Subject:	FW: Exothermal Reactions in Bottom Ash Monofills ref MEHL application WO 127-03
Attachments:	ATT00001..txt; schluss_poster.pdf; Klein_JHazardMat_2003.pdf; Klein_JHazardMat_2001.pdf; ATT00001..txt; schluss_poster.pdf; Klein_JHazardMat_2003.pdf; Klein_JHazardMat_2001.pdf; res-02-helsinki.pdf

From: Paddy Boyle [mailto:paddyboylerush@hotmail.com]
Sent: 07 August 2012 20:07
To: Wexford Receptionist
Subject: Exothermal Reactions in Bottom Ash Monofills ref MEHL application WO 127-03

To: Mr Frank Clinton
Programme Manager
Environmental Licensing Section
EPA

Ref: Our request for a meeting with EPA Inspector Brian Meeney and Dr Marcus Ford dated 16/7/2012.

Dear Mr Clinton please copy and forward additional attached Email from Dr Baumann and article "Bottom Ash and APC residue management" by J. Vehlow,(2002), Forschungszentrum Karlsruhe institute for Technical Chemistry.data

Please also see link to Dr. Baumanns website on Bottom Ash Monofill Trials in Bavaria (1998-2002)
http://www.ws.chemie.tu-muenchen.de/groups/hydrogeo/research/reactions-bottom-ash/

To: Mr Brian Meeney, EPA inspector and Dr Marcus Ford

Dear Sirs,
The attached Email, files and links are all with ref to the proposed method of disposal of bottom ash at MEHL as outlined in the EIS and An Bor Pleanala Oral hearing, inspectors report and Final decision. It should also be read in conjunction with internal EPA reports on the current disposal of Botton ash from the Indaver incinerator at White Water landfill County Louth and most probable indications of exothermic reactions and associated overheating problems.

Our were concerns were expressed at a meeting with representatives of Indaver at the outset of this project at the proposal to transport and deposit uncured bottom ash at the MEHL facility, but Indaver refused to countenance pre-storage of the fresh ash to allow for stabilization and the lowering of the pH of the material to safer levels. (see Article by J. Vehlow, Fig.7).
Our concerns were based primarily on the following

- The distance to the nearest residence was less than 100m (it subsequently transpired at the ABP oral hearing that the distance at the Indaver landfill at Antwerp is approx. 2,500m). The possibility of concentrated windblown irritant ash and other unknown emissions at the MEHL hilltop site was a very real concern to residents.
- The possibility of overheating of the bottom ash and destruction of the PVC liner and dessication of the underlying clay barrier, with consequent escape of leacate containing raised levels of heavy metals (see J Vehlow, Fig.3) and attached link to Munich Technical University Hydrogeology Dept website (see above). Here again the underlying conditions at the Antwerp facility is entirely different as the facility is located at Antwerp Port, the groundwater most probably not potable, and the dilution factor of escaping leachate or surface water emissions much greater than at Hollywood.

We were highly disappointed at the dismissal by the EPA of our concerns as unfounded.

In our recent phone contact with Dr. Baumann he confirmed that he also had great difficulty convincing the Bavarian authorities of the very real problems involved in the safe disposal of bottom ash by conventional methods, but these were subsequently proven by the joint trials and computer predictions. (see attached reports kindly supplied by Dr. Baumann in his recent Email to us)

1

We now perceive the problems to be addressed briefly as follows

- Bottom ash not stabilized to some degree before disposal will always constitute a danger of overheating.
- The optimum stabilization time is 12 weeks in two meter high exposed piles.
- Untreated bottom ash will have the heating effect of ~ 45 watts per m3, reducing according to the "storage" time allowed.
- Heat escapes mainly from the landfill surface so that the surface must be kept uncoered at all times even up to a year after closure.
- Depositing must be done by layering, the thickness of the layers (1m- 3m) inversely proportional to the time allowed for storage.
- The time delay between subsequent layers (1-3months) is inversely proportional to the underlying layer thickness and the ambient temperature.
- Precipitation has little effect on cooling.
- Some minimum storage must take place at the incinerator (1-3 weeks) to allow the ash to dry to between 10-20% moisture content, but this must be followed by a minimum storage at the landfill of a similar timescale.
- Dr Baumann confirmed that the temperature did if fact reach 100degrees C in some trials.
- He also said that the reason the phenomenon occurs is mainly due to the high insulation properties of the ash and the presence of air in the ash pores.
- My own opinion is that the current proposal to deliver wet ash will only lead to eventual delayed serious overheating as the curing will be delayed by the initial presence of water in the pores, and subsequent layering would be seriously delayed.
- Similarly the suggestion that a cooling system could be installed using pipework and surface water or leachate injection could have the opposite effect on overheating to that intended. in the case of wet ash leading to increased rate of reaction and in the case of drier ash leading to oversupply of air.

The only current proven safe method of disposal of bottom ash is as outlined above by Dr Baumanns team and is therefore current BAT. It follows from this study that MSWI bottom ash disposal should be only allowed at a remote location and pose no danger to groundwater in the event of accidental overheating.

Given this indisputable fact the residents of Hollywood would therefore request the EPA to reconsider whether the Hollywood hilltop site with its proximity to residents and unprotected groundwater aquifer is a safe location at which to deposit uncovered MSWI bottom ash for the next 30 years.

Yours truly,

Patrick Boyle, BE, M.Phil. for Hollywood and District Conservation Group

From: tbaumann@tum.de
To: paddyboylerush@hotmail.com
Subject: Re: Exothermal Reactions in Bottom Ash Monofills
Date: Sat, 4 Aug 2012 10:37:32 +0200

Dear Mr Boyle,

please find attached two reprints on the temperature development in a
municipal waste incinerator bottom ash disposal and a poster
(unfortunately in german) summarizing the results of our research
project sponsored by the Bavarian State Ministry of the Environment.

Our measurements, mineralogical data, and modelling results indicate
that the temperature development can be controlled by removing metals,
intermediate storage and layered emplacement into the landfill. While
removal of metals decreases the exothermal reactions, intermediate
storage promotes the development of less reactive coatings thus leading
to diffusion limited processes and a layered emplacement assists the
heat transfer to the surrounding, thus avoiding hot spots in the
disposal.

2

I hope that you will find this information useful and I will be ready to
answer further questions in late September.

Best
Thomas Baumann

-- PD Dr. Thomas Baumann Head of Hydrogeology Group Institute of Hydrochemistry Technische Universitaet
Muenchen Marchioninistr. 17 D-81377 Muenchen Voice: +49 89 2180-78234 Fax: +49 89 2180-78255
http://www.ws.chemie.tu-muenchen.de/hydrogeo

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Joe Reilly Sub No. 72

Subject:	FW: Exothermic reactions etc Wo127-03

From: Paddy Boyle [mailto:paddyboylerush@hotmail.com]
Sent: 07 August 2012 20:45
To: Wexford Receptionist
Subject: Exothermic reactions etc Wo127-03

Dear Sirs ref my submission at 20.03 today please make the following correction

"We were highly disappointed at the dismissal by **ABP** of our concerns as unfounded".

Yours truly,
Patrick Boyle

1

Exothermer Stoffumsatz in MVA-Schlackedeponien

T. Baumann, R. Biber, C. Speiser, R. Nießner

Institut für Wasserchemie, Technische Universität München, Marchioninistr. 17, D-81377 München

1 Motivation

- Nach der Ablagerung von Schlacken aus Müllverbrennungsanlagen auf Deponien setzen vielschichtige Umsetzungsprozesse ein.
- In der Summe sind diese Prozesse exotherm, d. h. es kommt zu einer Erwärmung innerhalb der Deponie, wobei Temperaturen bis 80°C erreicht wurden.
- Die Erwärmung kann zu einer Schädigung der Deponiedichtungssysteme durch thermischen Streß und Austrocknung führen.
- Das Forschungsvorhaben sollte die in Schlackedeponien ablaufenden Prozesse aus chemisch-mineralogischer Sicht klären, die Temperaturentwicklung einer typischen Schlackedeponie erfassen und Handlungsempfehlungen für den zukünftigen Betrieb von Schlackedeponien geben.

2 Beschreibung der Deponie

Luftbild der Deponie Großmehring (10.08.1998/98004'1, M   1:5000) mit den einzelnen Bauabschnitten und der Lage der beiden Meßprofile (Profil 1 und Profil 2) im Bereich von Bauabschnitt IV
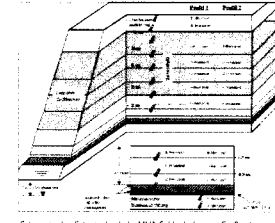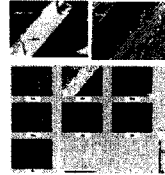
Schematischer Schnitt durch die MVA-Schlackedeponie Großmehring mit der Lage der in Profil 1 und Profil 2 installierten Temperatursensoren

3 Umsetzungsreaktionen und Energiebilanz

Relevant für die Temperaturentwicklung sind:

- Metallkorrosion
- Kalklöschung
- Bildung von CSH-Phasen
- Carbonatisierung
- Inertisierung durch Überzüge

Korrodiertes Metallfragment in glasiger Matrix mit Ca-Hydrat-Kruste

Prozeß	Reaktand	Stoffmenge Reaktand		Produkt	Stoffmenge Produkt		Energie
		kg	Gew.-%		kg	Gew.-%	MJ
Fe-Korrosion	Fe°	18.7	0.85	FeOH₃	35.8	1.6	132
Al-Korrosion	Al°	3.54	0.16	Al(OH)₃	10.23	0.46	28
Kalklöschung	CaO	44	2	Ca(OH)₂	58	2.6	51
Carbonatisierung	Ca(OH)₂	23	1.05	CaCO₃	31.1	1.41	35
CSH-Bildung	Ca(OH)₂	23	1.05	CaH₂SiO₄	41.7	1.9	44
Carbonatisierung	CaH₂SiO₄	166	7.5	CaCO₃	124.5	5.7	31
Sulfidzersetzung	Sulfide	0.1	0.01	Hydroxide	0.24	0.01	1
						Σ =	322

Berechneter Stoffumsatz pro m³ Schlacke und einen Zeitraum von 2 Jahren

4 Temperaturentwicklung
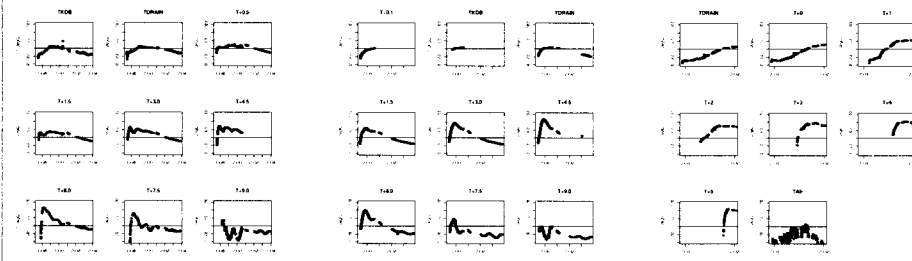
Temperaturentwicklung bei lagenweisem Einbau

Temperaturentwicklung bei massigem Einbau

Temperaturentwicklung bei Verzicht auf Zwischenlagerung

- Temperaturmaxima im Kern der Deponie bis 85 °C
- Temperatur an der Basisdichtung über 40 °C
- Massiger Einbau verursacht schnelleren und höheren Temperaturanstieg
- Verzicht auf Zwischenlagerung verursacht auch bei lagenweisem Einbau sehr hohe Temperaturen

5 Simulation und Prognose

- 1-dimensionale numerische Modellierung der Wärmeausbreitung im Deponiekörper
- Kalibrierung mit Daten von Profil 1, unabhängige Validierung mit Profil 2
- Simulation verschiedener Einbauszenarien (Zwischenlagerung, Schichtdicke, ...)
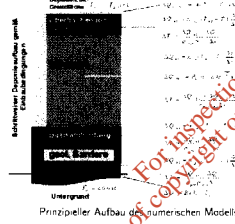
Prinzipieller Aufbau des numerischen Models
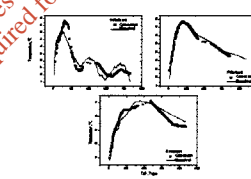
Gemessene und gerechnete Temperaturentwicklung im Profil 1 der Deponie Großmehring (Validierung)
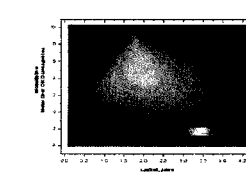
Simulierte Temperaturentwicklung auf einer Schlackedeponie (5wochige Zwischenlagerung, Schichtmächtigkeit 1 m). Geringe Erwärmung der Deponie

6 Handlungsempfehlungen

- Zur Verringerung der Reaktivität der Schlacke sollte diese vor der Ablagerung entschrottet werden.
- Bei Zwischenlagerung ist auf ausreichenden Luftaustausch mit der Atmosphäre zu achten.
- Während der Zwischenlagerung sollte die Schlacke ausreichend bewässert werden, um eine frühe Elution der leicht löslichen Bestandteile zu ermöglichen.
- Zur effektiven Reduzierung der anfänglichen Wärmeproduktion sollte die Zwischenlagerzeit mindestens drei bis vier Wochen betragen.

- Der Einbau auf der Schlackedeponie sollte mit möglichst großem Oberflächen-/Volumenverhältnis durchgeführt werden um eine optimale Energieabfuhr zu ermöglichen. Weitere Schlackenlagen wirken wegen der geringen Wärmeleitfähigkeit als Isolationsschicht.
- Die Aufbringung einer Oberflächenabdichtung sollte nicht zu früh, in keinem Fall aber unmittelbar nach der Ablagerung durchgeführt werden, da dadurch die freie Wärmeabfuhr gehemmt wird und die Inertisierungsprozesse stagnieren.

