
<tr>
<td>Landfill</td>
<td>35</td>
<td>12</td>
</tr>
<tr>
<td>Pre-treatment</td>
<td>20</td>
<td>7</td>
</tr>
<tr>
<td>Fusion (fossil fuel, no pre-treatment)</td>
<td>100</td>
<td>30</td>
</tr>
<tr>
<td>Fusion (fossil fuel, scrap removal)</td>
<td>130</td>
<td>45</td>
</tr>
<tr>
<td>Fusion (electric heating)</td>
<td>120</td>
<td>40</td>
</tr>
<tr>
<td>Fusion processes in Japan</td>
<td>100</td>
<td>30</td>
</tr>
<tr>
<td>Fusion processes estimates (IAWG)</td>
<td>180</td>
<td>60</td>
</tr>
</tbody>
</table>

Table 3 compiles estimates of costs of various treatment options for bottom ashes taken from literature [Vehlow 1997]. Considering German conditions it makes sense to utilise bottom ashes, since the expenses for the pre-treatment are similar to those for land filling. Furthermore, it can be expected that the latter ones will increase with time. Fusion, however, should only be applied if the high costs can be justified by either respective revenues or long-term benefits of other kind.

### 5.3 Filter ashes

Boiler and even more filter ashes are classified as special wastes in many legislative regulations and their final destination is in most countries a disposal on special and expensive disposal sites. That is why numerous attempts have been made to detoxify these materials in order to get access to less expensive disposal routes. The applied principles are pointed out in Fig. 12.
A broad spectrum of different processes has been proposed and tested in different scales. Table 4 tries to categorise the various treatment options. Without going into detail it seems evident that solidification or stabilisation does not alter the toxic inventory of the material. The established transformation or diffusion barrier does only last for a limited time. Two processes are in full scale application: the 'Bamberg Model', where filter ashes are stabilised on a landfill by mixing with the sludge of the wet scrubber discharge neutralisation [Reimann 1990], and the Swiss filter ash cement stabilisation after washing [Tobler 1989].

Thermal treatment can be performed at moderate temperatures (400 °C) to destroy dioxins or at high temperatures (>1300 °C) to produce glassy products. The latter option has been tested in many variants during the early nineties. Most processes allow a certain recycling of metals. Vitrification is mainly favoured in Japan. The molten products are distinguished by excellent elution stability. Care has to be taken to avoid air pollution by evaporation of metal compounds. The energy consumption of all of these processes, however, is very high and that is why such processes did not conquer the market in Europe.

A third strategy - more in line with the demand for simple and in-plant measures - is followed by the 3R Process which combines an acid extraction of soluble heavy metal compounds (by use of the acid flue gas cleaning solution) with a thermal treatment of the compacted extraction residues in the combustion chamber [Vogg 1984]. A scheme of the process is shown in Fig. 13. The technical demonstration revealed that the 3R Process is a sink not only for mobile heavy metals but also for toxic organics [Vehlow 1990].

The costs of the various filter ash treatment options are estimated on the basis of published data in Table 5 [Vehlow 1997]. Again, as in the case of bottom ash treatment, the costs of technical processes should be comparable in most industrialised counties whereas the disposal fees will change from country to country.

The table reveals that the specific costs of the technical measures are rather high, but due to the small residue streams the expenses per ton of waste are low and similar for all

<table>
<thead>
<tr>
<th>Table 4</th>
<th>Procedures for treatment of filter ashes</th>
</tr>
</thead>
<tbody>
<tr>
<td>principle</td>
<td>process</td>
</tr>
<tr>
<td>solidification/ stabilisation</td>
<td>without additives (Bamberg Model) cement based systems (Portland cement, alinite) waste pozzolanic systems (coal fly ash) chemical stabilisation (sulphides, TMT 15™) organic additives or matrix (bitumen)</td>
</tr>
<tr>
<td>thermal treatment</td>
<td>PCDD/F destruction (Hagenmaier drum) sintering (mineral respeciation) fusion (melting without additives) vitrification (melting with additives)</td>
</tr>
<tr>
<td>combined process</td>
<td>acid extraction + sintering (3R Process)</td>
</tr>
</tbody>
</table>

Fig. 13 Scheme of the 3R Process
Hence the economy will not be the decisive factor for the selection of a specific process and local conditions like access to adequate disposal sites will be more important.

