

## Part II

# Waste Incineration

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## Summary

The aim of this report is to create life cycle inventories of waste disposal in municipal solid waste incinerators (MSWI) and hazardous waste incinerators (HWI). The inventories are based on technology used in Switzerland. The inventories are waste-specific, i.e. the composition of a specific waste under consideration influences the resulting emissions and expenditures. For example direct cadmium emissions are only present in inventories for cadmium-bearing wastes. To perform this complicated multi-input allocation, Excel calculation tools are created, which calculate the inventory depending on a user-definable waste composition. The functional unit of waste incineration is kilogram waste.

The waste composition is defined in terms of content of chemical elements (C, H, O, S, N etc.). A vector of 41 chemical elements plus water content and heating values is considered. Energy production is also calculated waste-specifically. Energy generation is treated as a by-product of waste incineration. The used default allocation is 0% on energy production and 100% on waste incineration service, based on proceeds. The fate of waste compounds during incineration is modelled using transfer coefficients for chemical elements, derived from measurements. The solid residues generated in incineration are landfilled in bottom ash landfills (slag compartments) and residual material landfills. Waste-specific compositions of solid incineration residues are calculated from the incineration model and are used as input to the landfill models described in part III. The waste incineration unit process modules contain all expenditures from incineration and landfilling.

Burdens from landfilling of solid incineration residues are significant and must be included in assessing waste incineration. However, valuation methods that heed toxic pollutants must be applied. In most waste disposal processes the energy consumption is of minor importance and not indicative of the total burdens. Energy or resources scores like cumulative energy demand CED can therefore *not* be used to estimate disposal burdens. Most burdens come from the waste material itself, be it CO<sub>2</sub> emissions from incineration of burnable material or metal emissions from landfilling of incineration residues.

Different chemical elements show different behaviour during incineration. A coarse clustering of behaviour typologies produces three principal types: 1.) Elements transferred largely to the purified incinerator exhaust, 2.) Elements transferred largely to the purified incinerator effluent, 3.) Elements transferred largely solid outputs of the incinerator. Variabilities within the clusters exist and are modelled with said transfer coefficients.

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# 1 Introduction

This report describes the life cycle inventories of waste incineration. Two types of incinerators are distinguished: municipal solid waste incinerators (MSWI) and hazardous waste incinerators (HWI). An overview of the municipal and industrial waste flows in Switzerland and Europe is given in part I.

The goal of this report is to describe *waste-specific* inventories, i.e. not only the disposal emissions per kilogram average waste are described, but tools are devised which calculate life cycle inventories (unit process data) for different waste fractions and different waste compositions. Waste compositions are heeded on th level of different chemical elements (e.g. lead, zinc, copper nitrogen.), but not as compounds (e.g. dioxins). The calculation tools are realised as MS Excel workbooks, which are available to the ecoinvent user. This report describes the principal calculations performed by the tools. The waste disposal inventories present in the ecoinvent database are generated using the calculation tools and the waste compositions described in part I of the report. The user can generate *additional inventories* for specific wastes, by entering a waste composition and waste characteristics. The necessary parameters are described in chapter 2.4 'Necessary waste data' on page 14.

From incineration various unburnable residues result, which are landfilled. The residues are calculated waste-specifically and are used as an input to the landfill models described in part III. The life cycle inventories of waste incineration contain the emissions from the incinerator *and* from the landfills (see chapter 2.2 'System boundaries' on page 9). The calculation tools for incineration and for landfills are linked as to generate these inventories automatically, without further user engagement.

The inventories are based on the technology encountered in Switzerland, but can be used as a good proxy for modern waste incineration in Europe.

