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Modeling waste incineration for life-cycle inventory analysis in Switzerland

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This paper proposes a mathematical model for life-cycle inventory analysis (LCI) of waste incineration in Switzerland. In order to model conventional and new incineration technologies adequately, fundamental aspects of the different technologies relevant for the LCI are discussed. The environmental impact of these technologies strongly depends on the assessment of the long-term emissions of the solid incineration residues and is therefore related to value based decisions about the time horizon considered. The article illustrates that the choice of the landfill model has a significant influence on the results of life-cycle assessment of waste incineration.

Keywords: landfill models, life-cycle assessment (LCA), life-cycle inventory model (LCI), long-term emissions, product-dependent allocation, waste incineration models

1. Introduction

1.1. Waste treatment in Switzerland

Switzerland was one of the first countries in Europe to install an incinerator for municipal solid waste (MSW) 100 years ago [1]. Since then, and especially after World War II, MSW incineration has expanded considerably. In 1998, of the 611 kg of waste per year produced by an average Swiss citizen 46% was incinerated, 41% recycled, and 11% landfilled (figure 1) [2]. One of the objectives of Swiss environmental policy is the reduction of waste by encouraging recovery and promoting the treatment of the remaining waste in an environmentally sustainable way. Waste should be landfilled only if the emission flows do not change the chemical or physical properties of the air, water or soil compartment. This regulation implies that the landfills will not need to be treated in the future [3]. In order to achieve these goals, recycling has been encouraged by providing a good infrastructure, adequate information policies and by introducing financial incentives. The population is taxed for each bag of produced waste (commensurate with volume), but the collection of paper, carton, glass, PET, metals, and compost remains free of charge in most municipalities. These incentives have lead to a decline in waste production and an increase in the recycling activities (figure 1). The biggest proportions of materials collected for recycling are paper, compost, and glass [2].

Municipal solid waste landfills are not considered environmentally sustainable due to the potential harm of organic releases. Hence, from the year 2000 onwards, Swiss legislation forbids direct landfilling of burnable waste [4]. This regulation has caused the gradual decline of MSW landfilling illustrated in figure 1. As a consequence, current waste treatment in Switzerland only consists of incineration and recycling activities. As the predominant treatment, incineration (figure 1) has been the primary concern of environmental legislation and research during the last ten years. In the 1970s and 1980s, the atmospheric emissions of MSW incinerators were identified as a major problem [5]. Pollutants such as HCl, NO_x, SO_x, Cd, Cu, Pb, Hg, and dioxins and furans contributed significantly to environmental problems like acidification, human toxicity, eco-toxicity, eutrophication, as well as summer and winter smog [6]. Since 1985, Swiss legislation limits the emissions of heavy metals, HCl, SO_x, particles, and NO_x [7] and thus requires the implementation of flue gas cleaning systems in municipal solid waste incineration (MSWI) plants. Consequently, research activities have been focused on the long-term release of heavy metals from the incineration residues landfilled. At the beginning of the 1990s, a break-through in new incineration technologies was expected to improve the quality of incineration residues. An extensive research program (1992–2001), called the *Integrated Program Waste*, was launched with an emphasis on technological development and some research contributions on economical, environmental, and social compatibility [8]. From this program, several new technologies have become available that allow the separation of heavy metals from the incineration residues (mechanically or thermally) and the production of a homogeneous and vitrified output [9].

In spite of the advantages of these technologies [10], there are not many plants in operation yet. One reason is that the novelty of the new technologies causes misgivings among plant operators and the population in general. Another obstacle is the absence of demand for recovered valuables such as heavy metals [8]. Finally, the stimulation of recycling activities has generated an over-capacity of MSW incinerators in some regions with the corresponding economic problems, thereby becoming an inhibiting factor for technologi-

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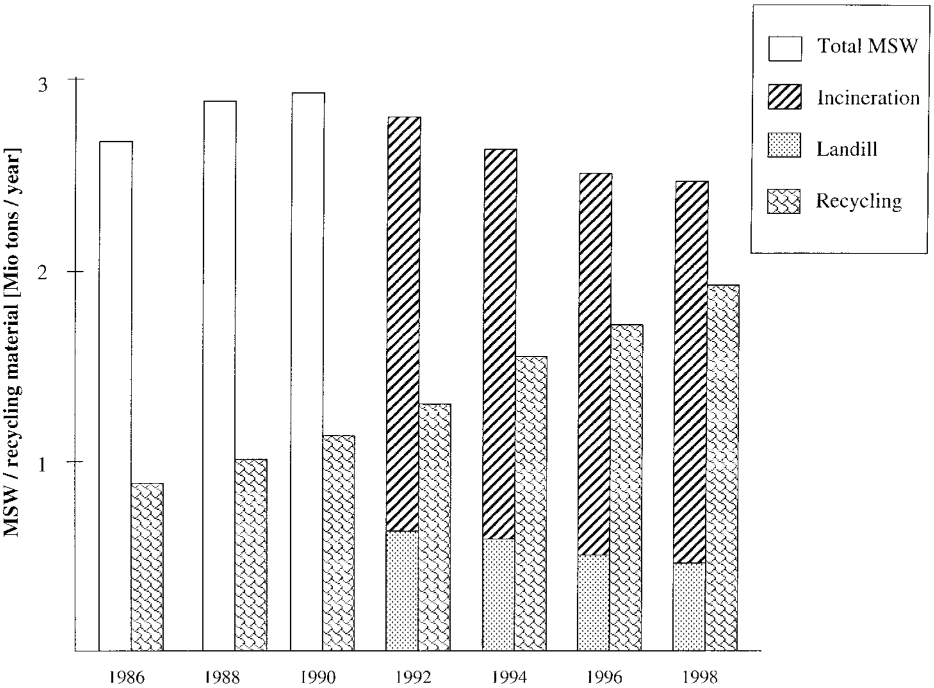


Figure 1. Annual generation of municipal solid waste (MSW) and recycling material in Switzerland (data on the fraction of landfilled and incinerated waste has only been available for the years 1992–1998) [2].

cal innovation. The scarce financial resources and the fluctuations in quantity and quality of waste flow have limited the demand for new incinerators, which have high fix costs and long lead times. The eagerness to implement new and barely tested technologies has been even more limited. Nevertheless, these technologies are still relevant since the long-term behavior of waste incineration residues represents an unsolved environmental problem. On the bright side, since 2000, waste previously landfilled is now also being incinerated. This trend will certainly improve the economic situation of plant operators and may raise the demand for new technologies.

Given this status-quo, there is an increasing demand within Switzerland to model the life-cycle of waste incineration. Such models should therefore enable an evaluation of current as well as recently developed incineration technologies including the assessment of the corresponding environmental impacts. In this paper we present a model that can be used in product oriented life-cycle assessment studies (LCA) covering the last step of a product's life cycle (waste treatment) [11,12]. We will discuss (a) some fundamental aspects of the different waste incineration technologies relevant for an LCI analysis and (b) assumptions required for an evaluation of long-term emissions of incineration residues. To this end, the model development will be emphasized throughout the article including a brief discussion of its applicability to the situation in Switzerland.

1.2. Existing models for LCA application

Previous life-cycle assessment studies have paid little attention to waste treatment. Many studies list the amount of solid waste without considering the emissions and resource use resulting from incineration or landfilling. Switzerland was one of the first countries to publish a model and comprehensive guidelines for the calculation of inventory data from landfills, waste incineration and wastewater treatment [14]. The model has been widely applied [10,11,15]. Nowadays, there are several similar approaches, e.g., in Europe [16–20]. All of these models are similar with respect to the representation of the incineration plant and basic allocation procedures. For instance, there seems to be an agreement in distinguishing between process and product related emissions. Process related emissions are independent from the waste input. On the contrary, product related emissions are caused by the concrete input components of the waste. The assignment of emissions to either one of those two groups is fairly similar in all models. However, there are a few exceptions like the NO_x emissions which can be formed by several reaction paths (e.g., fuel NO_x, thermal oxidation of N_2). In the different models they are either characterized as product dependent [19], process dependent [18,20], or as a combination of both [14,16]. Concerning the product related emissions, transfer coefficients are used to relate waste input and output (emissions and solid outputs). No consensus has been reached yet on whether these coefficients should be regarded as constant [17,20] or as a function of the waste input [18,19,21].

All above-mentioned models are restricted to the modeling of the conventional grate process, although several new incineration technologies have become available in this past decade. Whereas the incineration plant is always considered to be part of the system, the model used in this work was the first to consider transport, infrastructure, and ancillary