Numerical modelling of the generation and transport of heat in a bottom ash monofill

R. Klein*, N. Nestle, R. Niessner, T. Baumann

*Institute of Hydrochemistry, Technical University of Munich, Marchioninistrasse 17,
D-81377 Munich, Germany*

Abstract

Municipal solid waste is incinerated to reduce its volume, toxicity and reactivity. Several studies have shown that the resulting bottom ash has a high exothermic capacity. Temperature measurements in municipal solid waste incineration (MSWI) bottom ash landfills have found temperatures up to 90 °C. Such high temperatures may affect the stability of the landfill's flexible polymer membrane liner (FML) and may also lead to an accelerated desiccation of the clay barrier. The purpose of this study was to gain detailed knowledge of temperature development under several disposal conditions in relation to the rate of ash disposal, the variation of layer thickness, and the environmental conditions in a modern landfill. Based on this knowledge, a simulation was developed to predict temperature development. Temperature development was simulated using several storage periods prior to the deposition and several modes of emplacement. Both the storage time and the mode of emplacement have a significant influence on the temperature development at the sensitive base of the landfill. Without a preliminary storage of the fresh quenched bottom ash, high temperatures at the bottom of a landfill cannot be avoided.
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Keywords: Bottom ash; Temperature development; Municipal solid waste incineration; Landfill

1. Introduction

Until the 1970s, bottom ash from municipal solid waste incineration was believed to be almost inert, but since then several studies have shown that many exothermic reactions may cause a temperature increase of up to 90 °C in the landfill [1].

High temperatures at the bottom of a landfill may affect the stability of the landfill liner system (flexible membrane liner, polymer membrane liner (FML) and mineral clay layer).

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E-mail address: alf.klein@ch.tum.de (R. Klein).

Temperatures above 40 °C may damage the stability of the FML (made of high-density-poly-ethylene, HDPE) due to depolymerisation and oxidation [2]. Due to diffusive transport of water and water vapour along the temperature gradient in the mineral clay layer, the clay barrier may desiccate and fail to retain leachate [3,4]. In order to prevent thermal damages to the liner system, it is necessary to minimise the temperatures in the landfill. There are several factors such as the storage time prior to the deposition and the surface-to-volume ratio influencing the temperature development in a landfill [1]. The most important reactions that cause a temperature increase in the stored bottom ash are the corrosion of iron and aluminium, the hydration of lime (CaO) and the carbonation of portlandite ($Ca(OH)_2$) [5–7]. Table 1 shows the identified reactions. Speiser [8] has pointed out that the corrosion of iron is followed by carbonation of portlandite which are the most relevant heat sources in bottom ash material.

Assessing the thermal capacity of the residues is essential since bottom ash has been deposited in landfills with poor landfill liner systems in Europe and in other countries during the last decade [7]. In the US, bottom ash was commonly landfilled without processing, even though metals and other materials can be recovered by magnetic separation and screening [9]. In some European countries (e.g. Germany, The Netherlands and France) approximately 60% of the bottom ash is reused in road construction or as raw material for the ceramic and cement industry [10–12], whereas in Switzerland almost 100% of the bottom ash is disposed in landfills [9].

Although the exothermic reactions in bottom ash are well known, their speed and the amount of heat released are still unknown. Klein et al. [1] have shown that the main temperature increase due to the exothermic reactions has a time scale of 2–3 months. Speiser [8] calculated an average specific heat production of $5.3 \, \text{W m}^{-3}$ of the bottom ash material during the first 2 years of deposition. The released energy in this period amounts to $313–331 \, \text{MJ m}^{-3}$. The bottom ash investigated in this study is comparable to a common bottom ash analysed in the EU [6].

The objective of this work was to develop a numerical model incorporating basic concepts from chemistry and physics to simulate the spatial and temporal distribution of heat in a bottom ash landfill. This objective was accomplished in two steps: (1) the observation of the temperature development in a bottom ash landfill under several modes of emplacement, and (2) the development of a heat generation and transport model and validation of this with the data obtained from field experiments. This numerical simulation provides the possibility of

Table 1
Exothermic reactions in bottom ash materials [5–7]

Reaction	Enthalpy of reactions, ΔH (kJ mol^{-1})
$2Al + 6H_2O \rightleftharpoons 2Al(OH)_3 + H_2 \uparrow$	−422
$FeS + (9/4)O_2 + (5/2)H_2O \rightleftharpoons Fe(OH)_3 + H_2SO_4$	−921
$CaO + H_2O \rightleftharpoons Ca(OH)_2$	−65
$Ca(OH)_2 + H_2CO_3 \rightleftharpoons CaCO_3 + 2H_2O$	−111
$Ca(OH)_2 + CO_2 \rightleftharpoons CaCO_3 + H_2O$	−120
$Ca(OH)_2 + SiO_2 \rightleftharpoons CaH_2SiO_4$	−140
$CaH_2SiO_4 + CO_2 \rightleftharpoons CaCO_3 + SiO_2 + H_2O$	−25

predicting the temperature development in a bottom ash landfill under different modes of emplacement.

2. Experimental

2.1. Field observations

Three vertical sensorfields (SF1, SF2, SF3) were embedded in two bottom ash landfills in the south of Germany. Temperatures were recorded using Pt-100 temperature sensors (R + S Components, Moerfelden, Germany, measurement range from −200 to +300 °C).

The bottom ash in SF1 was deposited in irregular time intervals (see Table 2) depending on the amount bottom ash to be disposed, over an 8-month period to a maximum thickness of ten meters [1]. SF2 was emplaced within 3 weeks to its final height of 10 m. The bottom ash for SF1 and SF2 was stored for 3–6 weeks before being deposited at the landfill. In SF3, bottom ash was emplaced in layers with a thickness of 1 m every 2 months up to a final height of 5 m. The bottom ash in this sensorfield was stored for a maximum duration of 3 days prior to deposition.

2.2. Numerical simulation

The landfill is represented in a computer model as a one-dimensional column, consisting of a geological barrier (GB) underneath the landfill, a liner system (LS), the main bottom ash (BA) body, and (optionally) a surface sealing (SS) (Fig. 1). The individual layers of this linear model used in this work are represented by discrete volume elements with a thickness

Table 2
Bottom ash deposition parameters during the installation of the test field

| Location within the landfill	Date of depositing, corresponding ambient temperature and bottom ash amount		
	SF1	SF2	SF3
At the FML	13 June 1997 (24 °C)	18 May 1999 (21 °C)	6 December 2000 (4 °C)
In the drain	27 June 1997 (22 °C)	18 May 1999 (21 °C)	6 December 2000 (4 °C)
0.5 m above drain	27 June 1997 (22 °C, 600 m^3)	18 May 1999 (21 °C, 300 m^3)	6 December 2000 (4 °C, 1280 m^3)
1.5 m above drain	17 July 1997 (26 °C, 800 m^3)	18 May 1999 (21 °C, 410 m^3)	7 February 2001 (−3 °C, 1500 m3)
3.0 m above drain	17 July 1997 (26 °C, 750 m^3)	18 May 1999 (21 °C, 580 m^3)	11 April 2001 (7 °C, 1620 m3)
4.5 m above drain	27 August 1997 (27 °C, 650 m^3)	18 May 1999 (21 °C, 750 m^3)	3 August 2001 (26 °C, 1800 m3)
6.0 m above drain	24 October 1997 (7 °C, 810 m^3)	18 May 1999 (21 °C, 620 m^3)	
7.5 m above drain	1 November 1997 (15 °C, 720 m^3)	6 June 1999 (23 °C, 580 m^3)	
9.0 m above drain	3 February 1998 (−1 °C, 760 m^3)	6 June 1999 (23 °C, 610 m^3)	
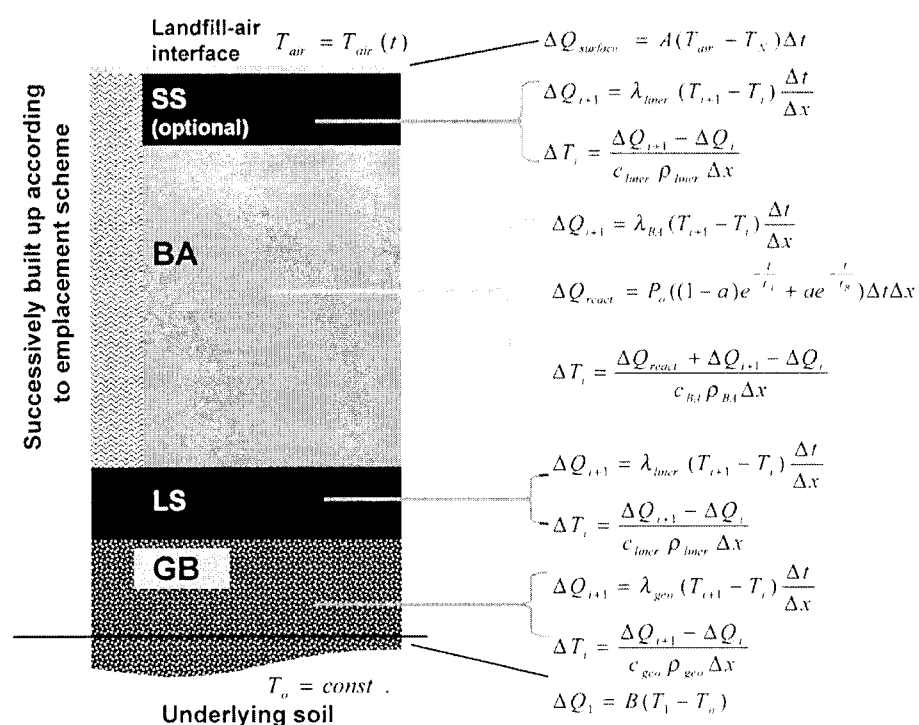
Fig. 1. Schematic structure of the linear column consisting of a geological barrier underneath the landfill (GB), a liner system (LS), the main bottom ash (BA) body as well as (optionally) a surface sealing (SS). The equations on the right side show how the heat balance of the individual layers used in the simulation model. The index 0 indicates the underlying soil, the index n corresponds to the air (i.e. the topmost layer).

of $d = 5$ cm. Heat conduction was computed according to Fourier's law:

$$q_{\text{eff}} = -\lambda_{\text{eff}} \frac{\partial \vartheta}{\partial z} \tag{1}$$

(q_{eff}: effective heat stream, λ_{eff}: effective heat conductivity, $\partial \vartheta / \partial z$: temperature gradient) with a discrete time step of $\Delta t = 30$ min. The heat capacities and thermal conductivities of the different layers in the landfill are given in Table 3. The bottom of the geological barrier was implemented as a fixed head boundary (i.e. a fixed-temperature element with a temperature of $8\,^\circ$C and an infinite heat capacity; experimentally, the natural groundwater temperature was found to vary only in a temperature range between 6 and $10\,^\circ$C). By choosing a sufficiently thick GB layer, influences of the boundary on the model area were kept to a minimum. Heat transfer between bottom ash and either surface sealing or atmospheric air (air temperatures were recorded at the dump location) was approximated by a linear heat transmission. Precipitation, wind and sunshine were known from field measurements to have minor impact on landfill temperature [1]. Vapour and fluid phase convection processes which also appear to have minor influence [1] are not explicitly considered in the model.

Table 3
Initial and boundary conditions for the model of the generation and transport of heat in a bottom ash monofill

Initial and boundary conditions	
Initial heating rate, $P_{(0)}$	Variable
Rate constant of the first exponential, t_A (h^{-1})	0.0006
Rate constant of the second exponential, t_B (h^{-1})	0.00005
Heat transition to the air A	Variable
Heat transition to the soil B	Variable
Fraction of the slow heat generation process, a	0.07
Model height	
Geological barrier	Variable
Liner system	Variable
Bottom ash	Variable
Surface sealing	Variable
Heat conductivity (W m^{-1} K^{-1})	
Bottom ash, λ_{BA}	0.7
Liner material (clay), λ_{liner}	1.3
Geological barrier, λ_{geo}	0.6
Specific heat capacity (kJ kg^{-1} K^{-1})	
Bottom ash, c_{BA}	0.8
Liner system, c_{liner}	1.85
Geological barrier, c_{geo}	0.88
Temperature	
Bottom ash	Variable
Geological barrier	Variable

For the calculations done in the model I, a biexponential decaying heating rate was used. The use of this biexponential decaying heating rate is a somewhat crude approximation for a much more complicated superposition of many endothermic and exothermic reactions with both concentration and transport limitations going on in the bottom ash. For each layer of the bottom ash body, the heat production due to exothermic reactions in the bottom ash is computed with an overall heating rate $P(t)$ given as

$$P(t) = P_{(0)}((1 - a)e^{-t/t_A} + ae^{-t/t_B}) \tag{2}$$

with $P_{(0)}$ representing the initial heating rate of bottom ash, t_A and t_B being the rate constants of the fast and slow reaction processes, respectively, and a being the fraction of the slowly-decaying reaction of the overall heating rate.

The parameters of the biexponential heating rate curve were adjusted by repeatedly running the model with different parameter sets, comparing the model results with the experimental data and choosing new sets of parameters in order to achieve both good correspondence with the experimental data and consistence with the mineralogical observations. As our results show, the parameter set obtained in this process allows a good simulation of the experimentally observed temperature profiles. A possible explanation for two different time scales for the reaction can be the accessibility of reactive material in the bottom ash, which is straightforward on the outside of the bottom ash grains but strongly transport-limited in their cores.

Most parameters of the model were taken from [13–17]. The parameters of the heating rate function were calibrated with field data from SF1.

For all the calculated simulations, the time profile of the air temperature (daily averages) was used as recorded at the landfill site from June 1997 to June 2001. Circadian temperature fluctuations must not necessarily be taken into account for the experimental data since such short-time temperature changes reach only less than 1 m into the landfill body [18,19].

3. Results

3.1. Sensitivity analysis

In order to highlight the significance of chemical, physical and installation parameters controlling heat generation and transport in a bottom ash monofill, a sensitivity analysis was performed. The focus of the analysis was on the parameters that directly affect temperature development in the landfill and in its liner system. Several simulations were performed to assess the model's sensitivity to its chemical, physical and technical parameters. These parameters include the rate of heat release as a result of the exothermic chemical reactions in the bottom ash material, heat transition processes to the bottom and the air, the heat conductivity and the specific heat capacity of the bottom ash and the liner system. To assess the effects of these parameters, one parameter at a time was varied while keeping the others at their basic values. Table 4 summarises the selected sensitivity analysis simulations with the corresponding rationale behind the value chosen for the parameters at each simulation. The simulations performed for this purpose (Fig. 2) lead to the following conclusions:

- The heating rate is the most important factor influencing the temperature increase in the bottom ash landfill, both at the centre as well as at the landfill liner system.
- Heat conductivity of the bottom ash comes next in order of importance.
- At the liner system, heat conductivity of the liner system has a minor influence on temperature development.
- The remaining parameters do not affect the maximum temperature reached in the bottom ash landfill.