## 2 Systems Characterisation

### 2.1 Process description

#### 2.1.1 Municipal solid waste incinerator

##### Incineration capacities and incinerated waste

Switzerland has 28 municipal solid waste incinerator plants in operation<sup>1</sup> (2000). The majority of them (22) have two or three furnace lines. Each furnace has a designed *thermal* capacity. The waste mass that can be treated by the incinerator depends on the heating value of the waste. In recent years the heating value of municipal waste has increased due to recycling efforts for glass, metals, and kitchen wastes. The relative share of wastes with high heating value like plastics, which are as yet rarely recycled in a household context, has increased. Accordingly, the *mass* capacity of built MSWIs has decreased<sup>2</sup>. Currently, the heating value of average Swiss municipal waste is at 12.42 MJ/kg, ranging from 10.4 to 14 MJ/kg for individual plants (BUWAL 2001h). The total installed thermal capacity of the 28 MSWI plants in Switzerland is 1'336 MW. Assuming a maximum operation time of 8000 hours per year, and a heating value of waste of 12.42 MJ/kg, the total theoretical mass capacity is 3'098 kt waste per year. Due to repairs and operation restrictions the effective mass capacity was 2'826 kt/a (91% of the theoretical capacity). The waste incinerated in 2000 was 2.8 Mio. tons per year, which is 90% of the theoretical capacity and 99% of the effective capacity.

Tab. 2.1 Waste in Swiss MSWI in 2000 (BUWAL 2001h)

Waste material	kt/a	w-%
Inland MSW	2'400	82.5%
Burnable construction waste	342	11.8%
Sorted waste wood	14	0.5%
Wastewater treatment sludge	42	1.4%
Hazardous waste	50	1.7%
Imports of waste <sup>1</sup>	49	1.7%
Other wastes	11	0.4%
Total	2'800	100%