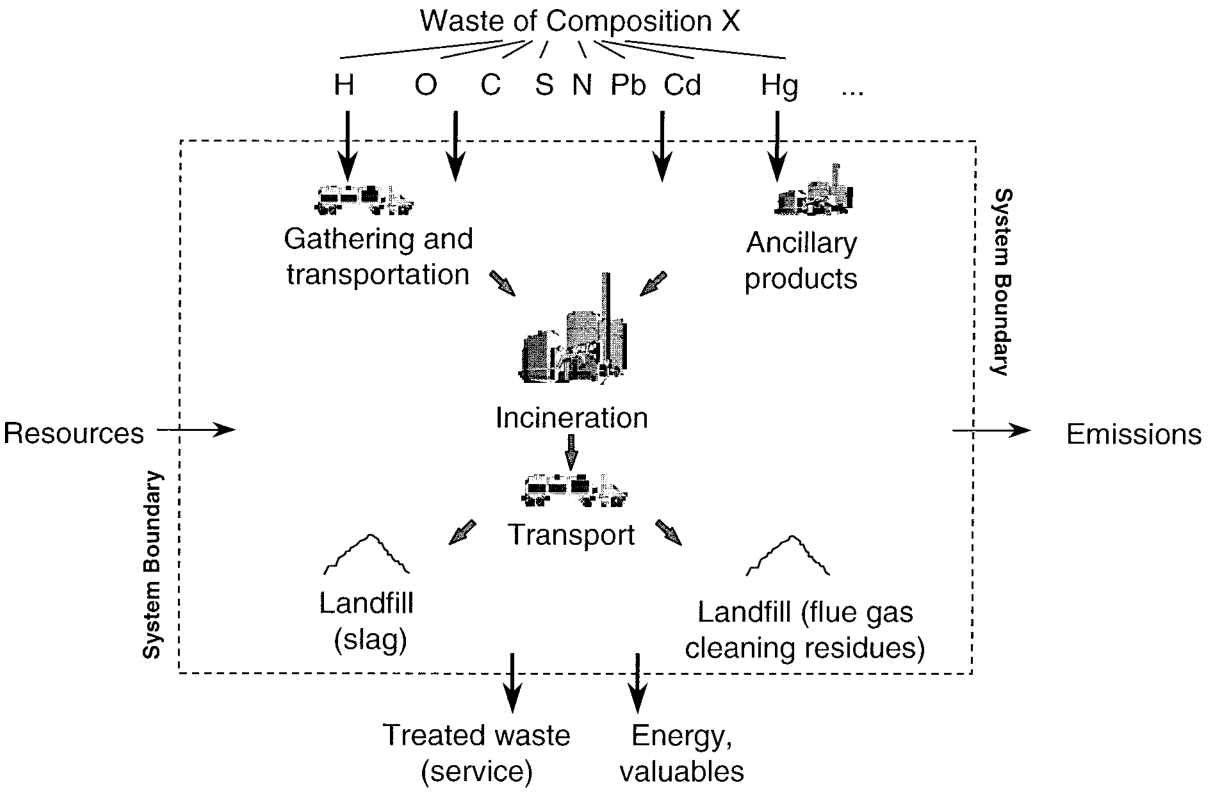


Figure 2. The system *waste incineration* consists of the incineration of waste, the landfilling of incineration residues, the gathering and transportation of waste, infrastructure, and the production of ancillary products. The system primarily serves to treat waste (the functional unit corresponds to the incineration of 1 kg of waste of a certain composition).

products [14]. The landfills of incineration residues generally lie within the system boundaries. However, the models vary significantly in regard to the temporal system boundary, which has a strong influence on the sum of anticipated emissions from landfills. The range of time horizons considered varies from a few decades [22] to infinity [14,16]. Recent studies have shown that if the same weight is assigned to long-term emissions as to immediate emissions, the impacts of the MSWI residue landfills greatly surpass the impacts caused by the incineration process [13,23,24]. In spite of the importance of landfill emissions, only Sundqvist et al. [16,17] and our group [21] have tried to model the emission concentrations as a function of time for LCA purposes. So far, these landfill models have been restricted to a selected group of substances such as heavy metals.

The present work describes our approach with respect to the aforementioned topics. Any process potentially contributing to the environmental impact of the system (figure 2) has been considered. An allocation procedure distinguishing between the transfer of inert and burnable materials in the incinerator is proposed. This approach is expected to represent physical–chemical relationships more precisely than the conventional approach of average transfer coefficients for all waste materials. The incineration model has been extended to new technologies in order to allow decision support when the construction of a new plant is at stake. The input to the landfill model comprises the amount and composition of solid output of waste incineration as calculated in the incineration model. Since the long-term releases of heavy metals from landfills have been identified as potentially relevant substances in previous work [10,13,24], emphasis is placed on the modeling of these emissions.

2. Aims of the model and system description

The goal of the model is to provide a tool to calculate inventory data, thereby enabling the environmental evaluation of the incineration of various waste input compositions and the comparison of various incineration technologies. The functional unit is defined as the incineration of one kilogram of waste of a certain composition. The waste input can be specified on the basis of materials or elemental composition. The system consists of the gathering and transportation of municipal waste, the incineration, the production of ancillary products, and the landfills of slag and other incineration residues (figure 2). The incineration can be done with different technologies. Therefore, the user of the model can choose between various technologies or a combination of them. The following technologies have been modeled:

- Grate incinerator with or without flue gas cleaning.
- Technologies with a reductive and oxidative zone (VS-Process [25] and Thermoselect [26]).
- Thermal filter ash treatment [27], mechanical [28] and thermal slag treatment modules [29] that can complement the grate incinerator and possibly the VS-Process.

Table 1

Co-products of the system waste incineration and potential reference systems. The emissions produced and the resources used by a reference system for the generation of the same amount of energy or metals can be subtracted from the inventory table of the system *waste incineration*.

Co-products of the system waste incineration	Potential reference systems
Net electricity (produced electricity – internal use)	Average European or Swiss electricity production
Net heat (produced heat – internal use)	Heat production by oil or gas
Metals	Metal production considering average Swiss production conditions

The integration of new technologies involves many uncertainties, since in most cases no full-scale plants have successfully been taken into operation yet. It is assumed that the new technologies behave as expected from results of pilot plants. The data should be revised as soon as results from full-scale operating plants are available.

With regard to the ancillary products, average Swiss production conditions are assumed [14]. Transport distances and energy efficiency are based on average values in Switzerland, but can be modified. It is assumed that slag is landfilled, whereas the other incineration residues are either deposited in regular landfills or in salt mines hundreds of meters below the earth's surface. Since the deposition of filter ashes in salt mines does not lead to predictable product dependent emissions, the modeling is restricted to a listing of process dependent emissions (infrastructure and transport) that are allocated as a total to the functional unit (see [30]). Therefore, the present paper focuses on the modeling of the emissions of slag and other residues on regular landfills.

3. Model design and allocation

Allocation is the partitioning of the input and output flows (emissions/resource use) to the products of a unit process. The system of waste incineration (figure 2) is a multi-input/multi-output problem because various waste fractions enter the system and various outputs (the service waste treatment, energy, and recycling products) leave it. Therefore, an allocation is necessary. In the present study, the allocation principle of system enlargement [31] is applied whenever possible. However, the emphasis lies on the allocation based on physical–chemical relationships [31] (product dependent allocation). In the following paragraphs, we will describe how the allocation was performed in each part of the system (figure 2).

3.1. Modeling the incineration process

According to the allocation principle of system enlargement, a bonus should be granted for a co-product that has the same quality as a product fabricated by an alternative system [31]. The system *waste incineration* primarily serves to treat waste, but in addition other co-products such as electricity, heat, and, in some cases, metals are generated. Therefore, the emissions produced and the resources consumed by a reference system for the production of the same amount of energy or metals might be subtracted from the inventory table. In our model, the decision is left to the LCA practitioner as to which co-products system enlargement should be applied and which reference system should be used (table 1).

The allocation principle of physical–chemical causalities is illustrated in the ISO Guidelines [31] using the example of waste incineration: all cadmium emissions of an incineration plant are directly related to the amount of cadmium in the waste entering the system (mass balance). Such a clear relation exists between almost every waste input substance and emission. Unfortunately, it is not sufficient to know that each kilogram of a substance also leaves the system as an emission. Rather, it is necessary to distinguish between emissions to the air, water and soil. In some cases such a clear causality between the waste input and the output to the environment cannot be observed, for instance, when emissions are formed due to process conditions such as combustion temperature. In that case, the emissions are allocated as a total to the functional unit (e.g., dioxins and furans, CO, VOC, and partially NO_x), irrespective of the input substances (process specific allocation) [14].

Concerning the product dependent emissions, we represent the relation between input waste and output to air, water, and solid incineration residues by transfer coefficients. The sum of transfer coefficients for each given substance equals 100%. A set of average transfer coefficients is provided for each technology using data from the literature [14,26,29,32], plant operators, and technology developers. The direct emissions to the air and water as well as the composition of slag and other incineration residues can be calculated by multiplying the waste component vector with the transfer coefficient matrix (equation (1)).

$$\begin{pmatrix} c_1 \\ c_2 \\ \dots \\ c_n \end{pmatrix} * \begin{pmatrix} tc_{11} & tc_{12} & \dots & tc_{1m} \\ tc_{21} & tc_{22} & \dots & tc_{2m} \\ \dots & \dots & \dots & \dots \\ tc_{n1} & tc_{n2} & \dots & tc_{nm} \end{pmatrix} = \begin{pmatrix} y_{11} & y_{12} & \dots & y_{1m} \\ y_{21} & y_{22} & \dots & y_{2m} \\ \dots & \dots & \dots & \dots \\ y_{n1} & y_{n2} & \dots & y_{nm} \end{pmatrix}, \tag{1}$$

where c is the composition of waste (g/kg waste), tc are the transfer coefficients (%, the sum of each row equals 100%), y represents the output (emissions and solid output, g/kg waste).

In equation (1), the number of elements considered equals n and the number of outputs is m. In our model, 33 elements are considered [14]. The value of m depends on the chosen technology option (table 2).

Table 2

Calculation of emissions and solid residue outputs from incineration exemplified by S, Cu, and Pb. The product of the waste composition vector and the transfer coefficients gives the composition of the incineration output. With respect to grate incineration, the inert and burnable waste composition vectors are multiplied with the transfer coefficient tables for inert and burnable waste respectively. The sum of the inert and burnable output is the resulting output. The flue gas is emitted to the air and the wastewater to the surface water.