Table 4
Summary of the sensitivity analysis simulations

| Variable	Basic values	Sensitivity values (basic value multiplied by the number in parentheses)
Heat conductivity of the bottom ash, λ_{BA} (W m^{-1} K^{-1})	0.7	(0.05, 0.1, 0.2, 0.5)
Heat conductivity of the liner material, λ_{liner} (W m^{-1} K^{-1})	1.3	(0.05, 0.1, 0.2, 0.5)
Specific heat capacity of the bottom ash, c_{BA} (kJ kg^{-1} K^{-1})	0.8	(0.05, 0.1, 0.2, 0.5)
Specific heat capacity of the liner system, c_{liner} (kJ kg^{-1} K^{-1})	1.85	(0.05, 0.1, 0.2, 0.5)
Initial heating rate of the bottom ash, $P_{(0)}$ (W m^{-3})	25	(0.05, 0.1, 0.2, 0.5)
Heat transition to the air A (W m^{-2} K^{-1})	1	(0.05, 0.1, 0.2, 0.5)
Heat transition to the soil B (W m^{-2} K^{-1})	20	(0.05, 0.1, 0.2, 0.5)
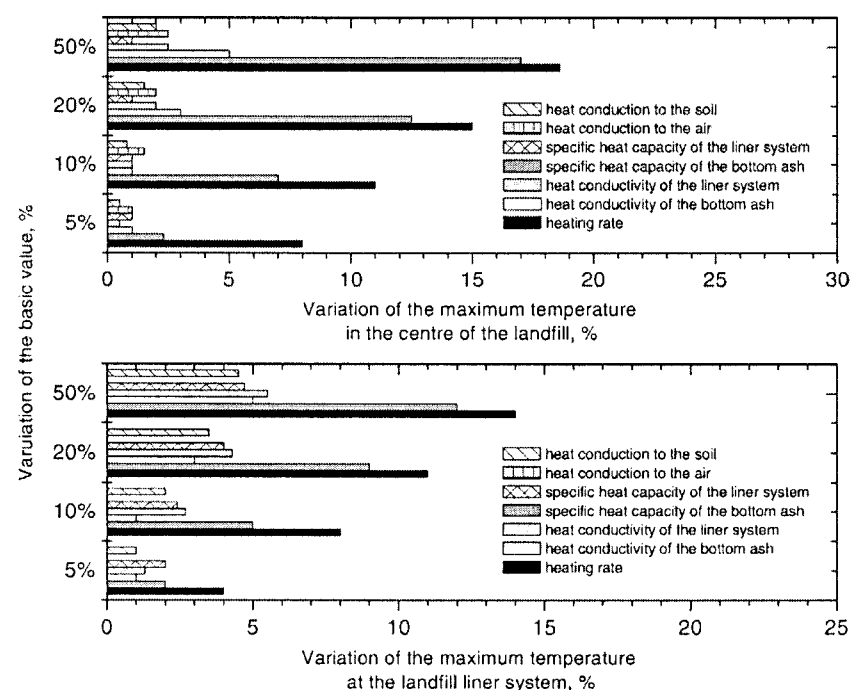
Fig. 2. Effect of variation of basic values on the maximum temperature in the centre of the landfill and at the landfill liner system.

• Heat exchange with the air seems to have no major influence on the temperature development at the landfill liner system.

3.2. Temperature development

Temperature development in selected landfill levels of SF1, SF2 and SF3 is shown in Fig. 3. There was an observed temperature increase immediately after the deposition of a bottom ash layer in each sensorfield. After reaching its maximum 90–160 days after bottom ash deposition, temperature decreased again in all observed landfill layers.

In the following we will present the simulation results for the installed sensorfields and a range of typical emplacement schemes which are summarised in Table 5.

3.3. Calibration and prediction

During model calibration, we have worked out the heating rate of the 3–6-week stored bottom ash material as used in SF1. In order to determine the heating rate of bottom ash when subjected to a previous storage period, the registered temperature development of SF1 was simulated by means of the model. A heating rate upon emplacement of approximately 25 W m^{-3} for the bottom ash material could be determined using the simulation. With
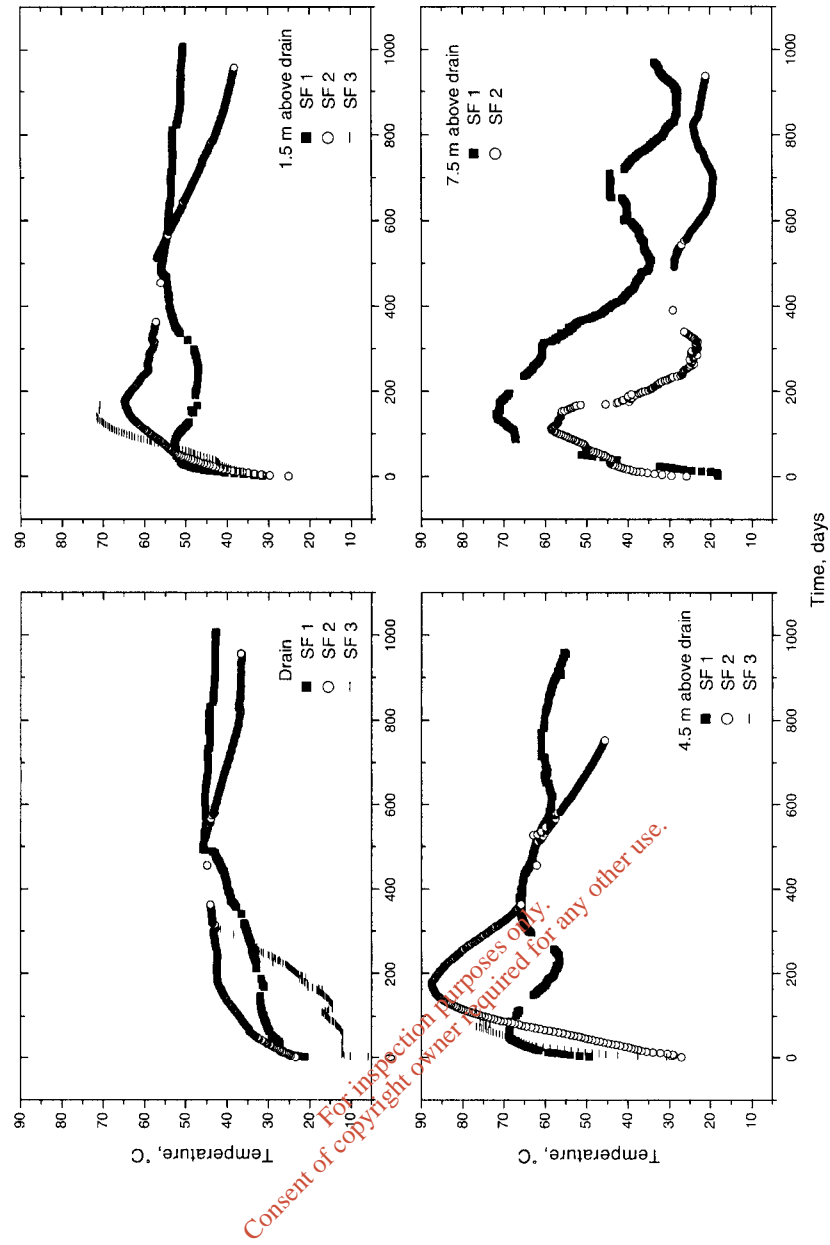
Fig. 3. Measured temperature development in the three sensorfields (SF1–SF3).

Table 5
Deposition procedure for the calculated temperature development in the several model runs of heat generation in a bottom ash landfill

Simulation no.	Emplacement mode	Bottom ash storage time	Heating rate upon emplacement ($W m^{-3}$)
A	Deposition in discrete intervals of 1 m every 2 months	3–6 weeks	25
B	Deposition within 2 weeks to its final height, surface sealing directly after the deposition of bottom ash	3–6 weeks	25
C	Deposition according to SF1, surface sealing after 3 years	3 months	15

the biexponential decrease of the initial heating rate described above, the experimentally observed temperature maximum of 87 °C in the centre of the landfill at SF1 after 4–5 months after deposition could be reproduced in the simulation. The maximum temperature at the landfill base was reached with 46 °C 18 months after the deposition of the first bottom ash layer. Fig. 4 shows the deviations of the calculated temperatures from the real data measured on the landfill site during the first 1000 days. As can be seen from the figure, the model closely describes temperature development in the lower (liner system) and central (4.5 m above liner system) landfill areas. In the upper landfill areas, there is slight deviation from the measured temperatures in the first winter minimum. This affect is possibly due to a variation in the bottom ash quality which is not accounted for in the simulation. There is an overall good correlation between the calculated and measured data ($R^2 = 0.834$, $N = 8443$).

With the initial heating rate of 25 $W m^{-3}$ and the biexponential decay, we have calculated a released energy of 250 $MJ m^{-3}$ for the first 2 years of storage in the landfill. This amount corresponds with the data observed by Speiser [8].

3.4. Validation and prediction (SF2)

After this calibration, the model was validated using the measured temperature data of SF2 (900 days measurements). With the heating rate value upon emplacement of 25 $W m^{-3}$ determined above, there was good agreement between simulated and observed data. Fig. 5 shows the deviations of the calculated temperatures from the real data measured on the landfill site during the first 850 days. With these data, a good correlation between the calculated and measured data ($R^2 = 0.867$, $N = 7521$) was found.

3.5. Validation and prediction (SF3)

In the second validation phase, the initial heating rate of the fresh quenched bottom ash material, as used in SF3 was measured. In order to determine the initial heating rate of the bottom ash, the measured temperature development during the first 6 months of storage in SF3 with its new emplacement mode was simulated by means of the model. An initial heating rate of approximately 45 $W m^{-3}$ for the bottom ash material in the absence of a preliminary storage period could be determined. With the biexponential decrease of the
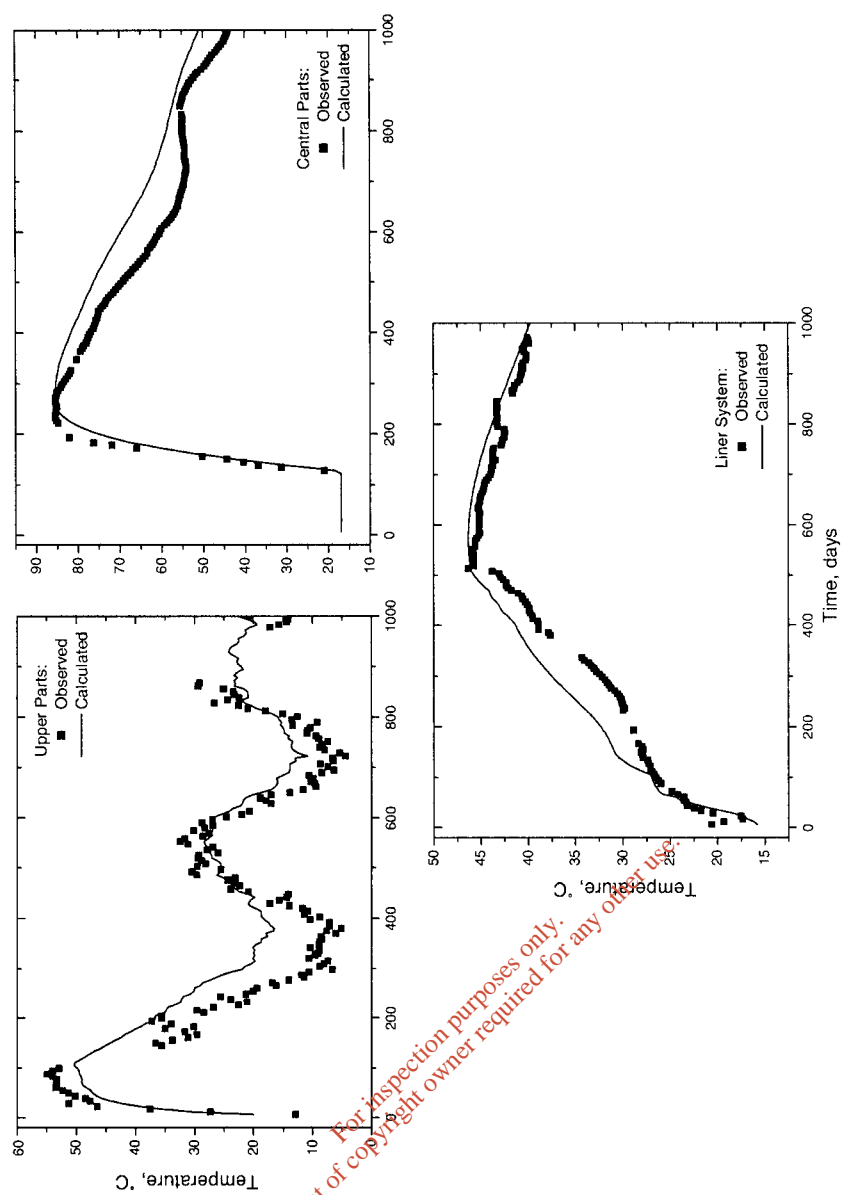
Fig. 4. Comparison of the numeric simulation and at the landfill measured temperatures in selected horizons of the landfill base (liner system), the central area (4.5 m above liner system) as well as the upper landfill area (1 m below surface) for the calibration of the model.
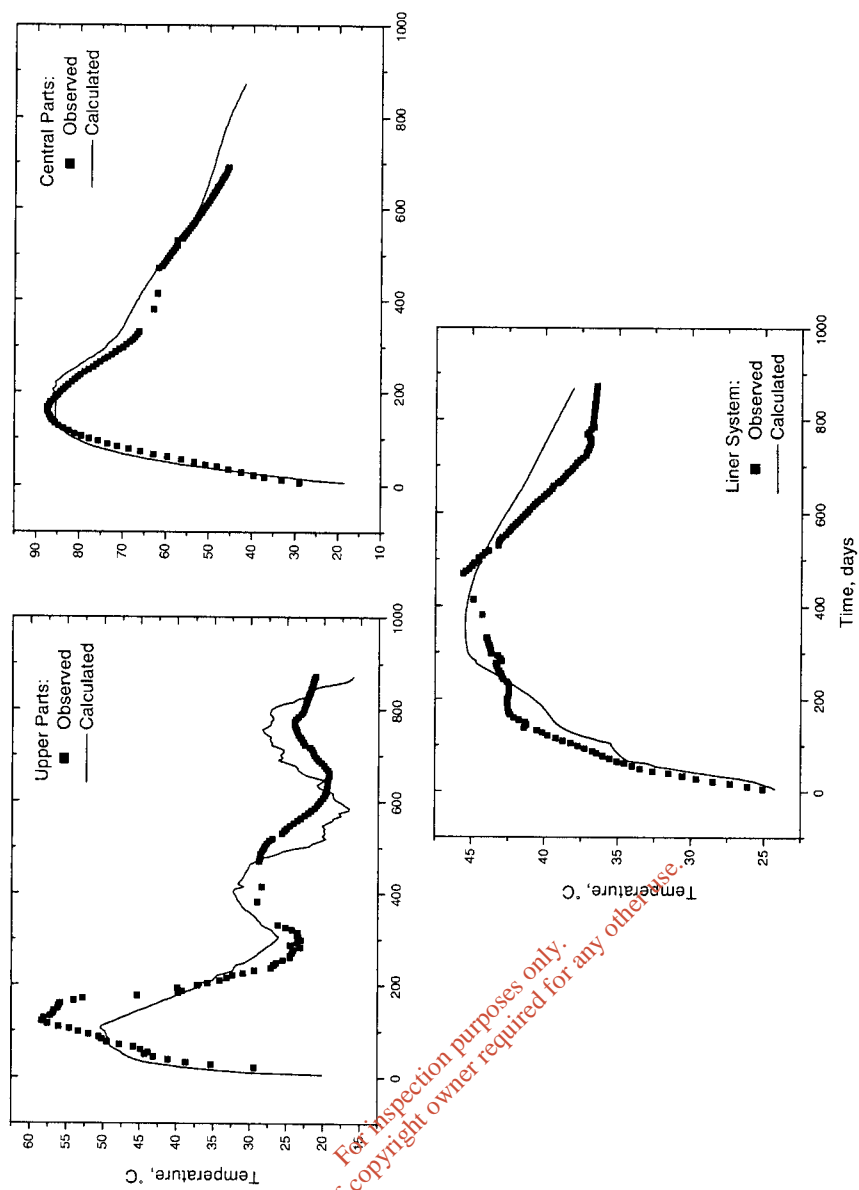
Fig. 5. Comparison of the numeric simulation and at the landfill measured temperatures in selected horizons of the landfill base (liner system), the central area (4.5 m above liner system) as well as the upper landfill area (1 m below surface) for the validation of the model (SF2).
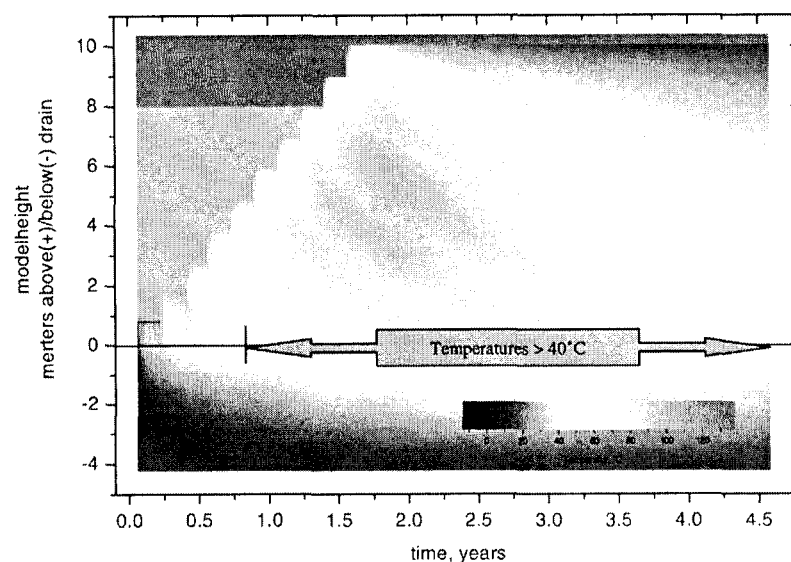
Fig. 6. Predicted temperature development in the second model validation (SF3). Initial heating rate for the fresh quenched bottom ash was set to 45 W m^{-3}, final bottom ash height to 10 m (deposited in discrete intervals of one meter every 2 months).