<sup>1</sup> mainly from Germany

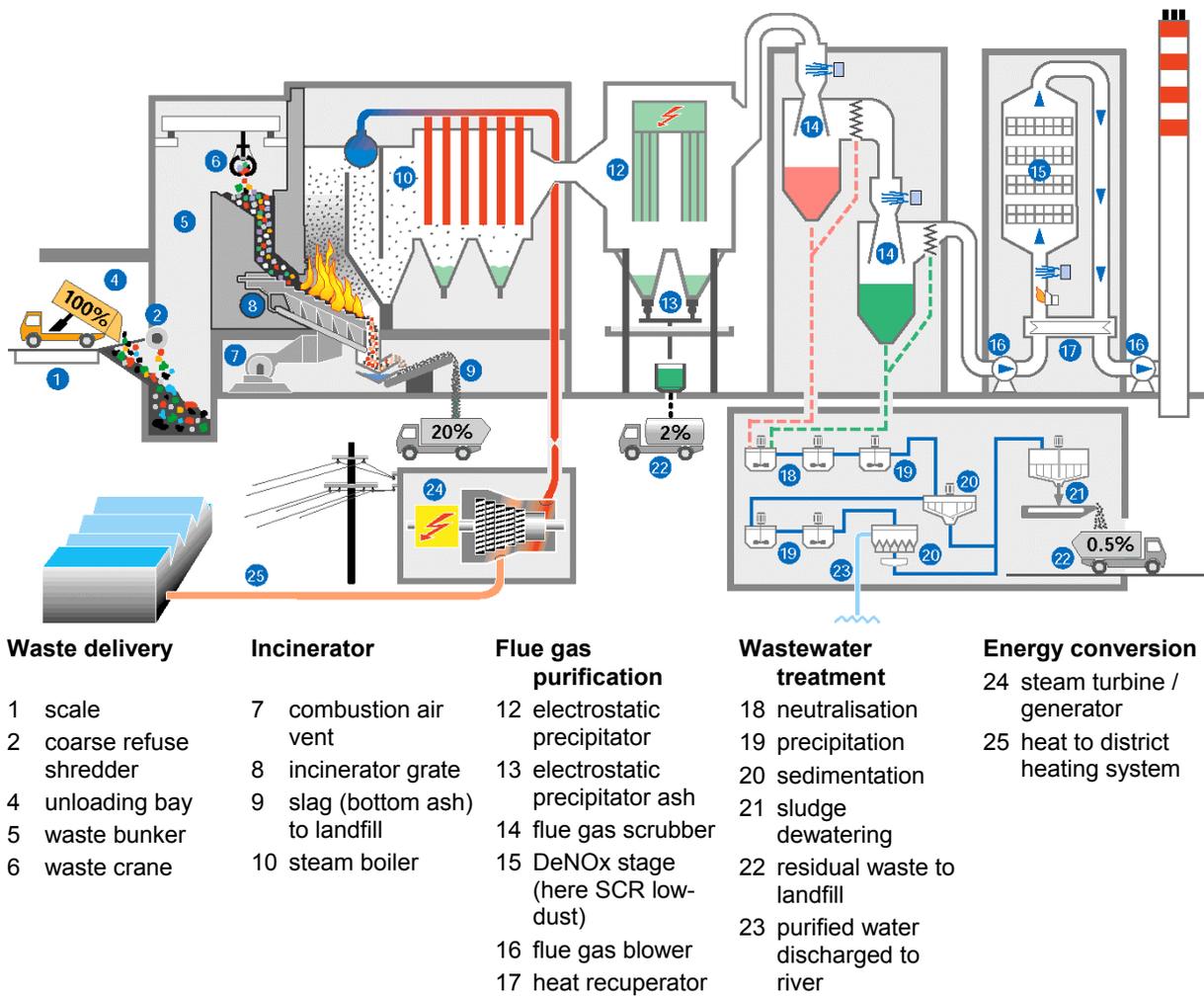
#### Typical design and process

Fig. 2.1 shows the operation principle of an MSWI. The number in brackets in the following text refer to the numbers in that figure. The typical design for a MSWI plant consists of two or three incineration lines in parallel. Each incineration line is equipped with a grate-type furnace (8). At the end of the grate the unburnable remains are collected as slag (bottom ash) and quenched in water (9). The raw gas is led to an integrated steam boiler (10). The recovered heat is passed to a steam turbine (24) to

<sup>1</sup> One new plant in the Canton of Fribourg (Châtillon/Posieux) is presently (2002) under construction. Plans exist for new plants in Thun BE, Tessin TI, and Lausanne II LA.

<sup>2</sup> So it can be stated that recycling of e.g. glass diminishes the waste processing capacities of MSWI, because it increases the heating value of the remaining waste. This fact is sometimes misinterpreted to the effect that MSWI operators would rather diminish glass recycling and therefore keep waste processing capacities of their plants high. High shares of unburnable wastes increase the amount of bottom slag generated, which is expensive for MSWI operators to dispose. MSWI operators are interested in keeping operating costs low and therefore prefer burnable wastes over inert, slag-destined wastes. On the other hand, glass recycling indeed diminishes the waste masses that need to be disposed of, while the total energy in waste remains the same. Hence, glass recycling might decrease the mass capacity of MSWIs (diminishing revenues to a MSWI usually calculated on a weight basis), but the thermal capacity remains constant.

generate electricity. The expanded steam is sometimes directed to a district heating network (25) or used as process steam for neighbouring industries.



**Fig. 2.1** Scheme of a typical Swiss municipal solid waste incinerator. Here MSWI Buchs AG with a DeNOx installation after the wet scrubber (SCR low-dust). The figures on the trucks give the approximate weight of solid output products, related to 100% waste input

After being cooled down in the steam boiler, the flue gas of the MSWI is then passed into an electrostatic precipitator for fly ash separation (12). Electrostatic precipitators (ESP) use the principle of electrostatic attraction to remove particles from the raw gas. They consist of rows of discharge electrodes (wires or thin metal rods), through which a high voltage is applied, and which run between an array of parallel rows of metal plates which collect the charged particles.

After the ESP, a multistage wet scrubber (14) is used to eliminate harmful components of the flue gas like SO<sub>x</sub>, HCl by washing the raw gas in a reaction tower. Designed to provide a high gas-liquid contact, the gases are cooled by water sprays in the first stage, removing HCl, HF, some particulates and some heavy metals. In the second stage calcium hydroxide or another suitable alkali is used to remove SO<sub>x</sub> and any remaining HCl.

The scrubbing liquid is neutralised (18), heavy metals are precipitated (19) and separated as a sludge (20) in a wastewater treatment facility. The treated water is usually discharged to a river. After the wet scrubber the purified flue gas enters a DeNOx installation<sup>3</sup> (15). Usually SCR or SNCR-DeNOx technology is employed (explained below). The purified flue gas is led into a stack. Approximately 75% of the original waste mass is transferred to gaseous compounds like carbon dioxide CO<sub>2</sub>, elemental Nitrogen N<sub>2</sub> and water H<sub>2</sub>O and minor trace gases.