Technology	Waste input		Transfer coefficients					Output				
Grate incinerator without proper flue gas cleaning (electric precipitator only)	Element	Amount in waste	Flue gas	Slag 1)	Other residues			Flue gas	Slag	Other residues		
		Inert fraction (C_{inert})	*Inert fraction (TC_{inert})*					*Total output (C_{inert} * TC_{inert} + C_{burn} * TC_{burn})*				
		(g/kg)	(%)	(%)	(%)			(g/kg)	(g/kg)	(g/kg)		
	S	0.24	0.0	100.0	0.0			0.12	0.32	0.12		
	Cu	0.0020	0.0	100.0	0.0			0.00028	0.011	0.0019		
	Pb	0.0030	0.0	100.0	0.0			0.0014	0.0044	0.016		
		Burnable fraction (C_{burn})	*Burnable fraction (TC_{burn})*									
		(g/kg)	(%)	(%)	(%)							
	S	0.33	37.3	25.5	37.2							
	Cu	0.011	2.5	80.3	17.2							
	Pb	0.019	7.4	7.2	85.4							
Modern grate incinerator with proper flue gas cleaning	Element	Amount in waste	Flue gas	Slag 1)	Other residues	Wastewater		Flue gas	Slag	Other residues	Wastewater	
		Inert fraction (C_{inert})	*Inert fraction (TC_{inert})*					*Total output (C_{inert} * TC_{inert} + C_{burn} * TC_{burn})*				
		(g/kg)	(%)	(%)	(%)	(%)		(g/kg)	(g/kg)	(g/kg)	(g/kg)	
	S	0.24	0.0	100.0	0.0	0.0		0.00066	0.42	0.12	0.023	
	Cu	0.0020	0.0	100.0	0.0	0.0		0.0	0.011	0.0022	0.0	
	Pb	0.0030	0.0	100.0	0.0	0.0		0.0	0.0043	0.018	0.0	
		Burnable fraction (C_{burn})	*Burnable fraction (TC_{burn})*									
		(g/kg)	(%)	(%)	(%)	(%)						
	S	0.33	0.2	55.4	37.3	7.1						
	Cu	0.011	0.0	80.1	19.9	0.0						
	Pb	0.019	0.0	6.6	93.4	0.0						
VS-Process with proper flue gas cleaning	Element	Amount in waste	Flue gas	Slag	Other residues	Wastewater		Flue gas	Slag	Other residues	Wastewater	
		(g/kg)	(%)	(%)	(%)	(%)		(g/kg)	(g/kg)	(g/kg)	(g/kg)	
	S	0.57	0.2	58.1	35.0	6.7		0.0011	0.33	0.20	0.038	
	Cu	0.013	0.0	87.0	13.0	0.0		0.0	0.011	0.0017	0.0	
	Pb	0.022	0.0	8.0	92.0	0.0		0.0	0.0018	0.020	0.0	
Thermoselect-Process with proper flue gas cleaning	Element	Amount in waste	Flue gas	Mineral output	Metals	Heavy metals	Other residues	Flue gas	Mineral output	Metals	Heavy metals	Other residues
		(g/kg)	(%)	(%)	(%)	(%)	(%)	(g/kg)	(g/kg)	(g/kg)	(g/kg)	(g/kg)
	S	0.57	1.0	18.0	7.0	0.0	74.0	0.0057	0.10	0.040	0.0	0.42
	Cu	0.013	0.0	19.0	78.0	3.0	0.0	0.0	0.0025	0.010	0.00039	0.0
	Pb	0.022	0.0	17.0	4.0	64.0	15.0	0.0	0.0037	0.00088	0.014	0.0033
Thermal filter ash treatment	Element	Amount in filter ash (filter ash from modern grate)	Metals	Remaining slag				Metals	Remaining slag			
		(g/kg)	(%)	(%)				(g/kg)	(g/kg)			
	Cu	0.0022	99.0	1.0				0.0022	0.000022			
	Pb	0.018	99.0	1.0				0.018	0.00018			
Mechanical slag treatment	Element	Amount in slag (slag from modern grate)	Metals	Remaining slag				Metals	Remaining slag			
		(g/kg)	(%)	(%)				(g/kg)	(g/kg)			
	Cu	0.011	60.0	40.0				0.0066	0.0044			
Thermal slag treatment	Element	Amount in slag (slag from modern grate)	Flue gas	Metals	Mineral output			Flue gas	Metals	Mineral output		
		(g/kg)	(%)	(%)	(%)			(g/kg)	(g/kg)	(g/kg)		
	S	0.42	63.0	27.0	10.0			0.26	0.11	0.042		
	Cu	0.011	2.0	88.0	10.0			0.00022	0.0097	0.0011		
	Pb	0.0043	97.0	1.0	2.0			0.0042	0.000043	0.000086		

1) The transfer coefficients for slag should be independent from the gas purification system. However, different plants were used as data sources [14,32].

Table 3

Description of the grate technology, the VS-, and the Thermoselect-Process with respect to the production and use of energy, the use of ancillary products, and the quality of solid incineration output.

Technology	Energy production	Internal energy use	Use of ancillary products	Quality and composition of solid outputs[1]	Potential recycling materials
Grate incinerator without proper flue gas cleaning (electric precipitator only) [14]	Heat: 26% of lower heating value[2] (3.2 MJ/kg waste) Electricity: 10% of lower heating value (1.2 MJ/kg waste)	Heat: 0.08 MJ/kg waste Electricity: 0.12 MJ/kg waste	–	Heterogeneous slag with 45% of volume ash, 40% melted material, 5% fractional glass [29], and 2–5% of weight TOC	Fe from slag
Grate incinerator with proper flue gas cleaning (wet flue gas cleaning, wastewater treatment, nitrogen reduction) [14,32]	Heat: 37% of lower heating value (4.5 MJ/kg waste) Electricity: 18% of lower heating value (2.2 MJ/kg waste)	Heat: 0.24 MJ/kg waste Electricity: 0.36 MJ/kg waste	NaOH, CaO, HCl, chemicals (organic and inorganic), NH_3	Heterogeneous slag with 45% of volume ash, 40% melted material, 5% fractional glass [29], and 2–5% of weight TOC	Fe from slag, HCl, and NaCl
VS-Process with proper flue gas cleaning (wet flue gas cleaning, wastewater treatment, nitrogen reduction) [14,25,32]	Heat: 37% of lower heating value (4.5 MJ/kg waste) Electricity: 18% of lower heating value (2.2 MJ/kg waste)	Heat: 0.24 MJ/kg waste Electricity: 0.36 MJ/kg waste	NaOH, CaO, HCl, chemicals (organic and inorganic), NH_3	More homogeneous than conventional slag but still heterogeneous, <90% of volume glassy matrix [52], no organic compounds, low heavy metal content except Cu	Fe from slag, HCl, and NaCl
Thermoselect-Process with proper flue gas cleaning (wet flue gas cleaning and wastewater treatment) [26,29]	Synthetic gas: 65% of lower heating value of waste and gas (9.1 MJ/kg), electricity from this synthetic gas: 32% of energetic content (2.7 MJ/kg after subtracting internal use)	Gas: 1.9 MJ/kg waste (plus 0.6 MJ/kg as synthetic gas) Electricity: 0.5 MJ/kg waste	O_2, NaOH, HCl, chemicals (organic and inorganic)	>95% of volume glass phase, relatively homogeneous, no organic compounds [29]	Fe, Pb, and Zn, S, HCl, and NaCl; the mineral output could be used by the building industry

[1] The focus is put on slag output because Swiss filter ash is generally deposited in German subsurface deposits without predictable further emissions (largely independent of composition and quality).

[2] Average lower heating value of waste in Switzerland: 12.3 MJ/kg [14].

Table 4

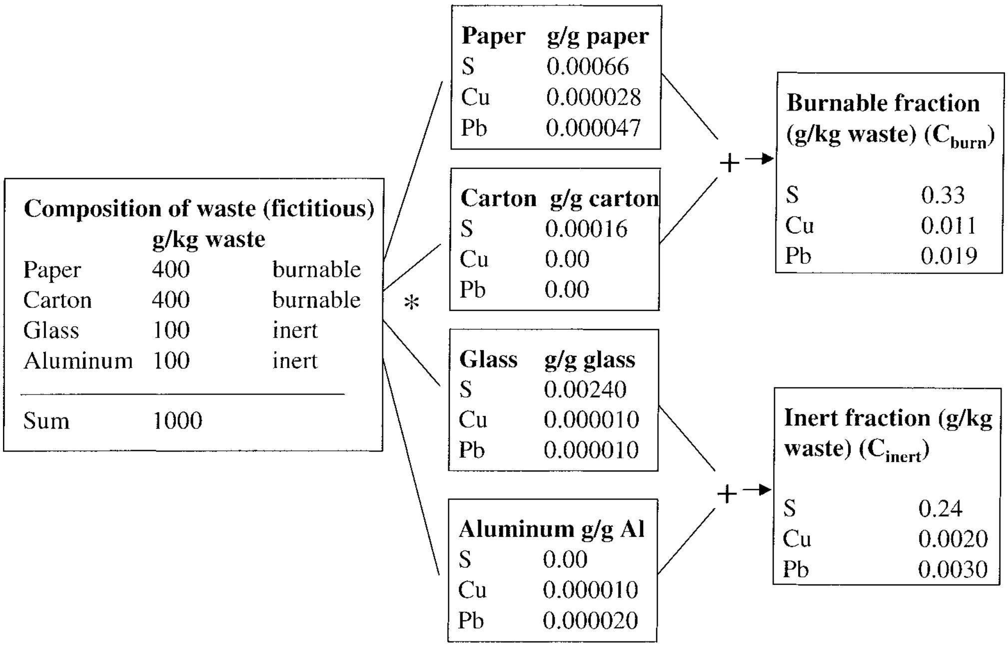
Description of thermal and mechanical residue treatment technologies with respect to their ability to be combined with other technologies, the use of energy and ancillary products as well as the production of recycling materials.

Technology	Potential technology combination	Additional internal energy use	Additional use of ancillary products	Potential recycling materials
Thermal filter ash treatment [27]	Grate incinerator or VS-Process	Electricity: 2.34 MJ/kg filter ash Heat: 5.58 MJ/kg filter ash Natural gas: 0.04 kg (lower heating value: 1.84 MJ) per kg filter ash	Coal, HCl (if used from flue gas, less CaO will be needed for neutralization)	It is not clear how easily recyclable the metal fraction is (therefore landfill and recycling scenario)
Mechanical slag treatment [28]	Grate incinerator	Electricity: 0.09 MJ/kg slag	–	Cu, Al in an easily recyclable form
Thermal slag treatment [29]	Grate incinerator	Electricity: 0.9 MJ/kg slag Oil: 1.62 MJ/kg slag, Natural gas: 0.8 MJ/kg mineral output	Additional NaOH, CaO, HCl, chemicals (organic and inorganic), NH_3 for flue gas cleaning	It is not clear how easily recyclable the metal fraction is (therefore landfill and recycling scenario), mineral output of a high quality: >99% of volume glass phase, very homogeneous and without organic compounds

Tables 2–4 show how the various technologies are represented in the model. Four key features, which distinguish the technologies from each other, were chosen as an illustration: transfer coefficients (table 2), energy balance, ancillary products, and quality of slag and potential recycling products (tables 3 and 4). Transfer coefficients are illustrated showing 3 out of 33 elements as representatives for typical emissions to air (S) and to water by landfill leaching (Cu, Pb) (table 2). The calculation of the amounts of ancillary products according to the target substances in the waste is not shown (for more details see [14]).