initial heating rate described above the observed temperature development during the first 6 months could be simulated by the model. The computer simulation results in a temperature maximum of 96 °C in the centre of the landfill (approximately 9 months after the deposition of this bottom ash layer) and 66 °C at its bottom. Fig. 6 shows the calculated temperature development in the landfill over a simulation time of 4.5 years. The high initial heating rate causes higher maximum temperatures in the bottom ash material that result also in higher temperatures in the landfill liner system, and thus may lead to thermal damage of the liner. Temperatures above 40 °C are calculated there from the sixth month after first deposition of bottom ash. Fig. 7 shows the deviations of the calculated temperatures from the real data measured on the landfill site. There is a good correlation between the calculated and measured data ($R^2 = 0.872$, $N = 4287$). With the calibrated and validated model several scenarios were calculated to generate an optimal handling scheme for municipal solid waste incineration (MSWI) bottom ash.

3.6. Simulation no. A: stepwise emplacement of previously stored ash

With the results achieved from the prior simulation, a step-wise emplacement strategy was simulated with bottom ash that was stored for 3–6 weeks before depositing at the landfill with a consequently reduced heating rate from initially 45 to 25 W m^{-3}. This reduced heating rate is also reflected in the temperature development in the landfill body. The maximum temperature reaches only 54 °C in centre and 38 °C at the basis of the landfill (Fig. 8). So there is no temperature above 40 °C at the liner system.
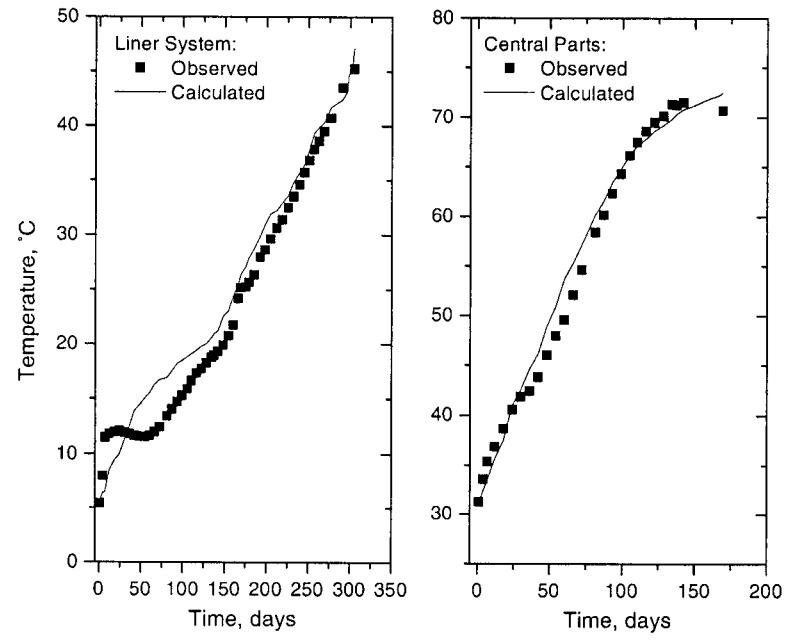
Fig. 7. Comparison of the numeric simulation and at the landfill measured temperatures in selected horizons of the landfill base (liner system) and the central area (3 m above liner system) for the validation of the model (SF3).
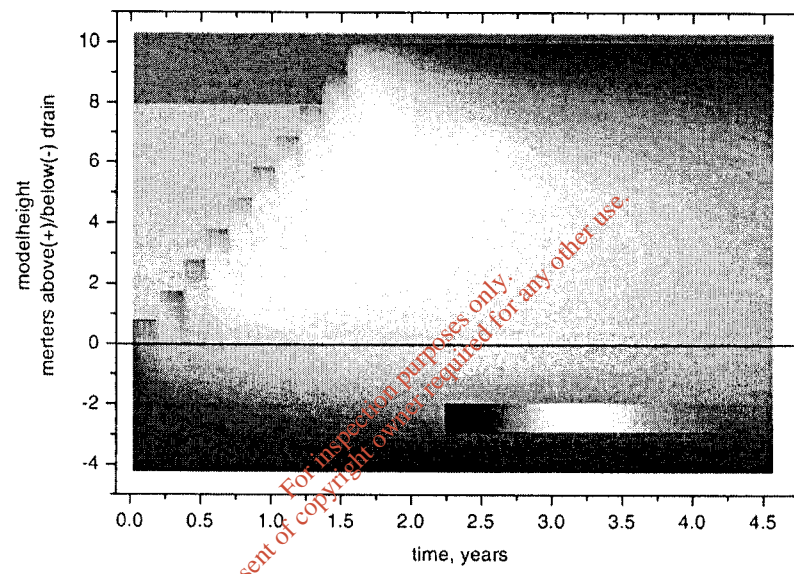

Fig. 8. Predicted temperature development in simulation no. A. Initial heating rate for the 3–6 weeks stored bottom ash was set to 25 W m^{-3}, final bottom ash height to 10 m (deposited in discrete intervals of 1 m every 2 months).
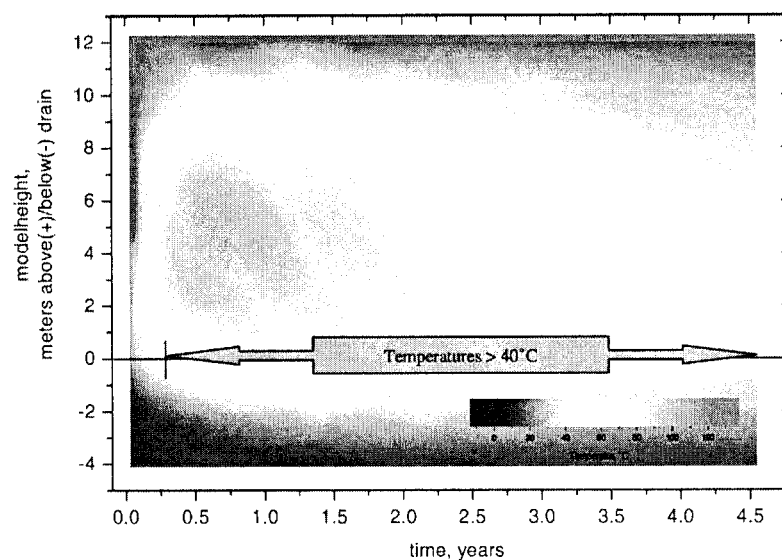
Fig. 9. Predicted temperature development in simulation no. B. Initial heating rate for the 3–6 weeks stored bottom ash was set to 25 W m^{-3}, final bottom ash height to 10 m (deposited in 3 weeks to its final height). Surface sealing was installed directly after the deposition of the bottom ash.

3.7. Simulation no. B: surface sealing

In the next simulation, the influence of a surface sealing on landfill temperature development was modelled. The simulated landfill has a bottom ash height of 10 m with a liner system (0.8 m) at its bottom and a geological barrier with a thickness of 3 m. In the model run, a surface sealing (2.5 m) was emplaced directly after the deposition of the 3–6 weeks stored bottom ash (initial heating rate: 25 W m^{-3}). With this sealing, the heat convection from the surface to the air is hampered. The result from this simulation shows that after a storage time of only 4 months, the temperature at the landfill centre rises to 97 °C (Fig. 9). Also at the liner system the maximum temperature (58 °C after a storage time of 7 months) is far beyond the critical temperature (40 °C) for the landfill liner durability. Here, temperatures above 40 °C are calculated from the third month after first deposition of bottom ash.

3.8. Simulation no. C: storage time

In the last simulation, the influence of the duration of preliminary bottom ash storage period on the landfill temperature was determined. The sensorfield was built-up according to SF1 and the surface sealing was installed after the final deposition of bottom ash. The initial heating rate was set to 15 W m^{-3}. This heating rate corresponds to a intermediate storage time of approximately 3 months. The calculated maximum temperature (56 °C in the centre of the bottom ash body) was obtained 300 days after the beginning of bottom ash
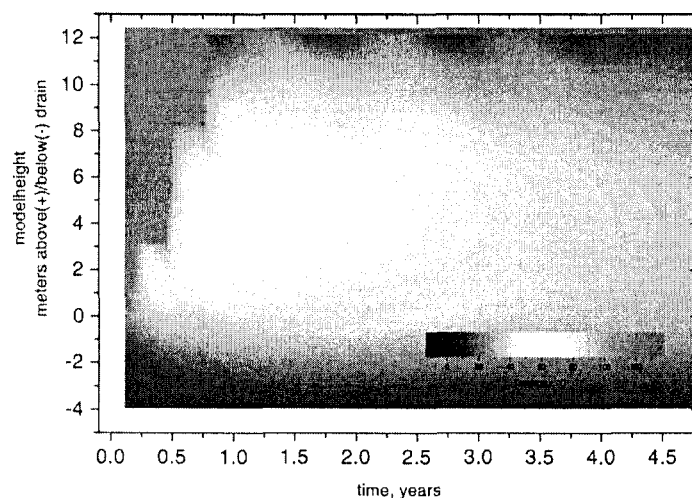
Fig. 10. Predicted temperature development in simulation no. C. Initial heating rate for the 3 months stored bottom ash was set to 15 W m⁻³, final bottom ash height to 10 m (deposited in unequal intervals during a period of 8 months). Surface sealing was installed directly after the deposition of the bottom ash.

deposition (Fig. 10). At the liner system, a maximum temperature of 35 °C was calculated 1 year after the beginning of the bottom ash deposition.

4. Conclusions

In this paper, the temperature development under different modes of bottom ash emplacement was studied. According to the simulation of temperature development in MSWI bottom ash landfills, temperatures from 54 to 97 °C were calculated in the vertical centre of the bottom ash body depending on the emplacement strategy. At the liner system, temperatures reached 35–46 °C. It was shown, that the temperature increases are inversely correlated with the surface-to-volume ratio of the freshly applied ash layer (as realised in simulation B). Furthermore, a preliminary bottom ash storage period prior to disposal is necessary to prevent possible thermal damage at the landfill liner system. The simulation results show that the storage time is the key factor influencing the temperature development in the landfill. A storage time of 3–6 weeks reduces the initial heating rate from 45 to 25 W m⁻³ (reduction of 46%) a 3 months storage time reduces the heating rate to 15 W m⁻³ (reduction of 67%). The risk of a damage at the barrier systems is increased if preliminary storage of bottom ash is not utilised.

Comparatively, it was shown that a storage time of 3–6 weeks and a reduced surface-to-volume ratio lead to maximum temperature values (54 °C in the centre and 38 °C at the liner system) close to those calculated for a storage time of 3 months and a high surface-to-volume ratio (54 °C in the centre and 38 °C at the liner system).