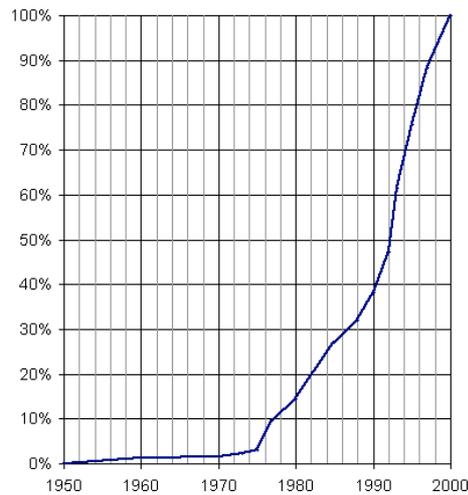


**Fig. 2.2 Waste bunker (5) and waste crane (6) of MSWI Turgi**

Fig. 2.3 shows the age distribution of Swiss MSWI plants.

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<sup>3</sup> Placement of the DeNOx facility depends on the technology employed: SNCR DeNOx takes place directly in the incineration chamber, SCR-high dust before the wet scrubber (i.e. in a high-dust environment), SCR-low dust after the wet scrubber (i.e. in a low-dust environment).



**Fig. 2.3** Age distribution of Swiss MSW incinerator lines by incineration capacity and by year of construction or complete makeover (based on data from BUWAL 2000). For example 40% of the installed MSWI capacity is older than 1990

### Differences in technology

All of the 28 Swiss MSWI feature an electrostatic precipitator for fly ash. Substantially all Swiss plants also feature a wet flue gas scrubber system<sup>4</sup>. According to (BUWAL 2000) 56.8% of the MSW in Switzerland was burned in MSWIs with a SCR facility<sup>5</sup>, 29.4% of MSW with a SNCR, and 13.8% of MSW had no DeNOx stage (in 4 plants)<sup>6</sup>.

<sup>4</sup> Only the smallest Swiss MSWI (Zermatt), which burns 0.2% of the total MSW in Switzerland, has no flue gas scrubber system at all. The MSWI St. Gallen has two incinerator lines with a semi-dry flue gas cleaning system. One of the two lines in MSWI Josefstrasse ZH also has a semi-dry flue gas cleaning system. All other MSWI plants have a wet flue gas scrubber system. Only 5% of the Swiss MSW is incinerated in plants with a semi-dry flue gas cleaning system. In a semi-dry flue gas cleaning system flue gas acids are absorbed with a limestone slurry.

<sup>5</sup> It is assumed that 32.2% are SCR high-dust, and 24.6% SCR low-dust facilities, based on data for 1995 (Zimmermann et al. 1996:B.86).

<sup>6</sup> These figures are not given in percentages of *number of plants*, but in percentages of *processed waste*. The figure for 'no DeNOx stage' is largely influenced by one single plant without DeNOx stage, that burns 10.5% of all MSW in Switzerland (Les Cheneviers GE).

Tab. 2.2 Main characteristics of different DeNOx technologies

	<b>SNCR</b>	<b>SCR-high dust</b>	<b>SCR-low dust</b>
<b>Reducing agent</b>	Ammonia NH <sub>3</sub> (also urea (NH <sub>2</sub> ) <sub>2</sub> CO could be used)	Ammonia	Ammonia
<b>Reduction takes place....</b>	...directly in the incineration chamber	...before the wet scrubber (i.e. in a high-dust environment)	...after the wet scrubber (i.e. in a low-dust environment)
<b>Catalyst</b>	–	V-Ti catalyst	V-Ti catalyst
<b>Catalyst lifetime</b>	–	reduced from high dust operation environment	normal lifetime of approximately 3 years
<b>Reaction temperature</b>	800 –1000°C	250 – 300°C	250 –300°C
<b>Ancillary energy</b>	–	–	natural gas burner

### Recycling efforts

In Switzerland, 13 of 28 MSWI plants use magnetic separators to extract ferrous scrap metal from raw slag. Slag scrap is a low value product and the main motivation is to reduce the expenses from slag landfilling. Approximately 50% of the MSW is burnt in MSWI with scrap separation. The amount of removed scrap varies considerably between 4 kg and 150 kg scrap per ton of slag, which probably depends on the quality requirements of the scrap recipient. The weighted average value for the plants that descrap is 50 kg scrap per ton of slag (BUWAL 2001i). Hence, in the *average* Swiss MSWI plant mix 26 kg scrap per ton of slag is separated. Slag is not used for road construction.