3.1.1. Grate incineration

Grate incineration is by far the most widely used technology in the world. Therefore it was possible to take all data concerning transfer coefficients, use of ancillary products,



Composition of waste (fictitious) g/kg waste		
Paper	400	burnable
Carton	400	burnable
Glass	100	inert
Aluminum	100	inert
Sum	1000	

Paper	g/g paper
S	0.00066
Cu	0.000028
Pb	0.000047

Carton	g/g carton
S	0.00016
Cu	0.00
Pb	0.00

Glass	g/g glass
S	0.00240
Cu	0.000010
Pb	0.000010

Aluminum	g/g Al
S	0.00
Cu	0.000010
Pb	0.000020

Burnable fraction (g/kg waste) (C_{burn})	
S	0.33
Cu	0.011
Pb	0.019

Inert fraction (g/kg waste) (C_{inert})	
S	0.24
Cu	0.0020
Pb	0.0030

Figure 3. Calculation of an inert and burnable waste composition vector from input materials: All input materials are classified as burnable or inert. After multiplying the amounts of materials in the waste with their elemental compositions, the sum of all inert composition vectors and the sum of all burnable composition vectors is formed.

and energy efficiency from measurements of existing plants ([14,32], tables 2 and 3). The grate technology considered in the model consists of an incinerator, an after burn chamber, a boiler to recover energy, an electric precipitator to eliminate dust, and optionally a (wet) flue gas cleaning, a wastewater treatment installation, and a NO_x reduction facility. The flue gas cleaning and wastewater treatment require the ancillary products listed in table 3. The energy efficiency varies considerably among plants depending on the location (access to district heating net for selling heat) and the age of the incinerator (e.g., incinerators without proper flue gas cleaning are usually old installations and have a lower energy efficiency). The proposed values for the energy efficiency (table 3) are only defaults, which should be adapted according to the boundary conditions.

The transfer coefficients are a function of the waste input and, therefore, should be adapted accordingly. The basic idea is that inert materials (glass, ceramics, big pieces of metals) are completely transmitted to the slag in the incineration process. Therefore, the average transfer coefficients are not valid for inert materials but some which state 100% values in the column for slag transmission and 0% in the other columns (see TC_{inert} in table 2). These transfer coefficients for inert materials represent the first set of coefficients. A second set for all other burnable substances can be calculated according to equations (2a) and (2b).

$$TC_{\text{burn}} = \frac{Y_{\text{avg}} - TC_{\text{inert}} * C_{\text{inert}}}{C_{\text{burn}}}, \tag{2a}$$

$$\begin{aligned} Y_{\text{avg}} &= C_{\text{avg}} * TC_{\text{avg}} \\ &= C_{\text{inert}} * TC_{\text{inert}} + C_{\text{burn}} * TC_{\text{burn}}, \end{aligned} \tag{2b}$$

where Y corresponds to the output matrix (incineration residues and emissions, g/kg waste), C is the waste composition vector (g/kg waste), TC is the transfer coefficient matrix (%), $_{\text{avg}}$ is the average waste, $_{\text{burn}}$ is burnable waste, and $_{\text{inert}}$ is inert waste.

Data about the average composition of the waste input and incineration output are available from field studies. Therefore, also the average transfer coefficients are known. The waste contains an inert and a burnable fraction. In equation (2) the transfer coefficients for burnable waste (TC_{burn}) are the only unknown variables, so that they can be calculated by resolving the equation. This model is easy to apply because, after calculating the transfer coefficients once, they can be used for all types of waste input without modification [30]. However, it requires a specification of the waste input, where the waste materials are classified as inert or burnable (figure 3). Table 2 shows an example for the calculation of the output of the incineration model for the grate technology. The slag produced by grate incineration is generally of poor quality because it has a very heterogeneous composition and a relatively small share of vitrified matrix.

3.1.2. VS-Process

The set up for the VS-Process is simple because it combines two conventional system components for establishing a new 2-step technology, i.e., a conventional grate (gasification with primary air, excess air: $\lambda = 0.7$) and a conventional rotary kiln (burning with combustion air) [25]. The data from the VS-Process were taken from large-scale experiments performed at a retired incinerator in Basel (CH). The VS-Process is similar to grate incineration and therefore similar values have been observed with respect to energy efficiency and ancillary products (table 3). However, due to the 2-step set up and the higher temperatures involved, the waste is homogenized to a larger extent and more substances evaporate. Therefore, the splitting of transfer coefficients, as performed in the model for grate incineration, is not so relevant (table 2). The resulting slag is of a higher quality than from conventional grate incineration because it is more homogeneous and stable due to a larger share of glass matrix (table 3).

3.1.3. Thermoselect-Process

The Thermoselect-Process is a new, integrated high-temperature technology. Compressed waste enters a gasification reactor (reductive zone, 600°C). In a subsequent step, pure oxygen is added in order to oxidize organic compounds to CO and CO_2 (temperatures $<2000°C$ in the gas phase). The inorganic compounds melt and flow into a homogenization reactor, where pure oxygen is added [29]. The homogenization reactor needs to be heated with natural gas, part of which can be substituted with the internally produced synthetic gas. The data for the modeling was taken from a pilot plant in Italy [26,29]. Some of the advantages of this technology are the generation of recyclable products and the vitrification of slag (reusable "mineral output" with a high quality) (tables 2 and 3). The Thermoselect-Process does not need a nitrogen reduction equipment (and no NH_3 as ancillary product) due to its low NO_x emissions. On the other hand, pure oxygen and additional energy sources are needed (table 3).

Apart from the integrated new technologies, there are technology units that can be combined with other technologies to improve their environmental performance (tables 2 and 4). For instance, a thermal ash treatment can complement a conventional grate or the VS-Process. The mechanical or thermal slag treatment can only be combined with the grate technology because these processes do not work (or do not make sense) in combination with glassy slag.

3.1.4. Thermal filter ash treatment (CT-Fluapur-Process)

The thermal filter ash treatment used in this model has so far been tested only at laboratory scale. The thermal detoxification of residues is performed in a fluidized bed reactor at 900°C. A portion of the incineration gas is used as heat source and as reaction gas enriched with HCl from the wet flue gas scrubber [27] (table 4). This process has had very high separation coefficients in laboratory tests (table 2) but it is not clear yet whether a market for the separated metal chlorides exists. Therefore both recycling and landfill scenarios should be assumed.

3.1.5. Mechanical slag treatment (DryEx/DryRec-Process)

There are various technology versions on the market, of which we used data from the DryEx/DryRec-Process of ABB as a reference. This process consists of the following components: wind sifting for the separation of the fine fraction (<2 mm), magnetic Fe removal, eddy current separator for extraction of non-ferrous metals, hammer mill for separation of metals sintered to mineral material [28]. This technology separates metals from the slag in a recyclable form using relatively little electric power. This process is simple, riskless, and cheap but the fraction of recovered metals is not very high (tables 2 and 4).

3.1.6. Thermal slag treatment (HSR-Process)

In contrast to the mechanical process, the thermal slag treatment (HSR-Process) is a sophisticated procedure. The corresponding data were measured at a regular plant not yet in full operation [29]. Slag is conducted into a melting reactor (oxidation zone) heated by an oil-oxygen-burner so that the ash oxidizes and melts. The melted slag flows into a reduction reactor, which is electrically heated with graphite electrodes. The metals sediment to the bottom whereas the melted silicates flow to the sedimentation reactor heated with a gas burner in order to make the remaining metals sediment. The flue gas of all three steps is purified in the regular scrubbers of the incineration plant [29]. This process needs additional energy and ancillary products but heavy metals are separated to a larger extend than with the mechanical treatment. The remaining mineral output has a very high quality (tables 2 and 4). However, to date, it is not clear whether there exists a market for the recovered metals. Therefore, we recommend to assume two scenarios for the fate of the metal fraction: recycling and landfilling.

As can be seen above, one of the main differences between the various technologies concerns the quality and composition of solid outputs, which might have a considerable environmental impact in the long run [10]. In order to make sound technology comparisons or product assessments, it is important to model the landfill for these output products properly.

3.2. Modeling the landfills

The leachability of MSWI residues is generally recognized as the most important factor in the assessment of potential hazards associated with landfills [33–36]. Several studies have pointed to the fact that the leachate composition and quantity of emitted pollutants may vary considerably with time [37–39]. As a consequence, different approaches are reported, which deal with an assessment of future landfill emissions such as the heavy metals. The choice of the time frame represents a value choice made by the user according to the goal and scope of the study (note that, in theory, the life-cycle concept requires the consideration of all emissions without temporal cut-off [40]). This choice determines whether or not landfill emissions are regarded as an environmentally relevant problem [10,30,40]. In the following, some simple model options will be proposed predicting the mass of pollutant released in the future. These models are based on measurements at landfill sites (model option 1), leaching tests (model option 2), or leaching tests combined with probability functions for the release of pollutants from MSWI residues (model option 3).

Data available from *field measurements* (model option 1) have been used to extrapolate the emissions for short-term

Table 5

Calculation of landfill emissions to the water according to model option 1 (measurements) and model option 2 (leaching tests). The composition of the slag is multiplied with the transfer coefficients in order to calculate the emissions to the water and the neglected fraction (considered as immobilized in the landfill). The table only lists the transfer coefficients for S, Cu, and Pb (example). Slag from grate incineration was used as input [14].