Acknowledgements

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Case study

Temperature development in a modern municipal solid waste incineration (MSWI) bottom ash landfill with regard to sustainable waste management

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Abstract

Municipal solid waste is treated in incineration plants to reduce the volume, the toxicity and the reactivity of the waste. The final product, municipal solid waste incineration (MSWI) bottom ash, was considered as a material with a low reactivity, which can safely be deposited in a MSWI bottom ash landfill, or which can be used, e.g. in road construction after further treatment. However, temperature measurements in MSWI bottom ash landfills showed temperatures up to 90°C, caused by exothermic reactions within the landfill. Such high temperatures may affect the stability of the flexible polymer membrane liner (FML) and may also lead to an accelerated desiccation of the clay barrier. At the beginning of this study it was uncertain whether those reported results would be applicable to modern landfills, because the treatment techniques in MSWI and landfills have changed, bottom and fly ash are stored separately, and the composition of the incinerated waste has changed significantly since the publication of those results.

The aim of this study was to gain detailed knowledge of temperature development under standard disposal conditions in relation to the rate of ash disposal, the variation of layer thickness, and the environmental conditions in a modern landfill.

Temperatures were measured at nine levels within the body of a landfill for a period of nearly 3 years. Within 7 months of the start of the disposal, a temperature increase of up to 70°C within the vertical centre of the disposal was observed. In the upper and central part of the landfill this initial temperature increase was succeeded by a decrease in temperature. The maximum temperature at the time of writing (May 2000) is about 55°C in the central part of the landfill. The maximum temperature (45.9°C) at the FML was reached 17 months after the start of the deposition. Since then the temperatures decreased at a rate of 0.6°C per month.

Temperature variation within each individual layer corresponds to the temperature of the underlying layer and the overall surface-to-volume ratio of the landfill. The temperatures in the uppermost

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layer are significantly influenced by the ambient temperatures. © 2001 Elsevier Science B.V. All rights reserved.

1. Introduction

In OECD countries and the US, 15–20% of municipal solid waste is treated by incineration [1]. Municipal solid waste incineration (MSWI) aims to reduces the volume, the toxicity and the reactivity of the waste. Although the volume of the waste is reduced by about 90%, the residues (bottom ash, fly ash) still amount to roughly 17 Mt per year world-wide [2]. This amount is expected to double within the next 10 or 15 years [3]. Bottom ash, which is the object of this study, represents about 80% of the residues and contains various substances that may pose a threat to groundwater quality [2–4].

Assessing the potential pollution risks of the residues is essential since bottom ash has increasingly been used as building material or has been deposited in landfills with poor landfill liner systems in Europe and in other countries during the last decade [5]. In the US, bottom ash was commonly landfilled without processing, even though metals and other materials can be recovered by magnetic separation and screening [6]. In some European countries (e.g. Germany, The Netherlands and France) bottom ash is partly reused (about 60%) in road construction or as raw material for the ceramic and cement industry [7–9], whereas in Switzerland almost 100% of the bottom ash is disposed in landfills [6].

Until the 1970s, bottom ash was believed to be almost inert, but since then several studies have shown that a number of exothermic reactions occur in this material [10–15]. Other studies have shown that exothermic reactions may cause a temperature increase in the landfill of up to 90°C [16,17] which may constitute a major hazard to the flexible polymer membrane liner (FML) and the mineral clay layer. Temperatures above 40°C may affect the stability of the FML (made of high-density-polyethylene (HDPE)) due to depolymerisation and oxidation. Sudden ruptures of the FML may follow [18]. Due to a diffusive transport of water and water vapour along the temperature gradient in the mineral clay layer, the clay barrier may desiccate and fail to retain leachate [19–21]. Johnson et al. [22] observed a rapid increase in bottom ash landfill discharge following rainfall. Within 1–4 days, approximately 50% of precipitation discharged in response to a rain event.

Due to their limited time scale, published studies on exothermic reactions [23–26] have to be considered as a 'snapshot', hence giving no information on the long-term development of the landfill temperatures. Moreover, many of the basic conditions have changed since then. The incineration technique has been improved and the composition of the municipal waste has changed. For instance, the heating value of domestic waste increased from 6000 to 8000 kJ/kg over the last two decades caused by recycling activities and an augmented share of plastic contents in domestic waste [27]. In contrast to former landfills, fly ashes nowadays are stored in underground repositories, and ferromagnetic scrap metal of a diameter >16 mm is usually separated out by a magnetic separator. With these changes the mineralogical and chemical composition of the deposited residue has changed as well, thus putting the extrapolation of published results to state-of-the-art landfills under question.

The present study aims to provide data on the long-term development of the temperatures within a recent bottom ash landfill under normal disposal conditions.

2. Experimental

2.1. Bottom ash description

The bottom ash in this study was produced by MSWI in Ingolstadt in the south of Germany (MVA Ingolstadt/Germany). The incinerator (installation year 1996) operates at temperatures between 850 and 1200°C. The incineration capacity of each furnace is roughly 11 Mg/h and the material remains in the combustion chamber for about 1 h. Following incineration, the bottom ash is quenched in a water basin. After this quenching process, the bottom ash is temporarily stored in piles up to 2 m in height at an open dump site for 1–3 weeks, in order to reduce the reactivity [28]. Prior to deposition in the landfill, magnetic materials are removed. The grain size distribution of the bottom ash (Fig. 1), determined according to DIN 18123 [29], shows a badly sorted material with grain sizes from silt to gravel.

The determined bulk density has a mean value of 2.13 ± 0.15 Mg/m^3. The geotechnical water content (weight of water in a sample relative to the oven dry weight of the sample, expressed as percentage, DIN 18121 [30]), measured after a 3 weeks storage period, ranges from 8 to 15% by weight.

Although the bottom ash studied is a very inhomogeneous material, it is in general comparable with other MSWI bottom ashes investigated elsewhere [12,31] although there
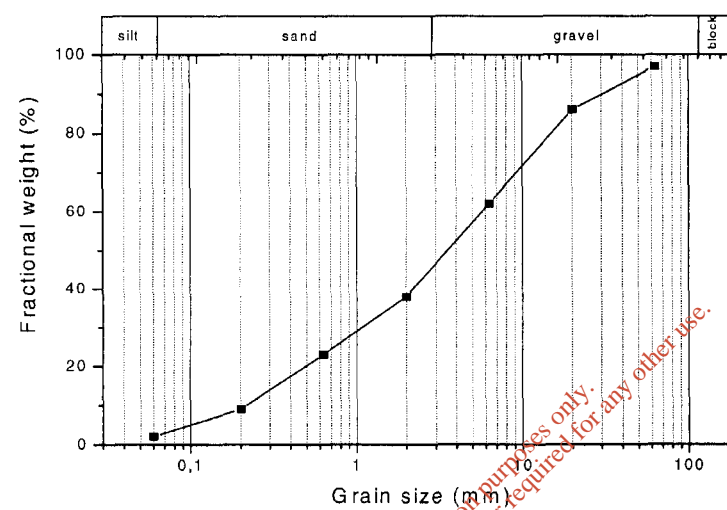

Fig. 1. Grain size distribution of the examined MSWI bottom ash as a function of fractional weight.

Table 1
Bottom ash composition (wt.%)

	Melting products and ashes	Metals	Ceramic	Stones	Glass	Organic waste
This study	82	8	2	1	6	1
Lichtensteiger (1996)	85	5	2	1	5	2
Reichelt (1996)	67	4	4	–	17	–

is a significant variation in the fraction of glass in the bottom ash, caused by increased recycling in municipal solid waste (Table 1).

The thermal conductivity of the investigated bottom ash ranges from 0.23 (dry) to 1.27 W/m K (saturated). It was determined with the thermal conductivity instrument TK04 (TeKa, Berlin/Germany). The samples were taken prior to deposition. The value for the deposited bottom ash at a water content between 10 and 20% by weight ranged between 0.5 and 0.6 W/m K.

2.1.1. Disposal site

The bottom ash landfill investigated in this study is located near Ingolstadt. The measured average ambient temperature in this area is 15°C, with a recorded maximum and minimum of 33 and −8°C during the observation period (June 1997–June 2000). The measured annual precipitation in this period was between 800 and 1000 mm with a maximum between May and July. The driest period was January–April. The summer rains tend to occur in short events with a high intensity.

The geology at the landfill location comprises fluvial and alluvial sediments. The elevation of the water table is approximately 2 m below the base of the landfill. The groundwater flows south towards the river Danube, which flows in an easterly direction approximately 800 m south of the landfill.

The landfill was constructed above ground adjacent to a hill side. The base of the landfill is a 0.6 m thick mineral clay layer, covered by a 2.5 mm FML made of HDPE. Between the FML and the bottom ash is a gravel drainage layer (16–32 mm grain size). The leachate is transported to a communal waste water treatment plant. Two geotextiles separate the bottom ash from the drainage layer and the drainage layer from the FML. A schematic of the test site is given in Figs. 2 and 3. The levelled ground directly below the clay liner consists of sand and gravel. Therefore the capillary rise of water from the ground water into the mineral clay layer may be hampered, leading to a forced desiccation.

Approximately 19,000 m^3 of bottom ash are deposited in the landfill per year at discrete and irregular intervals. The landfill is subdivided into four separated disposal sectors (Fig. 3) [32]. Sectors I–III were already completely filled at the start of the study. Sector IV was filled with bottom ash during the study period. The MSWI fly ash is stored elsewhere in a hazardous waste disposal site. Sector IV, where the sensors are located, has a filled surface area of 16,500 m^2 and a total bottom ash capacity of approximately 100,000 m^3. The sensors are located in the centre of sector IV, so no influence from the other sectors is to be expected. The surface of sector IV has not yet been covered or cultivated, so there is direct contact between the deposited bottom ash and the atmosphere.
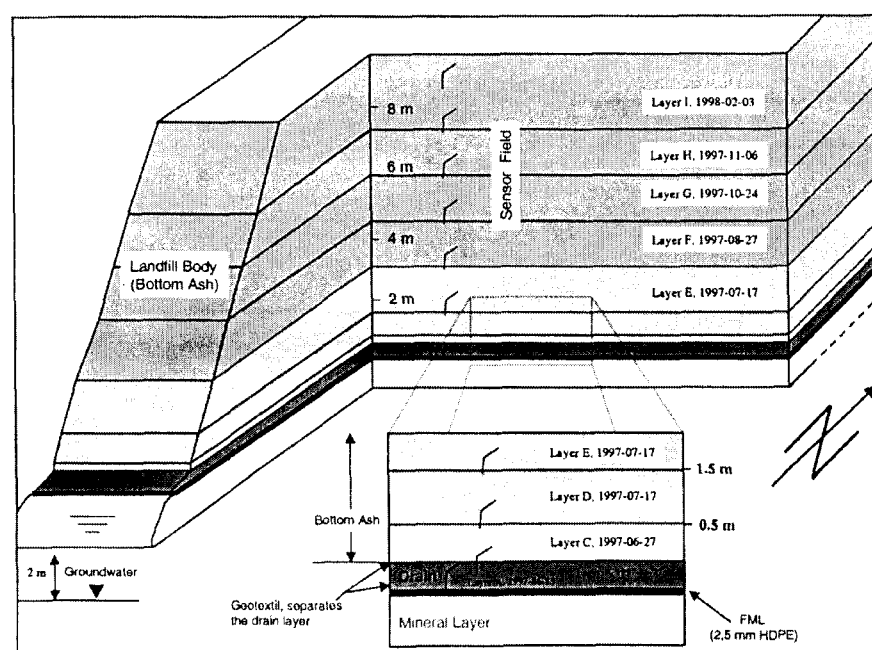
Fig. 2. Schematic cross section through the bottom ash landfill in Ingolstadt (Germany) showing locations of the temperature sensors installed within discrete layers (A–I).

2.1.2. Materials

Temperatures were recorded using Pt-100 temperature sensors (R + S Components, Moerfelden, Germany, measurement range from −200 to +300°C with an error of 0.3%) embedded directly into the bottom ash. The sensors were installed at the top of each layer before the deposition of a new layer (except of sensors in layer I which was placed in the middle of the layer, 9 m above drain, see Table 2, Fig. 2), thus reflecting the temperature development under ordinary disposal management conditions. Each of the nine discrete layers was equipped with two sensors, placed at a horizontal spacing of approximately 1 m.

The bottom ash was deposited in irregular time intervals (depending on bottom ash amount in the MSWI). The ash remained piled for 1–3 weeks on the landfill before it was levelled flat to 150 cm thick layers by dredging. The bottom ash piles were located in the eastern part of sector IV and in sector III. Bottom ash was not compacted and no temporary liner was used to cover the landfill between deposits. There has been no other activity in the test field area during the measurement period.

Data were recorded using a DL2e data logger (Delta-T-Devices, Cambridge, UK) at intervals of maximum 24 h. Additionally, in order to detect any temperature fluctuations, data were recorded at intervals of 1 h from 6 April to 13 April 2000. The following climatic
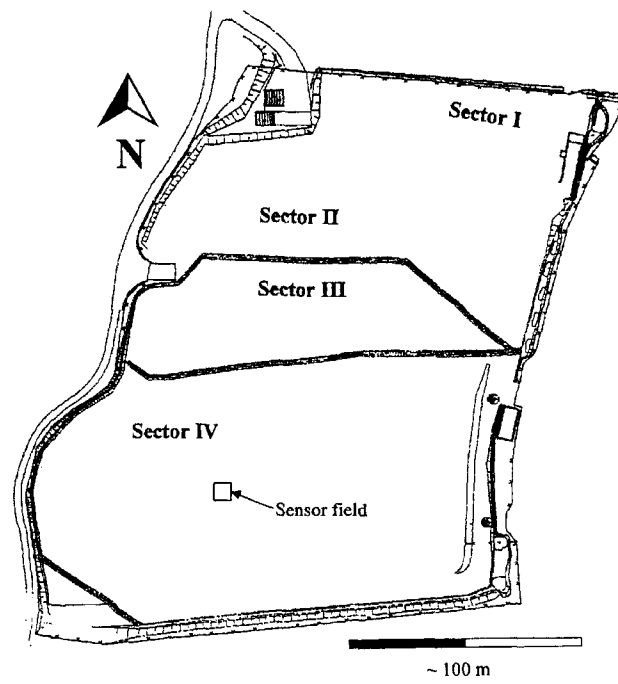
Fig. 3. Schematic section of the bottom ash landfill in Ingolstadt (Germany) showing locations of the temperature sensor field and the four landfill sectors.