In 2000 three plants (Bern BE, Niederurnen GL, and Zuchwil SO) all or part of the scrubber sludge is treated to concentrate the contained metals for recycling. Two plants (Bern BE, Niederurnen GL) also wash their electrostatic precipitator ashes and separate heavy metals for recycling and are able to dispose the remains together with the slag (bottom ash).

### Energy production

All Swiss MSWI utilise the energy contained in the waste to produce useful heat and/or electricity. In 2000, Swiss MSWIs gross production was 2'526'800 MWh heat and 1'284'200 MWh electricity (BUWAL 2001f). The total energy input was 9'880'262 MWh<sup>7</sup>. The gross heat generation efficiency from waste is 25.6% and the gross electricity generation efficiency is 13% (BUWAL 2001f). The GSD for these two efficiency factors was assumed to be small (101%). All plants consume some of the generated energy and sell the remaining energy. Net heat can be used in district heating networks or as industrial steam, depending on local circumstances. If no suitable heat consumer is available, electricity production can be increased, as grid feed of electricity is simpler. Accordingly a tendency can be observed that MSWI plants tend to produce more electricity instead of heat, as shown in Fig. 2.4. There are more plants that produce little or no heat, than there are plants producing little or no electricity. The small plant Zermatt at the lower left only produces heat for self-consumption. The MSWI Lausanne produces heat for an district heating grid, but no electricity.

<sup>7</sup> This includes 1.8% energy from auxiliary boilers.

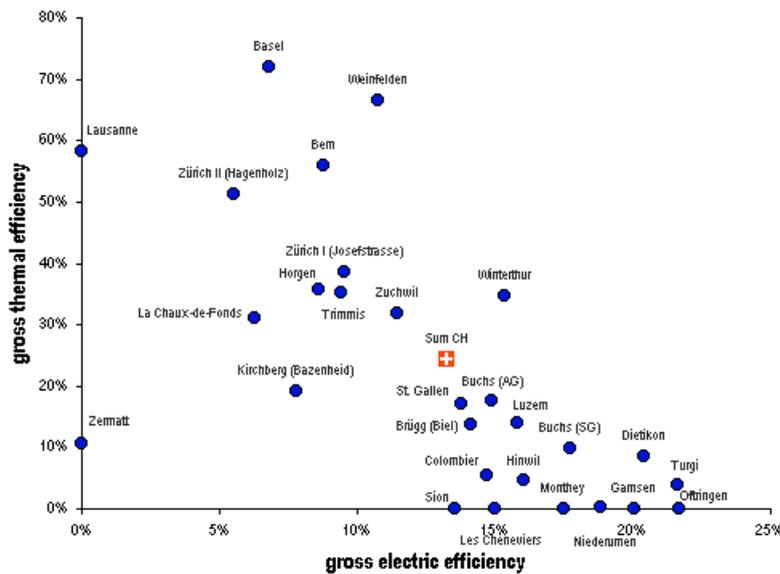


Fig. 2.4 Plot of gross thermal efficiency versus gross electric efficiency in 28 Swiss MSWI plants in the year 2000

## 2.1.2 Hazardous waste incinerator

### Incineration capacities and incinerated waste

In 2001 Switzerland had 13 HWI plants in operation. Most of these plants are operated by the chemical/pharmaceutical industry. The total capacity of the 13 plants is 136'700 tonnes per year; with individual plant capacities ranging from 200 (Knoll, Liestal) to 37'000 t/a (Novartis, Schweizerhalle). The total hazardous waste incinerated in 1999 was 366'000 tons (BUWAL 2001a). HWIs incinerated about 37% of all incinerated hazardous waste. Additionally approx. 50'000 t/a or 14% were incinerated in MSWIs (BUWAL 2001b) and 65'000 t/a or 18% in cement kilns (BUWAL 2001c). The remainder of approximately 114'000 t/a or 31% was disposed in industrial furnaces<sup>8</sup>.