Landfill model	Slag input		Transfer coefficients		Output	
	Element	Amount in slag (slag from modern grate) (g/kg waste)	Emissions to water (%)	Neglected (%)	Emissions to water (g/kg waste)	Neglected (g/kg waste)
Measurements (model option 1)	S	0.69	8.3	91.7	0.057	0.63
	Cu	1.1	0.00004	99.996	0.000044	1.1
	Pb	0.22	0.01	99.992	0.000022	0.22
Leaching tests (model option 2)	S	0.69	100.0	0.0	0.69	0.0
	Cu	1.1	45.0	55.0	0.50	0.61
	Pb	0.22	1.0	99.0	0.0022	0.22

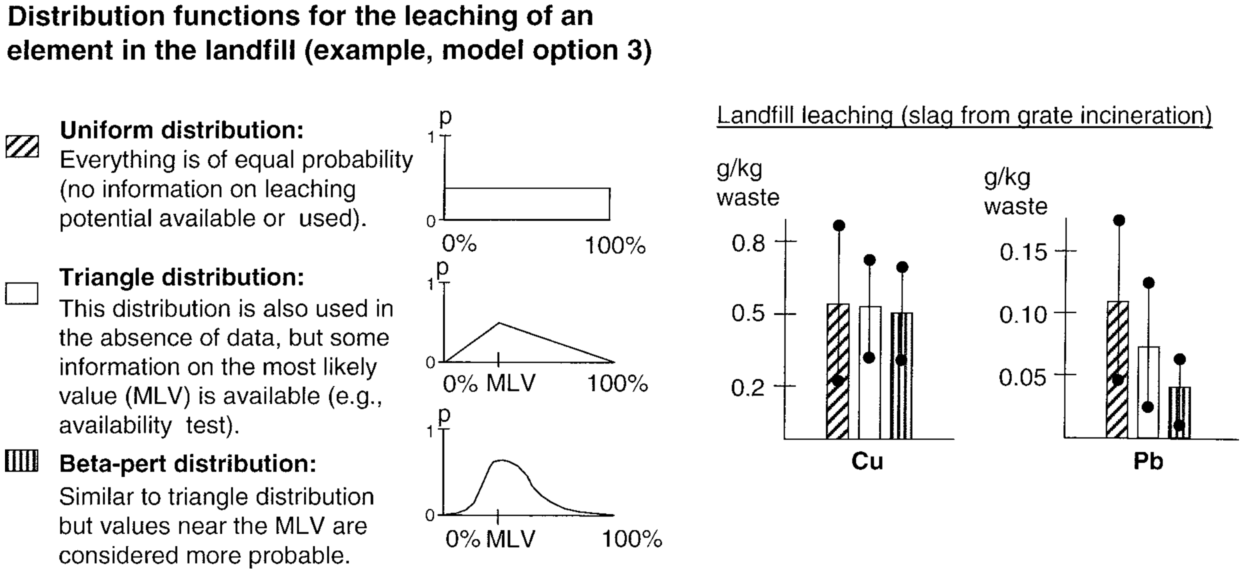


Figure 4. Emissions of Cu and Pb from a slag landfill assuming three different distribution functions for the transfer of heavy metals to the water: uniform, triangle, and beta-pert distribution (example, other distributions could be used as well). Between 0% and 100% of each landfill compound can be emitted to the environment (minimum and maximum values in the distribution). In the triangle and beta-pert distribution, the most likely value was set equal to the results of leaching tests. The bars represent the mean values of the Monte Carlo simulations. The lines indicate the 60% confidence interval.

horizons (less than 100 years). Based on these data, transfer coefficients can be constructed in analogy to the procedure applied for the incineration model. Leaching of heavy metal cations from MSWI slag is small in this time period due to the high pH value in the landfill (table 5, model option 1).

The overall emission potential of MSWI residues has also been estimated using *leaching tests* (model option 2), such as [41]. The test results were used to develop transfer coefficients [14] (table 5). Finally, by using *probability functions for the transfer coefficients of each landfill component* (model option 3) into the aqueous matrix, any emission between 0 and 100% of the landfill ingredients is possible in the future (minimum and maximum values of the distributions). From a subsequent simulation, e.g., Monte Carlo, mean values, standard deviation, etc., can be deduced. Figure 4 proposes some possible distribution functions: uniform, triangle, and beta-pert. The latter two distributions demand information on the most likely value (MLV). In our case, the MLV was set equal to the results of leaching tests. Figure 4 also shows the results of a Monte Carlo simulation using these distribution functions and taking slag of an average Swiss composition as input (1.1 g Cu and 0.22 g Pb per kg waste).

In contrast to the above models, *geochemical models* (model option 4) have been applied for a time dependent evaluation of the long-term chemical composition of MSWI residues, for instance with respect to the dissolution of important minerals/glasses [42–45]. Sundqvist et al. [16,17] have tried to model the releases of heavy metals from landfills for filter ashes as a function of time. A simple geochemical model for heavy metal release from MSWI bottom ash will be discussed here in more detail. These concepts are not commonly applicable in current LCA yet because no methods have been developed for the impact assess-

Table 6

Hypothetical initial composition of the MSWI slag and solubility constants K_{s0} used for thermodynamic calculations (temperature: $T = 25°C$; ionic strength: $I = 0$ M).

Mineral/solid phase	Wt %	Content (mol/kg slag)	Ref.	Solubility products[1]	log K_{s0}	Ref.
$CaCO_3$	3.5	0.35	[53]	$\mathbf{CaCO_3} = Ca^{2+} + CO_3^{2-}$	−8.48	[54]
$CaSO_4 \cdot 2H_2O$	1.8	0.10	[53]	$\mathbf{CaSO_4{\cdot}2H_2O} = Ca^{2+} + SO_4^{2-} + 2H_2O$	−4.58	[54]
$3(CaO)Al_2O_3 \cdot 3(CaSO_4) \cdot 32H_2O$	1.3	0.010	Assumption[2]	$\mathbf{3(CaO)Al_2O_3{\cdot}3(CaSO_4) \cdot 32H_2O} + 12H^+ = 6Ca^{2+} + 2Al^{3+} + 3SO_4^{2-} + 38H_2O$	57.5	[55]
$Ca(OH)_2$	3.7	0.50	[56][2,3]	$\mathbf{Ca(OH)_2} + 2H^+ = Ca^{2+} + 2H_2O$	22.8	[54]
$Mg(OH)_2$	2.5	0.42	Assumption[2]	$\mathbf{Mg(OH)_2} + 2H^+ = Mg^{2+} + 2H_2O$	16.8	[54]
$Cd(OH)_2$	0.00020	0.000013	Total Cd-content[4]	$\mathbf{Cd(OH)_2} + 2H^+ = Cd^{2+} + 2H_2O$	13.7	[54]
$Zn(OH)_2$	0.30	0.026	Total Zn-content[4]	$\mathbf{Zn(OH)_2} + 2H^+ = Zn^{2+} + 2H_2O$	11.5	[54]
$Cu(OH)_2$	0.90	0.087	Total Cu-content[4]	$\mathbf{Cu(OH)_2} + 2H^+ = Cu^{2+} + 2H_2O$	8.64	[54]
$Pb(OH)_2$	0.10	0.0054	Total Pb-content[4]	$\mathbf{Pb(OH)_2} + 2H^+ = Pb^{2+} + 2H_2O$	8.15	[54]
$Al(OH)_3$	10.6	1.4	Assumption[2]	$\mathbf{Al(OH)_3} + 3H^+ = Al^{3+} + 3H_2O$	8.11	[54]
$CdCO_3$	Secondary mineral[5]			$\mathbf{CdCO_3} = Cd^{2+} + CO_3^{2-}$	−12.1	[54]
$ZnCO_3$	Secondary mineral[5]			$\mathbf{ZnCO_3} = Zn^{2+} + CO_3^{2-}$	−10.0	[54]
$CuCO_3$	Secondary mineral[5]			$\mathbf{CuCO_3} = Cu^{2+} + CO_3^{2-}$	−9.63	[54]
$PbCO_3$	Secondary mineral[5]			$\mathbf{PbCO_3} = Pb^{2+} + CO_3^{2-}$	−13.1	[54]
CdO	Secondary mineral[5]			$\mathbf{CdO} + 2H^+ = Cd^{2+} + H_2O$	13.8	[54]
PbO	Secondary mineral[5]			$\mathbf{PbO} + 2H^+ = Pb^{2+} + H_2O$	12.9	[54]
ZnO	Secondary mineral[5]			$\mathbf{ZnO} + 2H^+ = Zn^{2+} + H_2O$	11.1	[54]

[1] Bold letters indicate solid phases (their solubility is quantified by K_{s0}).

[2] With the assumed values the system considers 2/3 of the total Mg and Al content of slag (in analogy to the share of considered Ca).

[3] Value for slag from Hagenholz plant.

[4] The total content was calculated with the incineration model.

[5] Mineral phase potentially formed in the geochemical simulation.

ment of emission concentrations and for a differentiated valuation of emissions occurring at different points of time. Besides precipitation and dissolution of pure solid phases, mechanisms that limit the solubility of heavy metals in the leached water are associated with the geochemistry of the solid phases present in the MSWI residue matrix. The relevant processes, such as heavy metal adsorption, complexation, cation exchange, and incorporation into solid phases as solid solutions have been investigated in detail in field and/or laboratory studies including the importance of hydrological processes [35,37–39]. A modeling of the heavy metal emissions of MSWI residues thus requires an extensive set of parameters describing the pollutant- and matrix-specific geochemical and hydrological properties with respect to the (im)mobilization processes. In our model, dissolved heavy metal concentrations are limited by dissolution/precipitation processes of the MSWI matrix as a function of geochemical factors such as solution pH and accompanying component concentrations that may be involved in complexation processes. Thus, the dependence of heavy metal dissolution from the pH-buffering capacity of the landfill matrix can be modeled by using a combination of thermodynamic equilibrium calculations and one-dimensional transport of the pore water through the solid landfill matrix. Neglecting kinetic factors in the calculation, it is generally accepted that the least soluble compound under a given set of conditions controls its aqueous concentration. The assumed initial MSWI slag concentrations and the relevant thermodynamic data are listed in table 6. The dimensions and hydrological conditions of the modeled landfill compartment are summarized in table 7. For all heavy metals, calculations of total dissolved concentrations at equilibrium with a given set of solid phases have been performed using PHREEQC [46,47]. Thermodynamic equilibria were adjusted for an ionic strength calculated from the components concentrations according to the Debye–Hueckel equation [47].