### 5.4 APC residues

Flue gas cleaning processes, at least in Germany, are in principle not allowed to discharge waste water and the evaporation of the scrubber effluents is mandatory for wet systems. The resulting residues and those of dry or semi-dry APC systems carry high levels of soluble salts, especially of alkali and earth-alkali chlorides or sulphates. Due to the high solubility a safe disposal can only be guaranteed on special and expensive sites. Attempts have been made to utilise parts of the ingredients of these residues in order to minimize the disposal problem. The challenge is the closing of the chlorine cycle. Different processes to recover NaCl [Karger 1990], HCl [Kürzinger 1989], or Cl₂ [Volkman 1991] have been tested. All such processes can only be successful if they end up with high quality products and if there is a long-term market for the products. Today e.g. in Germany only few MSWI plants produce HCl.

A different - and finally very cheap - way of disposal of filter ashes (and APC residues) has been opened recently in Germany where authorities enforce the backfilling of cavities in old mines. Salt caverns are already being filled by semi-dry flue gas cleaning residues from MSWI in big bags [Plomer 1995]. This strategy - which is even accepted as 'utilisation' - may be justified with the similar chemical as well as physical properties of the original salt and the disposed residues. However, for likewise activities in old coal mines this argument can hardly be used.

<table>
<thead>
<tr>
<th>Table 6</th>
<th>Cost estimates for land filling and treatment of scrubbing residues</th>
<th>€/Mg of MSW</th>
</tr>
</thead>
<tbody>
<tr>
<td>dry sorption</td>
<td></td>
<td>36</td>
</tr>
<tr>
<td>dry sorption without residue disposal</td>
<td></td>
<td>23</td>
</tr>
<tr>
<td>dry sorption with utilisation for backfilling of caverns</td>
<td></td>
<td>32</td>
</tr>
<tr>
<td>semi-dry sorption</td>
<td></td>
<td>29</td>
</tr>
<tr>
<td>wet scrubbing with waste water discharge</td>
<td></td>
<td>25</td>
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<td>wet scrubbing with spray dryer</td>
<td></td>
<td>28</td>
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<tr>
<td>wet scrubbing with external evaporation</td>
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<td>29</td>
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<tr>
<td>wet scrubbing with HCL/gypsum production</td>
<td></td>
<td>35</td>
</tr>
</tbody>
</table>

Since the mass flow and properties of residues from gas cleaning depend on the applied strategy, it seems not useful to discuss their specific disposal or treatment costs. Hence the attempt has been made in Table 6 to compare the costs of the respective flue gas cleaning strategies. The base of the data and their validity is the same as in the above outlined cost considerations.

Like in the case of the filter ashes the economy of the various options does not differ significantly and again local conditions will be decisive for the most adequate strategy. In Germany the underground 'utilisation' looks economically promising. The gate fee has dropped down to approx. 40 - 70 € per ton of material. As a consequence dry scrubbing processes may be promoted which is in contradiction to the legislative demand for
residue minimization. If the strategy gains wide application, however, it will change the management of residues from APC systems in future at least in Germany, where a great number of old mines is waiting to be filled.

6 Conclusions and recommendations
For the optimisation of waste incineration in view of high quality bottom ashes and the safe and sustainable management of filter ashes and APC residues some fundamental strategies are recommended:

- Adequate combustion control and careful sintering of the bed material at the back end of the grate guarantee an excellent burnout and cause a good fixation of heavy metals.
- Simple washing of the bottom ashes, preferentially in-plant in a modified quench tank, reduces the leaching of chlorides to very low levels.
- The resulting products have a high potential for utilisation, e.g. according to German regulations in road construction.
- Post-combustion treatment of bottom ashes increases the incineration cost without improving the elution stability significantly.
- The economy is no decisive parameter for the special treatment of fly ashes and air pollution control residues.

Most problems in the field of residue management are well understood today and in most cases appropriate technologies exist already. It is obvious that primary and in-plant measures have to be preferred rather than secondary post-combustion techniques.

All processes intended for quality improvement have carefully to be analysed whether they result in real ecological benefits, whether all potential impacts upon the environment are taken into consideration, and whether these benefits pay in view of effort and expenses. Especially the last criterion - which has been mentioned as a decisive factor even in the latest German waste directive - is often pushed aside in political discussions.

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