<sup>8</sup> No detailed data is available for industrial furnaces. BUWAL (2001g) gives a total figure of 240'000 t/a of waste fuels in industrial furnaces for 1999, which includes non-hazardous wastes such as waste wood and wastewater treatment sludge.

Tab. 2.3 Capacity of Swiss HWIs sorted by size (BUWAL 1999b , BUWAL 2001k)

	Capacity t/a
HWI Schweizerhalle, Valorec/Novartis AG	37'000
HWI Visp, Lonza AG	24'000
HWI Basel, Valorec/Novartis AG	16'000
HWI Chenevier, Département de l'Intérieur, de l'Environnement et des Affaires Régionales (DIER)	14'000
HWI Muttenz, CLARIANT AG	13'000
HWI Monthey CIMO, Compagnie Industrielle de Monthey SA	10'000
HWI Dottikon, EMS Chemie AG	7500
HWI Sisseln, Roche AG	7300
HWI Lupfig, Amstutz Altöl AG, Obfelden	2500
HWI Pratteln, Schenectady AG	2500
HWI Basel, F. Hoffmann La Roche AG	2000
HWI Pratteln, Rohner AG	700
HWI Liestal, Knoll AG	200
Total	136'700

Tab. 2.4 Incineration of hazardous waste in Switzerland 1999

Disposal	Annual mass, (1999) t/a	w-%	Source
Hazardous waste incinerators	136'700	37%	BUWAL 2001a
Municipal solid waste incinerators	50'000	14%	BUWAL 2001b
Cement kilns	65'000	18%	BUWAL 2001i
Industrial furnaces	114'000	31%	<sup>1</sup>
Total	366'000	100%	BUWAL 2001a

<sup>1</sup> difference to total

### Typical design and process

In many respects a hazardous waste incinerator is similar to a municipal solid waste incinerator. Various types of HWI furnaces exist. All HWI incinerators usually need auxiliary fuel input to maintain high combustion temperatures. For liquid wastes, like waste solvents, a spray injection furnace converts the waste to fine droplets, which evaporate quickly (pulpvaporization process). For a combination of solid, semi-solid or liquid wastes, a rotary kiln incinerator can be used (see Fig. 2.5). The main unit consists of a slightly inclined rotating cylinder (2) typically 3 m in diameter and 10 m long. Wastes and auxiliary fuel are introduced into the high end of the kiln (1). Solid wastes and drummed wastes are fed directly into the rotary kiln by a conveyor system, crane or ram. Drums and cartons of waste can be shredded prior to introduction to the incinerator. Liquids, pumpable sludges and additional fuel oil are injected through a nozzle into the kiln or the secondary combustion chamber (4). Combustion takes place while the cylinder slowly rotates at 1 to 5 rotations per minute. Combustion temperatures range from 800 to 1600°C and residence times from several seconds to hours. Partially combusted waste gases are passed to a secondary combustion chamber (2) for further oxidation, typically for 2 seconds at 1200°C. A typical commercial rotary kiln installation can handle up to 50'000 ton of waste per year. As a new development, also fluidised bed incinerators can be used as HWI.