Table 2
Bottom ash depositing parameters during the installation of the test field and the corresponding temperature gradients during the first 50 days of depositing

Layer	Localization within the landfill	Date of depositing	Ambient temperature ($^\circ$C)	Temperature of the underlying layer ($^\circ$C)	Average temperature gradient ($^\circ$C per day)
A	at the FML	13 June 1997	24	8.5	0.12
B	in the drain	27 June 1997	22	17.5	0.16
C	0.5 m above drain	27 June 1997	22	21.2	0.23
D	1.5 m above drain	17 July 1997	26	32.5	0.4
E	3.0 m above drain	17 July 1997	26	36.4	0.4
F	4.5 m above drain	27 August 1997	27	51.8	0.71
G	6.0 m above drain	24 October 1997	7	68.7	1.02
H	7.5 m above drain	1 November 1997	15	69.1	0.99
I	9.0 m above drain	3 February 1998	−1	67.5	Climatic changes

parameters were recorded daily using equipment provided by Delta-T-Devices (Cambridge, UK): Air temperature, air humidity, solar radiation, and rainfall. Data are available over a time period of 36 months from June 1997 to June 2000.

2.1.3. Heat transport

Heat is transported in the bottom ash landfill mainly by two ways. First, there is a conductive heat transport from one layer to each other. The second way is a convection heat transport from the bottom ash to the atmosphere.

The conductive heat transport j can be calculated with the thermal conductivity of the bottom ash λ and the temperature difference between two landfill layers $(T_2 - T_1)$

$$j = \lambda(T_2 - T_1) \tag{1}$$

The convection heat transport from the bottom ash to the atmosphere Φ is defined as the product of the temperature difference from the bottom ash to the atmosphere $(T_S - T_L)$, the surface A, the time period Δt and the thermal coefficient α_C (6.2 W/m² K for the bottom ash surface)

$$\Phi = \alpha_C A(T_S - T_L)\Delta t \tag{2}$$

## 3. Results

3.1. Temperature development

The development of the temperatures (daily mean) in the different layers of the field site is given in Fig. 4. The mean temperature difference between the two sensors in each layer was between 0.1 and 0.5°C with an average of 0.24°C.

In every layer the temperature development started with an increase immediately after deposition. During the next 2.8 ± 0.3 months, the bottom ash temperatures increased by about 75°C, depending on the layer position. The average rate at which the temperatures rose was between 0.16 and 1.02°C per day (Table 2).

In layers A and B (FML and drain) the initial temperature rise (0.14°C per day in layer A and 0.16°C per day in layer B during the first 4 weeks) was followed by a levelling off for the next 2 months. Afterwards a second increase of temperatures, now at a rate of 0.065 ± 0.005°C per day was observed. The maximum temperature (45.9°C in layers A and B) was reached 17 months after the deposition of these layers. Subsequently, the temperatures in layers A and B decreased at a rate of 0.6°C per month (layer A), respectively 0.54°C per month (layer B). The temperature increase in these two layers is a result of the temperature increase in the bottom ash layers deposited above them and the heat flux from these layers. The gravel in the drainage (layer B) and the FML (layer A) do not generate their own heat.

Layer C (the lowest bottom ash layer) showed an initial temperature increase of up to 44°C (at a rate of 0.25°C per day) during the first 2 months of storage. The temperature increase showed a first levelling off after a storage time of 18 days. After depositing layer D, layer C showed a renewed small rise in the gradient of temperature increase. This
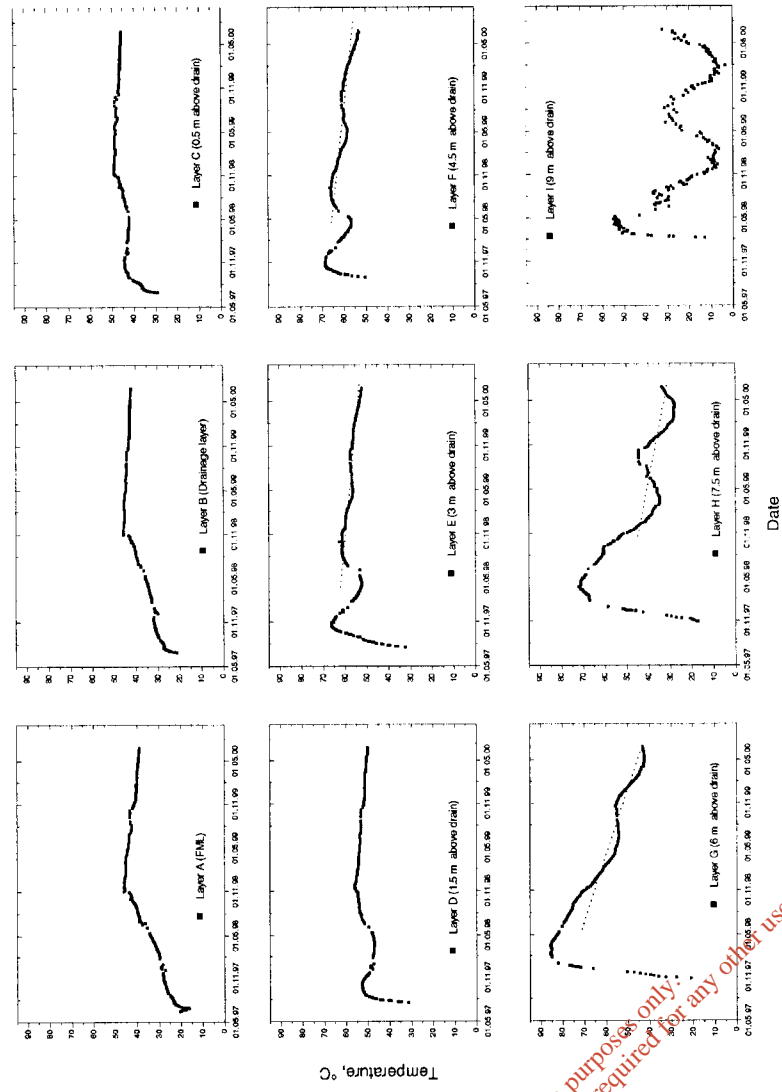
Fig. 4. Recorded temperatures in the various landfill layers. The different factors, explaining the variation of the temperatures are given in the text.

increase was followed by a 6 month temperature decrease (0.36°C per month). With a second temperature increase, this layer reached its maximum after 14 months of storage time (49°C for layer C). From that time temperatures decreased at an overall rate of 0.3°C per month.

Layer D showed a similar temperature development with an initial temperature increase of 0.35°C per day. It reached its maximum temperature after 14 months of storage time (56°C) and decreased then with an rate of 0.3°C per month.

In layers E–G, the temperature development after the initial increase (with its maximum at 87°C in layer G) shows an oscillation with a period of approximately 12 months. The monthly average temperatures (dotted line in Fig. 4) decline at a rate of 0.3°C per month in layers E and F and 0.9°C per month in layer G.

Layer H shows a similar temperature development. After a storage time of 80 days, the temperature increase in layer H levelled off. By depositing layer I, the temperature in layer H rose again for the next 50 days and reached its maximum with 72.2°C. The trend in this layer indicates a decline of temperatures at the rate of 0.6°C per month.

At the top of the landfill, layer I, the initial increase was followed by a rapid decrease and a following oscillation with a period of 12 months. The minimum temperatures were reached during winter, the maximum temperatures during summer. The temperature curve also shows an oscillation with a shorter period (24 h) reflecting the daily ambient temperature fluctuation (Fig. 5).
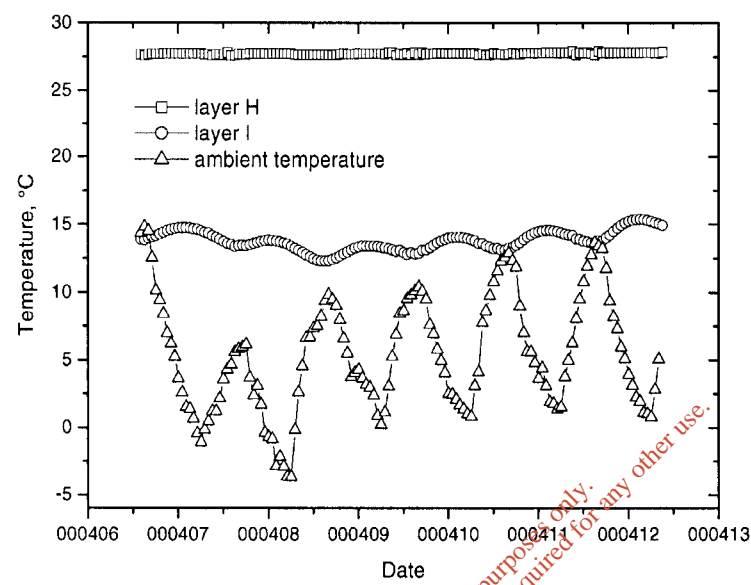

Fig. 5. Influence of measured daily temperature fluctuations (recorded for 1 week at intervals of 1 h) on selected bottom ash layers.

Three years after deposition, temperature development in the upper layers shows an overall decrease with a seasonal component. The lower layers in the lower landfill follow this overall trend, but they do not show the seasonal influence.

4. Analysis

There are several factors which are suspected to influence temperature development. A simplified description of the temperature change (ΔT) within a representative elemental volume (REV) leads to Eq. (1) as the sum of heat production (E_{exo}) due to exothermic reactions minus the heat consumption from endothermic reactions (E_{end}) plus external input (F_{in}) minus heat loss (F_{out}).

$$\Delta T = E_{exo} - E_{end} + F_{in} - F_{out} \tag{3}$$

Within this equation, the amount of exothermic and endothermic reactions is unknown. The heat exchange to and from the REV is a function of the temperature gradient, the thermal conductivity and the convection heat transfer between the REV and its environmental (e.g. other bottom ash REV, drain, atmosphere). On the field scale, each layer is considered as a REV.

The key factors influencing the temperature development thus can be defined as

1. the temperature gradient to the underlying layer or, if there is no underlying layer, the ground of the landfill,
2. the temperature gradient to the ambient temperature or, if another layer is on top of the REV, the temperature gradient to the upper layer,
3. the thermal conductivity between the REV and its environment,
4. the convection heat transfer from the bottom ash to the atmosphere,
5. the ratio between heat production and the heat flux at the boundaries of the REV, which is expected to be a function of the surface-to-volume ratio of the REV,
6. the effect of the precipitation as transport and reaction medium.

In the following section, the effects of these factors will be assessed semi-quantitatively based on the measurements of temperature development.

4.1. Temperature at the bottom of each layer

There is a positive correlation ($R^2 = 0.983$, $N = 6$) between the temperature gradient from the next deposited bottom ash layer to the underlying layer (at the time of depositing the next layer) and the rate of temperature increase in the newly deposited layer (Fig. 6). This effect is based on an addition of the internal generation of heat in each bottom ash layer (layers A and B do not generate their own heat) and the heat conduction from the underlying layer.

The highest rate of increase (temperature increase per day, see Table 2) was observed in layer G, where the temperature of the underlying layer (layer F) had reached a temperature of almost 69°C when layer G was deposited. The lowest rate was observed in layer C, where
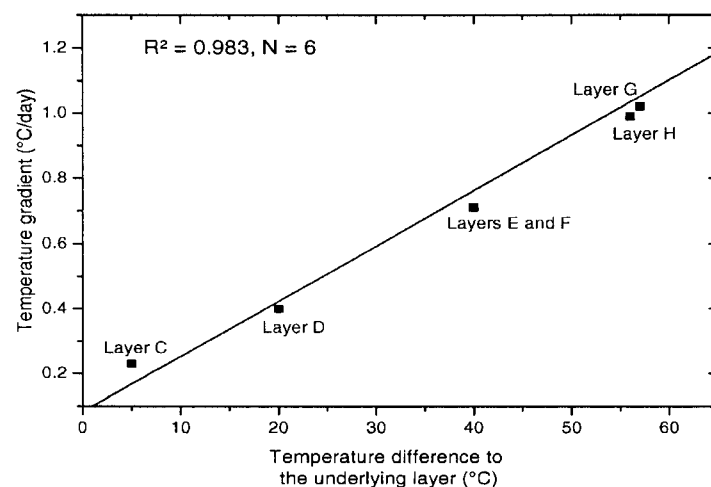
Fig. 6. Calculated gradient of temperature increase of the different layers vs. the temperature of the underlying layer in time of depositing the next one (shown is the regression line).

the underlying layer, which does not generate heat at all, had a temperature of only 21°C (see Table 2).

4.2. Ambient temperatures

There is a statistically significant correlation ($R^2 = 0.788$, $N = 522$) between the temperatures in the top layer (layer I) and the ambient temperature (Fig. 7). This effect is observed to be less pronounced with increasing depth in the landfill. Layers E to H show an oscillation in bottom ash temperature after having reached their maximum temperatures. This oscillation has a period of approximately 12 months and reflects the annual ambient temperature development with a delay of 28 days for layer H, 58 days for layer G, 82 days for layer F and 112 days for layer E. This growing delay reflects the thermal buffer capacity of the bottom ash.

4.3. Surface-to-volume ratio

Heat flux (Φ) from the bottom ash towards the cooler air is an important factor influencing the thermal development in the landfill.

With an upwards conductive heat transport in layer I of 2–35 W/m^2 (with an average of 15 W/m^2) and an average convection heat transport of 70–250 W/m^2 (with an average of 105 W/m^2) from the heated bottom ash of layer I to the air during the first 200 days of deposition, the addition of each new layer hampers the heat exchange between the bottom ash and the atmosphere.
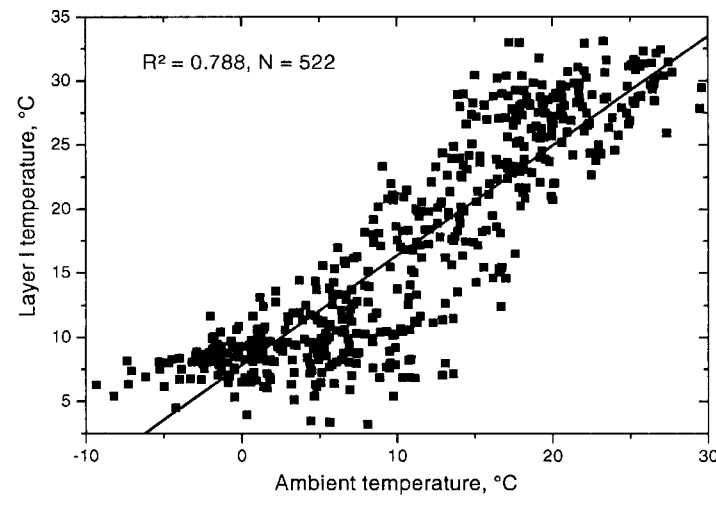
Fig. 7. Recorded ambient temperature plotted vs. recorded temperature in layer I (shown is the regression line).