3.3. Modeling the background processes

The inventory data from background processes (production of ancillary products, infrastructure, and transportation) were taken from the Swiss data base Ecoinvent [48]. The amounts of ancillary products needed are calculated with the incineration model (product dependent allocation). The emissions and the resource use resulting from infrastructure are allocated as a total to the functional unit (process depen-

Table 7
Hydrological conditions and dimensions of the landfill compartment modeled (area-dependent values refer to a surface area of 1 dm^2).

Parameter	Base simulation	Unit
Volume	50	L
Number of boxes	1	
Height of a box	5	m
Porosity	50	%
Density of slag	1.5	t/m^3
Temperature	25	°C
Infiltration rate	400	mm/yr
$p(CO_2)$	$10^{-3.5}$	atm
Solution pH	5	

dent allocation) [14]. Default values are suggested for the transport distances, but can be modified by the user.

4. Model application in Switzerland

4.1. Incineration of various waste input materials

The Swiss Agency for the Environment, Forests and Landscape (SAEFL) used the model of Zimmermann et al. [14] for an extensive calculation of waste inventory data of packaging materials [11]. This study has been widely applied in all sorts of analysis. The results have been integrated into the software-tools Simapro [15], the most widely used LCA software in Europe. The software tools for grate incineration are publicly available [14].

4.2. Comparison of various incineration technologies

The results of a complete comparative LCA of obsolete, current and new technologies have been published elsewhere [10,13,30]. In order to illustrate our approach described above we will discuss some selected results here. The implementation of modern air pollution control measures has substantially improved the atmospheric emissions of waste incineration (table 2). If a credit is given for the energy produced in the incineration plant, the releases from the solid incineration residues cause the principal environmental impact of modern waste incinerators. New technologies have a better environmental performance than conventional technologies assuming that the results of pilot plants are representative and that long-term time horizons are considered. A combination of the thermal treatment of bottom ash and fly ash performs better than other technologies when the methods Eco-indicator 95 and Swiss Ecopoints are used [49,50]. However, a ranking of the new technologies is problematic as long as leaching data for vitrified residues and adequate landfill models are not available. The process steps incineration and landfills cause most potential impacts when the methods Swiss Ecopoints and Eco-indicator 95 are applied [49,50]. In comparison to these two process steps, the impact of infrastructure, production of ancillary products, and transport over typical average distances of 10 km in Switzerland are of minor importance. These results can be used as a decision support when the construction of a new incineration plant or renovation of an existing one is at stake.

5. Discussion and outlook

Several waste incineration models have been designed for LCI purposes in the past decade. These studies report an agreement on basic allocation procedures in the modeling of the incineration plant itself. However, this paper suggests a differentiation between coefficients for inert and burnable waste in the case of grate incineration. New integrated technologies homogenize the waste and operate at higher temperatures. Therefore the splitting of the waste input into a burnable and inert fraction is less relevant. A screening with the method Eco-indicator 95 [49] revealed that by introducing the splitting of transfer coefficients instead of using one average set of values for all materials, the overall assessment of grate incineration changed by 17% for glass, 10% for paper and 17% for a plastic mix (Eco-indicator 95 points) [51].

In the present work, much emphasis has been put on the modeling of new technologies that are not yet in operation. Although uncertainties are large, the integration of new technologies is considered to be very helpful for decision making on technology choices (see section 4.2 for some selected results). Default values have been assigned to key variables (fate of output products, energy efficiency, transport distances) representing the situation in Switzerland. Sensitivity analyses have shown that the results may depend on the value of these parameters. Transport of waste material over typical average distances of 10 km is negligible but becomes relevant when distances exceed 1,000 km [10].

A flexible approach has been chosen for the landfill model. Predictions of future long-term emissions can be made on the basis of field measurements, leaching tests or geochemical modeling. The choice of model basis and of the temporal system boundary depends on the goal and scope of each individual LCA study. As a consequence, various model options with different degrees of sophistication and temporal system boundaries are discussed.

The use of *field measurements* (model option 1) is restricted to a short-term assessment because the conditions in the landfill (e.g., pH-value of the residue matrix) are very likely to change in the future. This model option has been applied for the short-term assessment of landfill emissions, extrapolating measurement data to time horizons of 35 to 75 years (depending on the type of landfill) [14]. According to this model option, the emissions of, e.g., Cu and Pb from the slag landfill would be 0.044 mg and 0.018 mg respectively per kg of incinerated waste of an average Swiss composition (table 5). Thus, a large share of substances such as the heavy metals remains in the landfill, and their future impact would be neglected in this model.

The application of the results of *leaching test* (model option 2) suggests an estimation of the total leaching potential of a landfill residues using availability tests, for instance,

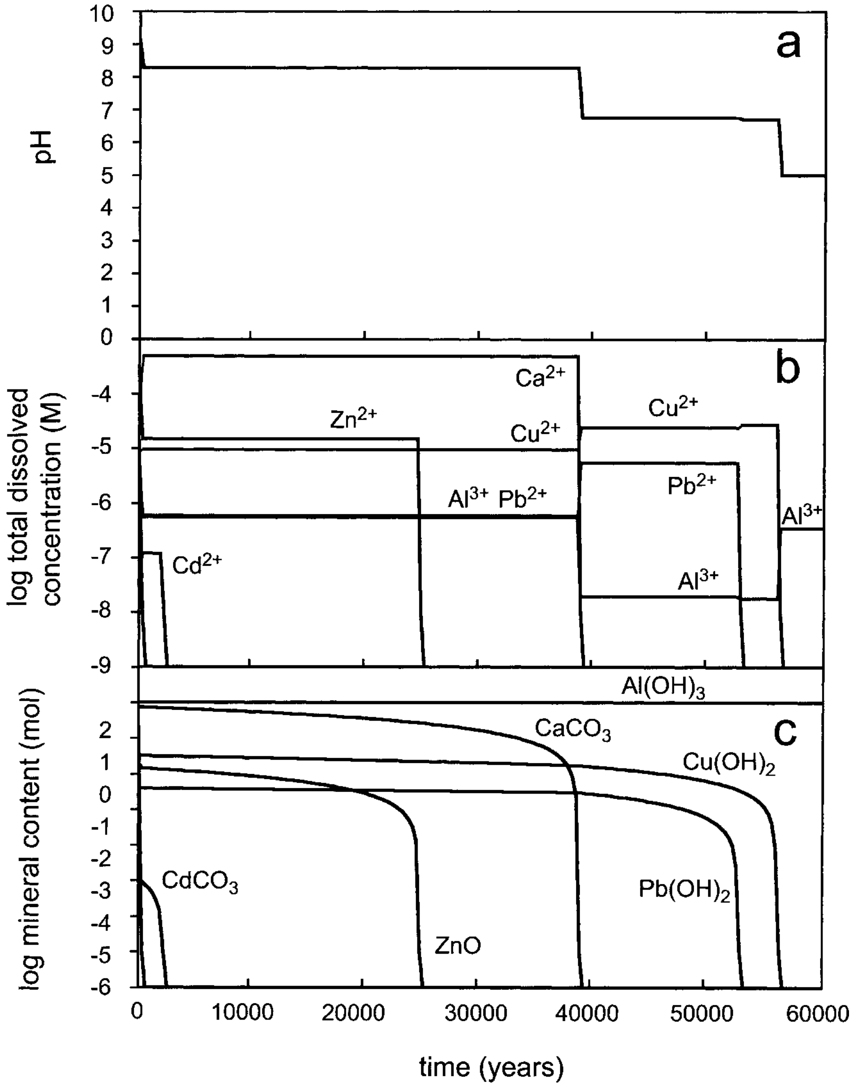


Figure 5. Results of the geochemical modeling assuming slag of an average composition given in table 6: (a) pH profile, (b) total dissolved concentration of metal species, and (c) contents of selected minerals in the simulated landfill per dm^2 surface area as a function of time (for the dimensions of the landfill see table 7).

from van der Sloot [41]. This method has been used by Zimmermann et al. [14] for the long-term assessment of landfills. Applying this model to Cu and Pb emissions (illustrative example), the total leaching potential of slag (average Swiss composition) over indefinite time would be 500 mg Cu and 2.2 mg Pb per kg of waste (table 5). These values are high in comparison to the results of the previous short-term assessment with model option 1. However, these results (model option 2) represent only a fraction of the total slag content of 1,100 mg Cu and 220 mg Pb per kilogram waste potentially leachable. This model (model option 2) is simple to apply [14] and provides some indication about what type of environmental impact may be expected. However, the leaching tests are performed over a relatively short time on laboratory scale and neglect potential geochemical changes of the landfill matrix. The use of *distribution functions* for the transfer coefficients indicating probabilities between 0% and 100% of the content in the landfill incorporates the uncertainties on the total leaching potential (model option 3). Figure 4 illustrates typical examples for the grate incineration of Cu and Pb, using the uniform, triangle and beta-pert distribution for the transfer coefficients (the most likely value was set equal to the results from leaching tests). In the case of Pb, the values are much higher than those originating from model option 2. Furthermore, the choice of the distribution function significantly influences the results. Although this approach has the advantage that no time frame needs to be explicitly chosen, the choice of the distribution function remains subjective.

All of the above landfill model options are well applicable in LCA but neglect whether low or high concentrations will be present in the landfill leachate and when the emissions will occur. In our geochemical model (model option 4), a simple set of initial conditions and minerals was defined (tables 6 and 7) in order to estimate the leaching of heavy metals due to constant infiltration of pristine water. Figure 5 shows the time course of the dissolution processes of the various metal carbonates and (hydr)oxides and the corresponding pH value in the simulated landfill. As shown in figure 5, landfill leachates are initially characterized by highly basic pH values (ranging from 8 to 11 in field measurements [34,37–39]). The mobility of heavy metals depends on the mineral phases in which the metals are bound as well as on the leachate composition and pH. For a rough estimation, trace metal cation concentrations can thus only be expected to rise to considerable concentrations when the leachate pH drops to neutral or even acidic values as the acid neutralizing capacity of the MSWI matrix decreases [33–35,39]. The concentrations of anionic metal species like metallates or metals complexed with organic ligands are significantly higher at elevated pH-values (data not shown). In the present model, changes in pH value correlate with the complete dissolution of a certain mineral phase (for instance, $CaCO_3$ after 38,500 years). The solubility of the metal carbonates and oxides is determined by their solubility constant. The time range of the release of heavy metals strongly depends on the initial mineral content and boundary conditions such as the partial pressure of CO_2.