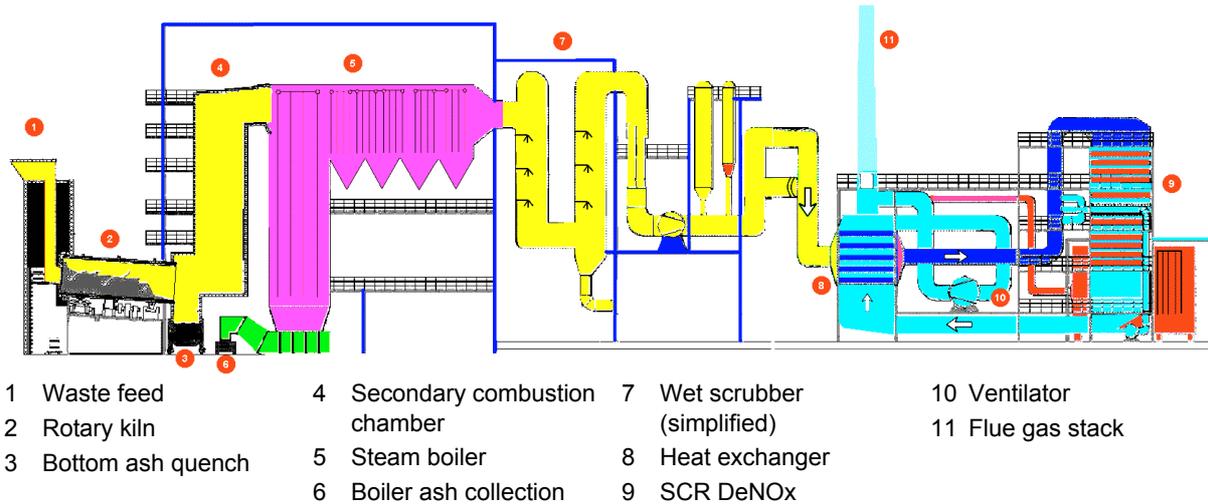


Fig. 2.5 Simplified scheme of a hazardous waste incinerator

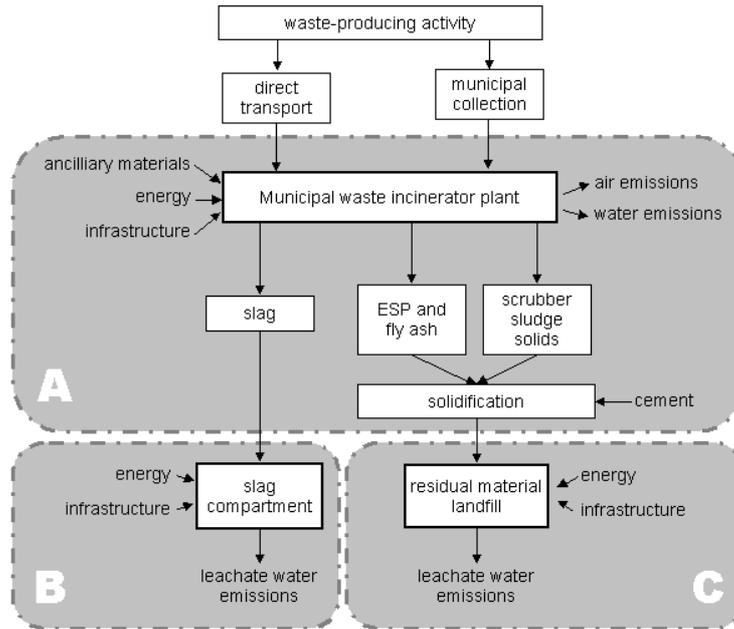
After combustion the flue gases are led through a steam boiler (5) and are quenched in a multi-stage wet scrubber, which removes acids (7). Some configurations also feature an electrostatic precipitator and selective, non-catalytic NO<sub>x</sub> reduction (SNCR-DeNO<sub>x</sub>) within the secondary combustion chamber. Non-combustible metal and other residues are discharged as bottom ash at the end of the kiln and quenched (3). Boiler ash is collected from the steam boiler and quenched (6). Plants with low-dust selective catalytic NO<sub>x</sub> reduction (9, SCR-DeNO<sub>x</sub>) need natural gas to reheat the flue gas before entering the DeNO<sub>x</sub> installation. Wastewater from the wet scrubbers is neutralised and pollutants are precipitated. Sludge and ashes are landfilled. Wastewater is discharged to surface waters. The purified flue gas is emitted through a flue gas stack (11).