There is a correlation ($R^2 = 0.987$, $N = 4$) between the surface-to-volume ratio (s/v) and the maximum temperature in the observed volume. The maximum temperature increases with decreasing s/v (Fig. 8) from 50°C (layer C) to 87°C (layer G) (see Table 2).
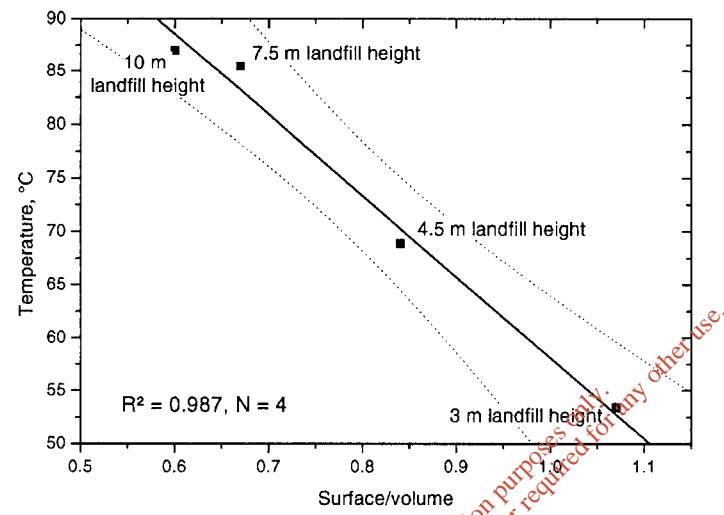

Fig. 8. Calculated surface-to-volume ratio of the growing landfill vs. the maximum temperatures in the middle of each volume at the given landfill height (shown is the regression line).
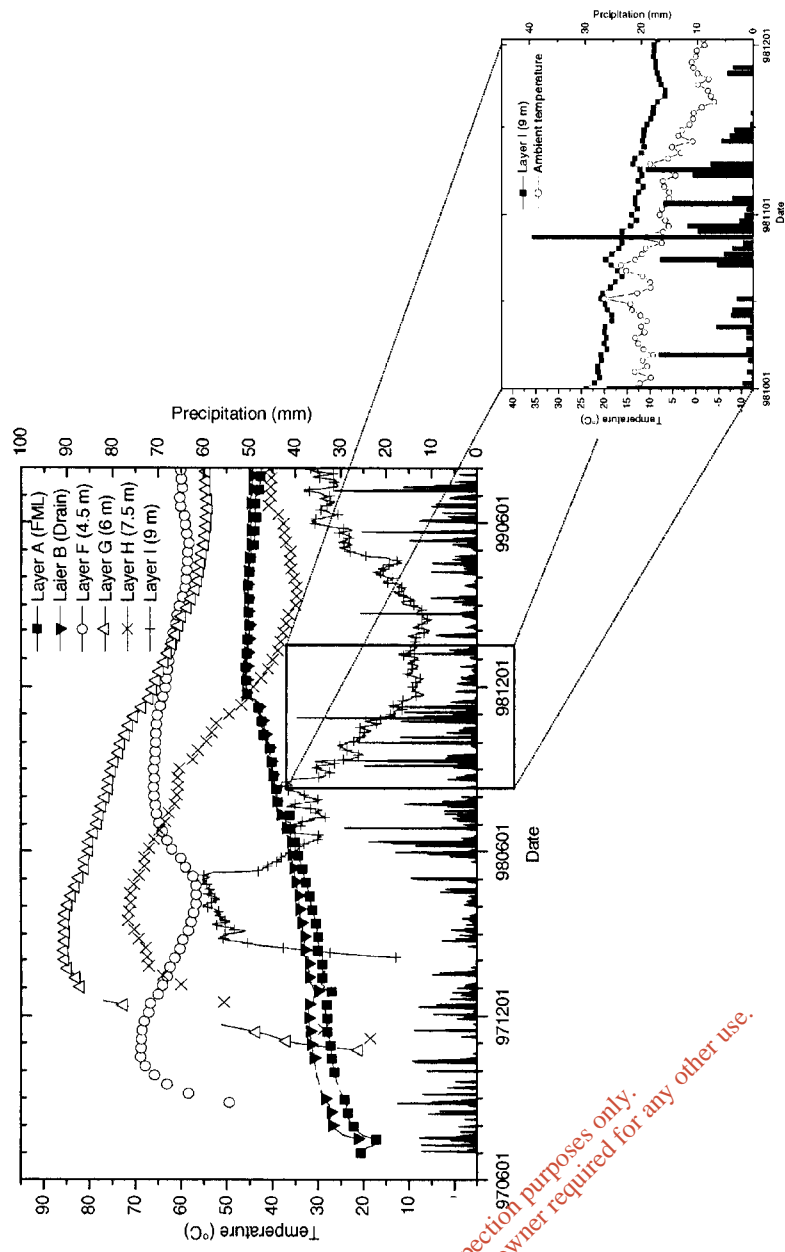
Fig. 9. Temporal change of bottom ash temperature in selected layers and precipitation intensity.

4.4. Precipitation

Rainwater seeping through the landfill body influences the temperature in two ways. First, it is a transport medium and contributes to the heat exchange. Second, it is a reaction medium and contributes to the heat production.

Although we observed that rainfall passes through the landfill within days (there is a direct discharge responding to rain events), precipitation seeping through the landfill body was not observed to have a significant effect on temperatures in the bottom ash (Fig. 9).

Seeping water passing the landfill showed a temperature increase regardless of the intensity of the rainfall of approximately 11.5°C. This is equivalent to an heat extraction of only 0.1 W/m^3 bottom ash from the landfill.

Even after an intensive period of rain (e.g. 85 mm within 6 days, 25 October 1998 until 11 November 1998) there was no observable influence on temperature development in the landfill body and on the temperature of the leachate. The temperature decrease in layer 1 during this rain period is mainly caused by ambient temperature fluctuations (Fig. 9). A dry period in spring (26 March 1999 until 30 May 1999, 120 mm within 70 days) also appears to have caused no change in the temperature development. Precipitating waters seeping through the landfill body, exhibited only a negligible cooling effect.

5. Conclusions

The monitoring of the temperatures in a MSWI bottom ash landfill over a 3-year-period showed a maximum temperature of 87°C 3 months after disposal followed by a decrease over the next 33 months. Temperatures at the FML reached a maximum of 45.9°C after 17 months. Subsequently, the temperature decreased at a rate of 0.6°C per month. We estimate that the temperature in this layer will stay in the critical region above 40°C (depolymerisation and oxidation in the FML, desiccation of the mineral clay layer) for the next year. These temperatures may jeopardise the integrity of the liner through depolymerisation of the HDPE and desiccation of the clay layer, resulting in leachate escaping into the groundwater.

From the temperature development, it can be seen that the main temperature increase due to the exothermic reactions have a time scale of 2–3 months, after which the reaction activity decreases. This suggests that the bottom ash should be stored in thin layers or small cones (which have a favourable s/v ratio) for at least 3 months prior to the final disposal.

The disposal should be given a significant amount of time to react before the next layer is deposited, since the temperature of the underlying layer controls the initial temperature development of the actual layer. From our investigations, it can be concluded that the disposal of the next layer should not start before the maximum temperatures of the underlying layer have been reached and the temperatures and the heat production in the underlying layer are decreasing again significantly. At the present stage of the experiments, we estimate that the time before depositing a new layer should be approximately 3–5 months.

If that time lag in the filling procedure is not possible, other cooling measures (e.g. reinjection of landfill leachate) have to be brought forward, since the precipitation shows a negligible cooling effect. In any case, if a sustainable liner system imperviousness has to be

guaranteed, the capping and recultivation of the landfill, which will hamper any heat, gas, water or vapour exchange between bottom ash and atmosphere should be done only after the reactions within the landfill have reached a minimum and no further temperature rise is to be expected (at least 1 year after the final deposition of the bottom ash). A premature recultivation may lead to an additional temperature increase within the landfill body unless the exothermic reactions have decreased significantly.

Acknowledgements

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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

o   guarantee an excellent burnout of carbon compounds,
o   promote the volatilisation of heavy metals like Hg and Cd out of the fuel bed, and
o   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].

1

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.
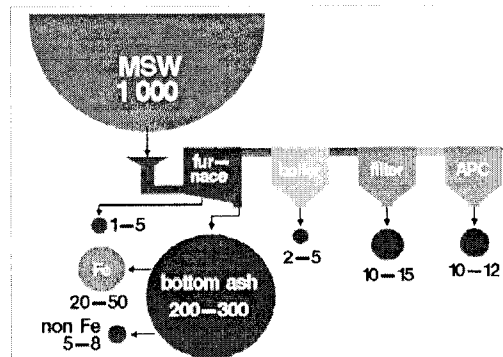

Fig. 1   Mass balance of a municipal solid waste incinerator (values in kg/Mg waste)

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/m³ [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.

Table 1   Selected German standards for disposal and utilisation of MSWI bottom ash

	unit	landfill class 1	road construction
LOI	wt-%	3	
TOC	wt-%	1	1
DEV S4			
soluble fraction	wt-%		
el. conductivity	mS/m	1000	600
Cl	mg/l		250
Cu	mg/l	1	0.3
Zn	mg/l	2	0.3
Cd	mg/l	0.05	0.005
Pb	mg/l	0.2	0.05

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

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.
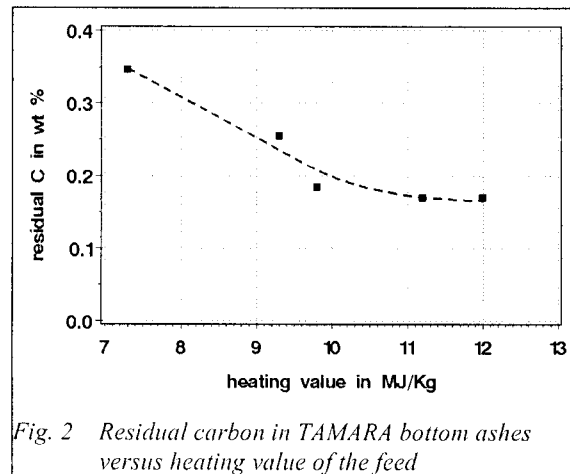

Fig. 2   Residual carbon in TAMARA bottom ashes versus heating value of the feed

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.

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

parameter	bottom ash ng/g	boiler ash ng/g	filter ash ng/g
I-TEQ	<0,001 - 0,01	0,02 - 0.5	0,2 - 10
PCB	<5 - 50	4 - 50	10 - 250
PCBz	< 2 - 20	200 - 1 000	100 - 4 000
PCPh	<2 - 50	20 - 500	50 - 10 000
PAH	<5 - 10	10 - 300	50 - 2 000

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
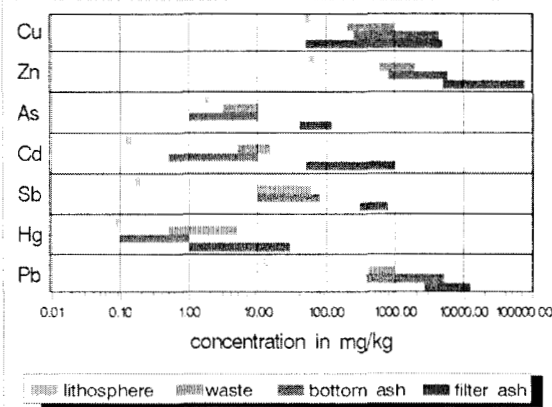
Fig. 3 Concentration ranges of selected elements in various materials

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].
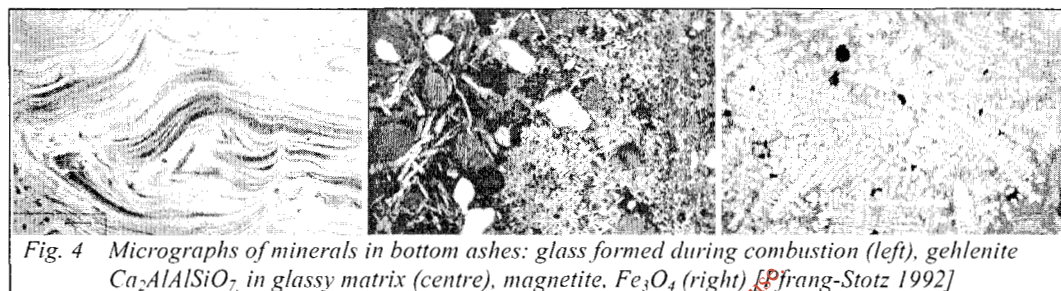

Fig. 4 Micrographs of minerals in bottom ashes: glass formed during combustion (left), gehlenite
$Ca_2AlAlSiO_7$ in glassy matrix (centre), magnetite, Fe_3O_4 (right) [Pfrang-Stotz 1992]

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:

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

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.
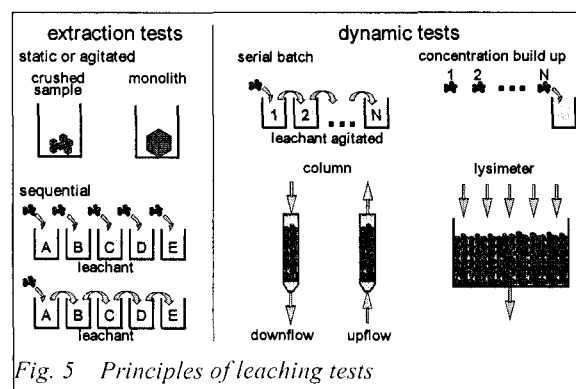