Although the geochemical model does not include all mineral phases and processes that are relevant for the dissolution of MSWI residues, it is sensitive to different metal inputs as well as important geochemical and hydrological parameters such as the pH value and the infiltration rate of rainwater. The application of simple geochemical models is limited by the availability of input data such as leaching data of vitrified output of new technologies. Further research is currently performed by a related project of the *Integrated Project Waste* to close these data gaps [57]. The geochemical models allow a more detailed evaluation of long-term pollutant release. By using these models, risk potentials could be estimated and concepts such as discounting can be performed in sensitivity analysis. Since the base sealing and the drainage system of the landfill will fail in the long run, a time-dependent method for estimating the retardation of heavy metals in the subsoil (as a function of spatial parameters) and for assessing the fate in the groundwater is needed. Such methods have recently been published [30]. The results of a case study show that the temporal dimension of the geochemical landfill model is very important in the overall assessment of the system *waste incineration*. Therefore, we conclude that the geochemical model predicting emission concentrations as a function of time is preferable to the other model options in LCA, although it involves many uncertainties and might substantially increase the complexity of the modeling.

Acknowledgements

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Appendix

Table 8
Transfer coefficients for burnable waste: grate technology without gas purification system (electric precipitator only) [14].

	Flue gas (in %)	Slag (in %)	Boiler ash (in %)	Electric precipitator ash (in %)
H_2O	100.0	0.0	0.0	0.0
O	91.8	7.0	0.2	1.0
H	100.0	0.0	0.0	0.0
C	99.4	0.5	0.0	0.1
S	37.3	25.4	16.0	21.3
N	99.0[a]	1.0	0.0	0.0
P	5.0	88.0	2.0	5.0
Cl	82.1	1.7	1.1	15.1
Br	17.0	1.0	2.0	80.0
F	7.0	43.0	10.0	40.0
As	4.0	55.0	3.0	38.0
Ba	1.0	88.0	1.0	10.0
Cd	12.1	3.1	7.7	77.1
Co	2.0	85.0	1.0	12.0
Cr	6.4	45.8	3.2	44.6
Cu	2.5	80.3	4.9	12.3
Hg	73.1	0.6	0.2	26.2
Mn	1.0	86.0	1.0	12.0
Mo	7.0	87.0	2.1	11.0
Ni	2.5	59.4	2.5	35.5
Pb	7.4	7.2	20.4	65.0
Sb	14.6	1.6	1.8	82.0
Se	8.3	8.0	13.0	70.7
Sn	3.3	49.6	2.0	45.1
V	1.0	89.0	1.0	9.0
Zn	6.5	18.3	11.4	63.7
Si	0.2	83.5	2.3	14.0
Fe	0.3	89.7	3.3	6.7
Ca	0.2	88.2	3.3	8.3
Al	0.2	79.5	3.1	17.2
K	1.0	74.9	3.0	21.1
Mg[b]	1.0	78.0	3.0	18.0
Na	1.9	76.5	2.8	18.8

[a] Emitted as N_2 (96.6%), NO_x (3.3%), N_2O (0.096%), CN (0.047%), and NH_3 (0.0050%).

[b] Average of Na and Al.

Table 9
Transfer coefficients for burnable waste: grate technology with gas purification system (average standard in Switzerland) [14].

	Flue gas (in %)	Slag (in %)	Boiler ash (in %)	Electric precipitator ash (in %)	Wastewater (in %)	Slurry (wastewater treatment) (in %)
H_2O	100.0	0.0	0.0	0.0	0.0	0.0
O	91.7	7.0	0.2	1.0	0.0	0.1
H	100.0	0.0	0.0	0.0	0.0	0.0
C	99.4	0.5	0.0	0.1	0.0	0.0
S	6.4	25.4	16.0	21.3	26.6	4.3
N	98.9[a]	1.0	0.0	0.0	0.1	0.0
P	0.1	88.0	2.0	5.0	0.1	4.8
Cl	1.1	1.7	1.1	15.1	79.9	1.1
Br	1.0	1.0	2.0	80.0	1.0	15.0
F	1.0	43.0	10.0	40.0	1.0	5.0
As	0.1	55.0	3.0	38.0	0.0	3.9

Table 9
(Continued.)

	Flue gas (in %)	Slag (in %)	Boiler ash (in %)	Electric precipitator ash (in %)	Wastewater (in %)	Slurry (wastewater treatment) (in %)
Ba	0.1	88.0	1.0	10.0	0.0	0.9
Cd	1.1	3.1	7.7	77.0	0.1	11.0
Co	0.0	85.0	1.0	12.0	0.0	2.0
Cr	0.0	45.5	3.2	44.6	0.3	6.4
Cu	0.2	79.8	4.9	12.3	0.2	2.5
Hg	10.5	0.6	0.2	26.2	1.0	61.5
Mn	0.0	86.0	1.0	12.0	0.0	1.0
Mo	0.2	87.0	2.1	11.0	0.0	0.0
Ni	0.0	59.1	2.5	35.5	0.3	2.5
Pb	0.2	7.0	20.4	65.0	0.0	7.4
Sb	0.2	1.6	1.8	82.0	0.0	14.4
Se	1.2	8.0	13.0	70.7	0.0	7.1
Sn	0.1	49.6	2.0	45.1	0.0	3.2
V	0.0	89.0	1.0	9.0	0.0	1.0
Zn	0.3	18.0	11.4	63.7	0.0	6.5
Si	0.0	83.4	2.3	14.0	0.0	0.2
Fe	0.0	89.6	3.3	6.7	0.0	0.3
Ca	0.0	88.1	3.3	8.3	0.0	0.2
Al	0.0	79.5	3.1	17.2	0.0	0.2
K	0.0	74.9	3.0	21.1	0.2	0.8
Mg[b]	0.0	78.0	3.0	18.0	0.7	0.3
Na	0.0	76.4	2.8	18.8	1.4	0.5

[a] Emitted as N_2 (99.3%), NO_x (0.55%), N_2O (0.12%), CN (0.0052%), and NH_3 (0.075%).

[b] Average of Na and Al.

Table 10
Transfer coefficients for burnable waste: grate technology with gas purification system (modern standard 1998) [1–3].

	Flue gas (in %)	Slag[a] (in %)	Boiler ash (in %)	Electric precipitator ash (in %)	Wastewater (in %)	Slurry (wastewater treatment) (in %)
H_2O	100.0	0.0	0.0	0.0	0.0	0.0
O	91.7	7.0	0.2	1.0	0.0	0.1
H	100.0	0.0	0.0	0.0	0.0	0.0
C	98.9	0.8	0.0	0.3	0.0	0.0
S	0.2	55.4	0.0	29.8	7.1	7.5
N	98.9[b]	1.0	0.0	0.0	0.1	0.0
P[c]	0.1	88.0	2.5	9.4	0.0	0.0
Cl	0.0	7.1	0.0	1.3	90.9	0.6
Br[c]	0.3	11.0	1.0	87.7	0.0	0.0
F	0.1	61.5	0.0	30.8	5.6	2.1
As[c]	0.0	55.0	3.0	38.1	0.0	3.9
Ba[c]	0.1	88.7	2.2	9.0	0.0	0.0
Cd	0.0	0.3	0.0	36.9	0.0	62.7
Co[c]	0.0	85.0	1.0	12.0	0.0	2.0
Cr[c]	0.0	45.5	3.2	44.6	0.3	6.4
Cu	0.0	80.1	0.0	18.5	0.0	1.5
Hg	0.0	0.6	0.2	36.6	1.0	61.5
Mn[c]	0.0	86.0	1.0	12.0	0.0	1.0
Mo[c]	0.2	86.7	2.1	11.0	0.0	0.0
Ni[c]	0.0	90.1	3.6	6.3	0.0	0.0
Pb	0.0	6.6	0.0	84.8	0.0	8.5
Sb	0.0	1.6	1.8	82.2	0.0	14.4
Se	0.0	8.0	13.0	71.9	0.0	7.1
Sn[c]	0.1	49.6	2.0	45.1	0.0	3.2
V	0.0	89.0	1.0	9.0	0.0	1.0
Zn	0.0	0.3	0.0	81.7	0.0	18.0
Si[c]	0.2	91.9	2.6	5.4	0.0	0.0
Fe	0.0	89.9	0.0	9.3	0.0	0.7
Ca[c]	0.2	86.2	3.7	10.0	0.0	0.0
Al[c]	0.2	85.3	3.6	10.9	0.0	0.0
K[c]	0.3	66.8	4.7	28.1	0.0	0.0
Mg[c,d]	0.1	91.7	2.1	6.1	0.0	0.0
Na[c]	0.9	61.4	9.4	28.2	0.0	0.0

[a] The transfer coefficients for slag are not identical to those in tables 8 and 9, because different plants were used as data sources.

[b] Emitted as N_2 (99.3%), NO_x (0.55%), N_2O (0.12%), CN (0.0052%), and NH_3 (0.075%).

[c] The transfer coefficients for P, Br, As, Ba, Co, Cr, Mn, Mo, Ni, Sn, Si, Ca, Al, K, Mg, and Na were calculated using the data from a plant, where the wet scrubber liquor is injected into a spray dryer [3]. Therefore, no wastewater and slurry are produced as output products.

[d] Average of Na and Al.

Table 11
Transfer coefficients for the VS-Process [4,5].