Fig. 5   Principles of leaching tests

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.
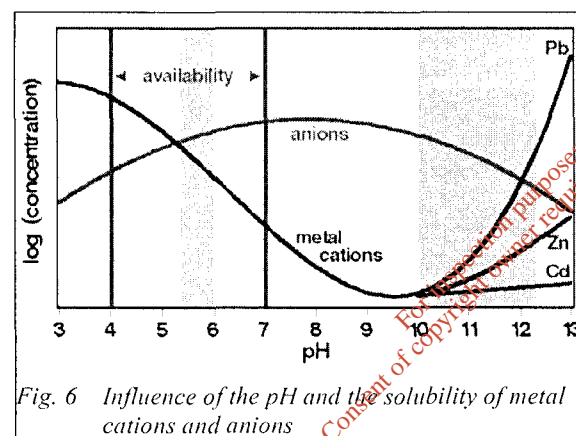
Fig. 6   Influence of the pH and the solubility of metal
cations and anions

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 CO_2 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 CO_2 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.
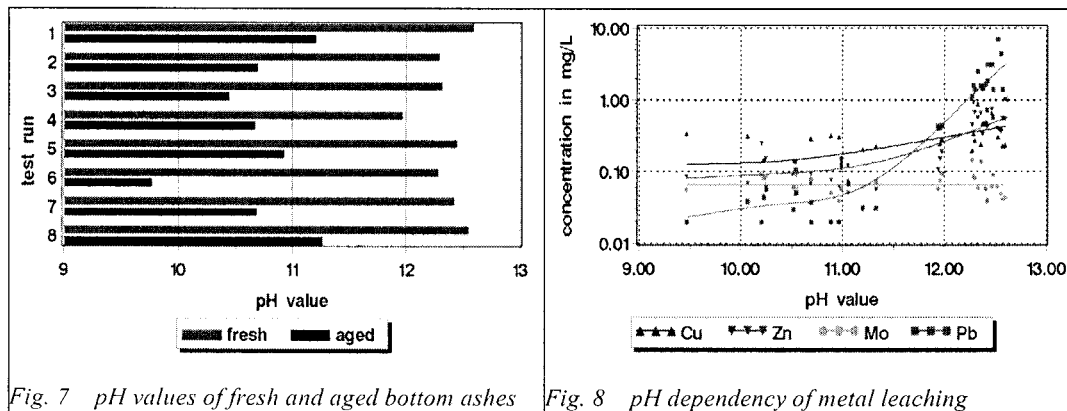
Fig. 7   pH values of fresh and aged bottom ashes | *Fig. 8   pH dependency of metal leaching*

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.
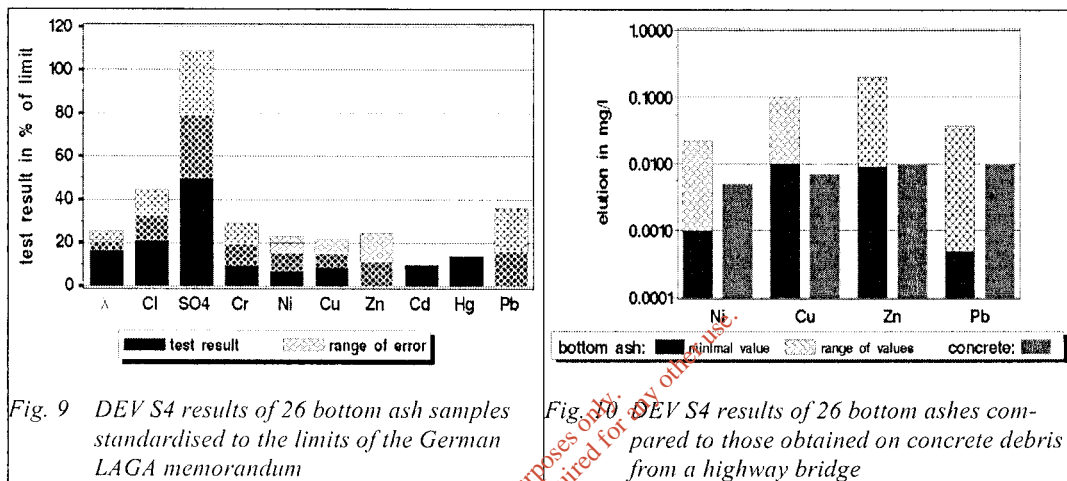

Fig. 9   DEV S4 results of 26 bottom ash samples standardised to the limits of the German LAGA memorandum | *Fig. 10   DEV S4 results of 26 bottom ashes compared to those obtained on concrete debris from a highway bridge*

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 seams 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].

Fig. 11 DEV S4 results of thermally treated bottom ashes and concrete

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.

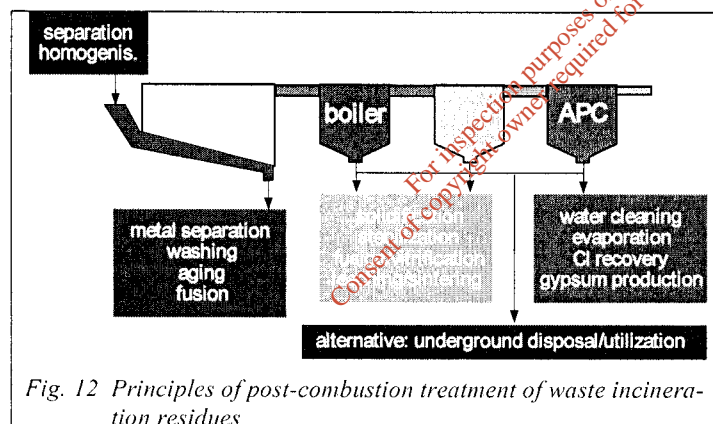*Fig. 12 Principles of post-combustion treatment of waste incineration residues*

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

	€/Mg of bottom ash	€/Mg of MSW
landfill	35	12
pre-treatment for utilisation	20	7
fusion (fossil fuel, no pre-treatment)	100	30
fusion (fossil fuel, scrap removal)	130	45
fusion (electric heating)	120	40
fusion processes in Japan	100	30
fusion processes estimates (IAWG)	180	60

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.

Table 4   Procedures for treatment of filter ashes

principle	process	
solidification/ stabilisation	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)
thermal treatment	PCDD/F destruction	(Hagenmaier drum)
	sintering	(mineral respeciation)
	fusion	(melting without additives)
	vitrification	(melting with additives)
combined process	acid extraction + sintering	(3R Process)

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.
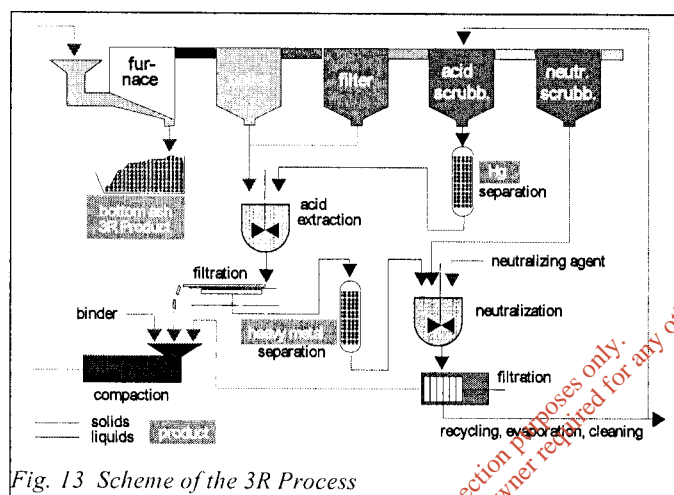

Fig. 13 Scheme of the 3R Process

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 5   Cost estimates for land filling and treatment of filter ashes (disposal costs not included)*

	€/Mg of filter ash	€/Mg of MSW
disposal on special landfill	200	3
utilisation in salt mines	100	1.5
cement solidification*	25	0.5
stabilisation*	80	2
solidification+stabilisation*	120	2
3R Process	120	2
fusion/vitrification	180	3

disposal strategies. 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 guarantied 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 Cl_2 [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 6   Cost estimates for land filling and treatment of scrubbing residues

	€/Mg of MSW
dry sorption	36
dry sorption without residue disposal	23
dry sorption with utilisation for backfilling of caverns	32
semi-dry sorption	29
wet scrubbing with waste water discharge	25
wet scrubbing with spray dryer	28
wet scrubbing with external evaporation	29
wet scrubbing with HCL/gypsum production	35

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:

o   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,.

o   Simple washing of the bottom ashes, preferentially in-plant in a modified quench tank, reduces the leaching of chlorides to very low levels.

o   The resulting products have a high potential for utilisation, e.g. according to German regulations in road construction.

o   Post-combustion treatment of bottom ashes increases the incineration cost without improving the elution stability significantly.

o   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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Subject: FW: My email dated 7/8/2012 ref W0 127-03 "Exothermic Reactions---etc."

From: Paddy Boyle [mailto:paddyboylerush@hotmail.com]
Sent: 08 August 2012 09:04
To: Wexford Receptionist
Subject: My email dated 7/8/2012 ref W0 127-03 "Exothermic Reactions---etc."

Waste for Whiteriver landfill will be tested - News - Drogheda-Independent.ie
18 month contract to dump waste at Whiteriver landfill - News - Argus.ie
Attention: Mr Frank Clinton, Mr Brian Meeney, Dr. Marcus Ford

Dear sirs,
Could you please add the information contained in the above links to the submission and note the correction
"Whiteriver" and not "White Water" Landfill, County Louth.
Yours truly,
Patrick Boyle

This email has been scanned by the Symantec Email Security.cloud service.
For more information please visit http://www.symanteccloud.com

1

Drogheda-Independent.ie

Waste for Whiteriver landfill will be tested

EPA WILL TEST AND MONITOR

LOUTH County Council has stressed that any waste from the Indaver Ireland waste-to-energy facility at Carranstown, Co. Meath will only be accepted in Louth with the approval of the Environmental Protection Agency (EPA) who will conduct rigorous testing and monitoring to ensure the material is suitable for transfer.

Des Foley, Director of Services, Louth Local Authorities, said: 'It's important that we reassure the community who live near the Whiteriver site that we will only accept material from Indaver's facility at Carranstown once we are fully satisfied that it is suitable for transfer and landfill disposal. We will work closely with the EPA in this regard.

'It should also be stressed that the materials we propose accepting from Indaver Ireland is the non-hazardous bottom ash. In total, it is proposed that 8,000 - 10,000 tonnes would be transferred to Whiteriver this year while it is also intended that – subject to EPA approval – material would be accepted in 2012 and 2013.'

Dealing with returned to the the material being Republic having been illegally dumped in Northern Ireland, Mr. Foley said: 'This arrangement is part of a longstanding commitment made by the authorities in the Republic to counterparts in Northern Ireland after the scale of the illegal dumping of material at 17 sites in Northern Ireland was established. In an ongoing operation, we will be taking 10,000 tonnes of this material between now and late October.

'More than 700 tonnes of what is nonhazardous waste has been safely transferred to date. The material has already biodegraded over several years so there will be no additional odour issues.

'The waste being brought to Whiteriver is coming from Co. Down and ours is the nearest Republic of Ireland landfill facility to which it can be transported

'All costs associated with this operation are being met by central government so there is no financial impact in Louth. The repatriated waste will also come within the limits we are allowed under the EPA's licensing regime for Whiteriver.

'The site is closed to commercial waste operators since mid-July and will remain open until October for domestic waste customers.' LOUTH-BASED tidal technology specialist OpenHydro and French utility company EDF are in the final stage of deploying the first of four 16m tidal turbines off the coast of Paimpol-Bréhat.

Last Wednesday the first turbine assembled at DCNS shipyard in Brest was towed from the harbour for a series of commissioning tests at sea prior to installation.

This is the first stage of a project which in 2012 will create the world's largest tidal array generating power onto the French grid. The turbines are supplied by Greenore-based OpenHydro and each has the capacity to generate over 2MW of energy.

OpenHydro and its partner DCNS have now completed the assembly of the first turbine for the PaimpolBréhat project. The installation is performed using the custom designed installation barge, the 'OpenHydro Triskell'.

James Ives, chief executive, OpenHydro, said: We are delighted to be working with EDF on what is set to be the world's first large-scale, gridconnected tidal energy farm and France's first offshore tidal installation. EDF's vision to develop this exciting project places France and EDF at the forefront of this new form of renewable energy generation.'

The turbine will now be towed to the deployment site off the island of Bréhat, near Paimpol in Côtes-d'Armor, where it will be deployed on the seabed at a depth of 35 meters. For two months the turbine and subsea base, which have a combined weight of 850 tonnes and a height of 22 metres, will be tested.

The objective of the test is to prepare for the implementation of the world's largest tidal array which is scheduled for installation in 2012. The electrical output of the completed tidal farm will power 4,000 homes.

OpenHydro has received support from Ireland's Ocean Energy Development Unit who have supported the turbine system design.

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Argus.ie

18 month contract to dump waste at Whiteriver landfill

By OLIVIA RYAN
Wednesday October 12 2011

INDAVER Ireland have signed an 18 month contract with Louth County Council to dump incinerator ash at the Whiteriver landfill, councillors were told last week.

The waste management company are preparing to move 80,000 tonnes of non hazardous ' bottom ash' to the mid Louth landfill at Philipstown. After serious opposition from local residents, and county councillors, a special meeting of Louth County Council was held on Wednesday last to debate concerns raised about the material set to be dumped at Whiteriver.

Director of Services, Des Foley gave an outline of the landfill history. Originally opened in 1984, the current planning permission is set to expire in 2018.

He explained that the council have operated the landfill on a six phased basis, with the first four phases complete.

The local authority are currently working on phase five, which should be completed by spring 2015. By that stage the landfill will have accumulated up to 150,000 tonnes of waste.

Mr. Foley said there are no plans to complete the sixth phase, adding that there is a 30 year aftercare liability on the council after the landfill closes.

He said that as and from last weekend Whiteriver is now closed to commercial waste and household waste dumping, as the landfill was approaching capacity on its yearly limits.

The Director of Service explained that Louth County Council had also agreed to a request from the Department of Environment to accept up to 20,000 'repatriated' waste from Northern Ireland.

The waste, which originated in the south, was found dumped across 17 sites in the north, and Whiteriver began taking in the returned rubbish at the end of August.

He told the members that the council has also entered into a contract with Indaver Ireland to accept ' bottom ash' material from their facility at Carronstown, subject to EPA approval.

Mr. Foley said the council are currently awaiting the results of testing by the EPA to determine the ash is non hazardous.

He added that bottom ash ' has many advantages over municipal waste. It is clean waste, with no leachate and no emissions.'

He also highlighted a new 'waste to energy' scheme at Whiteriver which will see a new generator launched.

- OLIVIA RYAN

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