	Flue gas (in %)	Slag (in %)	Boiler ash (in %)	Electric precipitator ash (in %)	Wastewater (in %)	Slurry (wastewater treatment) (in %)
H_2O	100.0	0.0	0.0	0.0	0.0	0.0
O	72.6	26.3	0.2	0.8	0.0	0.1
H	100.0	0.0	0.0	0.0	0.0	0.0
C	99.7	0.0	0.0	0.3	0.0	0.0
S	0.2	58.1	0.0	28.0	6.7	7.0
N	98.9[a]	1.0	0.0	0.0	0.1	0.0
P	0.1	88.0	2.5	9.4	0.0	0.0
Cl	0.0	14.0	0.0	1.2	84.2	0.6
Br	0.3	11.0	1.0	87.7	0.0	0.0
F	0.1	61.5	0.0	30.8	5.6	2.1
As	0.0	55.0	3.0	38.1	0.0	3.9
Ba	0.1	88.7	2.2	9.0	0.0	0.0
Cd	0.0	0.2	0.0	42.8	0.0	57.0
Co	0.0	85.0	1.0	12.0	0.0	2.0
Cr	0.0	75.0	1.0	21.9	0.1	2.0
Cu	0.0	87.0	0.0	12.4	0.0	0.6
Hg	0.0	5.0	0.2	35.0	1.0	58.8
Mn	0.0	86.0	1.0	12.0	0.0	1.0
Mo	0.2	86.7	2.1	11.0	0.0	0.0
Ni	0.0	76.0	1.0	22.6	0.0	1.0
Pb	0.0	8.0	0.0	87.4	0.0	4.6
Sb	0.0	46.0	1.0	45.1	0.0	7.9
Se	0.0	22.0	11.0	61.0	0.0	6.0
Sn	0.1	62.0	1.5	34.0	0.0	2.4
V	0.0	89.0	1.0	9.0	0.0	1.0
Zn	0.0	18.0	0.0	71.0	0.0	11.0
Si	0.1	96.5	1.1	2.3	0.0	0.0
Fe	0.0	97.0	0.0	2.8	0.0	0.2
Ca	0.1	91.7	2.2	6.0	0.0	0.0
Al	0.1	90.6	2.3	7.0	0.0	0.0
K	0.3	67.0	4.7	28.0	0.0	0.0
Mg[b]	0.1	94.0	1.5	4.4	0.0	0.0
Na	0.2	91.8	2.0	6.0	0.0	0.0

[a] Emitted as N_2 (99.3%), NO_x (0.55%), N_2O (0.12%), CN (0.0052%), and NH_3 (0.075%).

[b] Average of Na and Al.

Table 12
Transfer coefficients for the Thermoselect-Process [6,7].

	Flue gas (in %)	Mineral residues (in %)	Metals (in %)	Sulfur product (in %)	Salts (in %)	Heavy metal concentrate (in %)	Condensate (in %)
H_2O	9.0	0.0	0.0	0.2	0.0	0.0	90.8
O	90.0	9.0	0.0	0.0	0.0	0.0	0.1
H	100.0	0.0	0.0	0.0	0.0	0.0	0.0
C	99.6	0.2	0.0	0.0	0.1	0.1	0.0
S[a]	1.0	18.0	7.4	65.4	3.5	0.0	4.7
N[b]	99.7	0.0	0.0	0.0	0.1	0.0	0.2
P	0.0	97.8	2.2	0.0	0.0	0.0	0.0
B	0.0	100.0	0.0	0.0	0.0	0.0	0.0
Cl	0.0	0.9	0.0	0.5	96.4	0.8	1.3
Br	0.0	0.0	0.0	12.2	13.6	52.5	21.8
F	0.1	0.0	0.0	0.5	6.0	93.2	0.1
J	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ag	0.0	0.0	0.0	0.0	0.0	0.0	0.0
As	0.3	22.2	8.5	6.2	2.4	60.1	0.5
Ba	0.0	100.0	0.0	0.0	0.0	0.0	0.0
Cd	0.1	3.3	0.1	13.1	11.1	72.4	0.1
Co	0.0	55.1	42.1	0.0	0.2	2.6	0.0
Cr[c]	0.0	120.0	18.0	0.1	0.0	1.3	0.0
Cu	0.0	19.0	78.0	0.4	0.0	2.6	0.0
Hg	1.4	18.2	0.0	20.6	0.1	59.8	0.1
Mn	0.0	97.2	1.7	0.0	0.0	1.0	0.0
Mo	0.0	19.0	78.0	0.4	0.0	2.6	0.0
Ni	0.0	50.2	43.5	0.4	0.2	5.8	0.0
Pb	0.0	17.0	4.1	12.7	1.9	64.3	0.0
Sb	0.0	73.9	13.5	0.0	0.0	12.7	0.0
Se	0.0	68.5	1.1	0.0	0.0	30.4	0.0
Sn	0.0	26.2	10.6	4.3	0.2	58.8	0.0
V	0.0	92.1	7.0	0.0	0.2	0.7	0.0
Zn	0.0	13.0	2.4	5.1	0.0	79.4	0.1
Si	0.0	100.0	0.0	0.0	0.0	0.0	0.0
Fe	0.0	45.0	54.0	0.1	0.0	0.9	0.0
Ca	0.0	99.4	0.1	0.0	0.4	0.0	0.0
Al	0.0	99.7	0.2	0.1	0.0	0.0	0.0
K	0.0	96.0	0.1	0.2	3.6	0.0	0.0
Mg	0.0	99.9	0.0	0.0	0.0	0.0	0.0
Na	0.0	85.6	0.1	0.4	14.0	0.0	0.0

[a] Emitted as SO_x (80%) and H_2S (20%).

[b] Emitted as N_2 (99.8%), NO_x (0.20%), and CN (0.0014%).

[c] The sum of transfer coefficients for Cr is higher than 100% because additional Cr from the fire protection material of the high-temperature reactor enters the process [7].

Table 13
Transfer coefficients for the thermal slag treatment technology (HSR) ([6], assumptions).

	Flue gas[a] (in %)	Metals (in %)	Mineral material (in %)
O	0.0	0.0	100.0
C	99.9	0.1	0.0
S	62.5	27.1	10.4
N	100.0	0.0	0.0
P	0.0	8.1	91.9
Cl	92.0	0.0	8.0
Br	92.0	0.0	8.0
F	92.0	0.0	8.0
J	92.0	0.0	8.0
As	85.1	0.0	14.9
Ba	0.0	0.0	100.0
Cd	99.7	0.0	0.3
Co	1.0	68.9	30.2
Cr	0.5	18.4	81.1
Cu	2.1	88.3	9.6
Hg	100.0	0.0	0.0
Mn	0.0	0.0	100.0
Ni	0.9	75.5	23.6
Pb	97.0	1.3	1.6
Sb	100.0	0.0	0.0
Se	62.5	27.1	10.4
Sn	5.3	91.8	2.9
V	0.5	18.4	81.1
Zn	87.1	2.4	10.5
Si	0.0	0.0	100.0
Fe	1.0	62.2	36.7
Ca	0.0	0.0	100.0
Al	0.0	0.0	100.0
K	0.0	0.0	100.0
Mg	0.0	0.0	100.0
Na	0.0	0.0	100.0

[a] This flue gas will be cleaned in the gas purification system of the incineration plant.

Table 14
Transfer coefficients for the thermal filter ash treatment (CT-Fluapur) [8,9]. If no measurements were available, the transfer coefficients to the mineral residues were set 100%.

	Heavy metal concentrate (in %)	Mineral residues (in %)
Cl	100.0	0.0
Br	0.0	100.0
F	0.0	100.0
As	0.0	100.0
Ba	0.0	100.0
Cd	99.9	0.1
Co	0.0	100.0
Cr	39.3	60.7
Cu	99.4	0.6
Hg	0.0	100.0
Mn	0.0	100.0
Mo	0.0	100.0
Ni	57.6	42.4
Pb	99.0	1.0
Sb	0.0	100.0
Se	0.0	100.0
Sn	0.0	100.0
V	0.0	100.0
Zn	99.1	0.9
Si	0.0	100.0
Fe	0.0	100.0
Ca	0.0	100.0
Al	0.0	100.0
K	100.0	0.0
Mg	0.0	100.0
Na	100.0	0.0

Table 15
Transfer coefficients for the slag landfills according to model option 1 (measurements) and 2 (leaching tests and estimations) [1].

	Model option 1: short-term emissions to water (in %)	Model option 2: long-term emissions from the slag of grate incineration to water (in %)	Model option 2: long-term emissions from the vitrified slag of the Thermoselect, VS, or HSR processes to water (in %)
H_2O	100.0	100.0	100.0
C	0.3	5.0	0.0

Table 15
(Continued.)

	Model option 1: short-term emissions to water (in %)	Model option 2: long-term emissions from the slag of grate incineration to water (in %)	Model option 2: long-term emissions from the vitrified slag of the Thermoselect, VS, or HSR processes to water (in %)
S	8.3	100.0	100.0
N	22.0	100.0	100.0
Cl	94.6	100.0	100.0
Cd	0.4	66.0	6.6
Cr	0.0	7.0	0.7
Cu	0.0	45.0	4.5
Hg	3.3	60.0	6.0
Mo	6.4	100.0[a]	10.0
Ni	0.1	0.1	0.0
Pb	0.0	1.0	0.1
Sb	0.1	100.0[a]	10.0
V	0.1	0.1	0.0
Zn	0.0	68.0	6.8
Fe	0.0	7.0	0.7
Ca	0.6	90.0	9.0

[a] No leaching tests were available for heavy metal anions. It was assumed that all anions are leachable due to the basic pH-value in the landfill and the very small anion exchange capacity of slag [10].

Table 16
Transfer coefficients for the landfills for other incineration residues according to model option 1 (measurements) and 2 (leaching tests and estimations) [1].

	Model option 1: short-term emissions to water (in %)	Model option 2: long-term emissions to water (in %)
H_2O	100.0	100.0
C	0.6	100.0
S	2.0	100.0
Cl	27.4	100.0
F	2.1	100.0
Cd	0.0	87.5
Cr	0.1	7.0
Cu	0.0	44.5
Hg	0.0	60.0
Mo	0.0	100.0[a]
Pb	0.0	23.0
Sb	0.0	100.0[a]
Zn	0.0	84.0
Fe	0.0	7.0
Ca	0.0	90.0

[a] No leaching tests were available for heavy metal anions. It was assumed that all anions are leachable due to the basic pH-value in the landfill.

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