## 2.2 System boundaries

### 2.2.1 System boundaries for municipal waste incineration

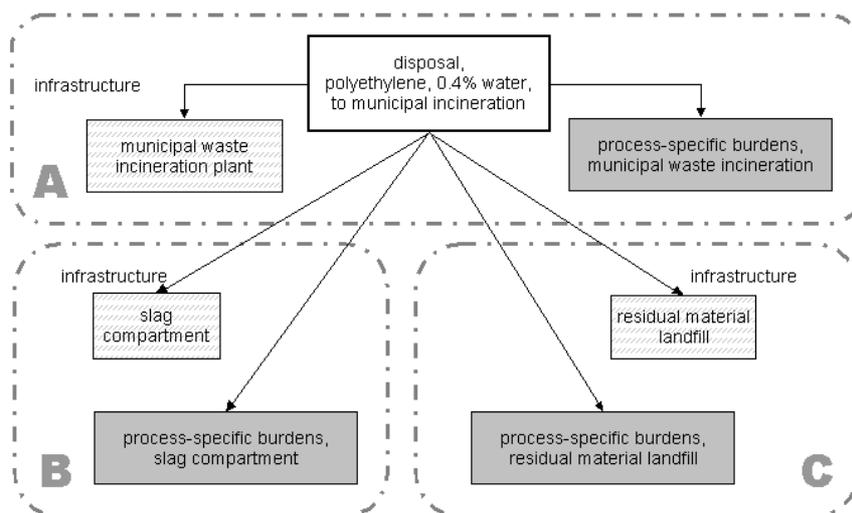
The goal of this study is to inventory disposal of different waste materials in MSWI, so that this data can be used in LCAs of waste-producing activities. Average Swiss technology is assumed. Fig. 2.6 shows the process chain for incineration of municipal solid waste. First the waste needs to be transported from the location of the waste-producing activity to the incinerator plant. *Within the scope of ecoinvent 2000 this transport is inventoried by the waste-producing activity and not the waste disposal process.* To avoid double counting the waste inventories produced in this report, do not contain the transport of the waste<sup>9</sup>. However, the inventory needed to account for municipal waste collection of bagged MSW or collected, non-hazardous industrial waste is performed in this report in part I.

<sup>9</sup> This is in contrast to the waste inventories performed before. In (Frischknecht et al. 1996, Hellweg 2000, Doka 2002,) the waste inventories all contained the transport to the MSWI.



**Fig. 2.6 System boundaries for the municipal waste incineration process chain**

The incineration of the waste leads to direct air and water emissions as well as land use burdens. Indirect burdens originate from the consumption of ancillary materials, energy and infrastructure materials. The solid remains from incineration are landfilled. Slag (bottom ash) is landfilled in a slag compartment; fly ash and scrubber sludge are solidified with cement and landfilled in a residual material landfill. The data modules devised in this study contain all the burdens from MSWI and the landfills. The burdens associated with the MSWI (area A in Fig. 2.6) are inventoried in chapter 3 'Life Cycle Inventory of Municipal Waste Incineration' starting on page 26. The burdens associated with the slag compartment and the residual material landfill (area B and C in Fig. 2.6) are described in part III on landfills of this report.



**Fig. 2.7 Data module structure for municipal waste incineration with the module names applied in the database. Here for polyethylene. Infrastructure modules are shown as hatched boxes; process-specific modules as grey boxes**

Fig. 2.7 shows the data structure for waste disposed in MSWI. The top central module needs to be requested by waste-producing activities for waste disposal<sup>10</sup>. The name of this module has always the structure '*disposal, XY, w% water, to municipal incineration*', where XY is the name of the waste, w% gives the water content of the waste at the time it is incinerated. Water content is an important feature regarding heat recovery from waste. The disposal of one kilogram dry polyethylene must be inventoried as  $(1/(100\%-w\%))$  kilogram (wet polyethylene) to municipal incineration by the waste-producing activity, using the water content given in the name of the module. Even dry waste is assumed to get wet in the waste bag. There might however be waste materials like paper that are never absolutely dry.

The central module contains all direct emissions from incineration, transport to landfills and *also the total direct emissions from the landfills*. That way, the central module contains *all waste-specific information*, i.e. exchanges that are unlike for different wastes<sup>11</sup>. The other modules featured in Fig. 2.7 are *constant* for all types of waste: process-specific modules (grey boxes) contain emissions and burdens that are only dependent on the type of disposal technology employed; infrastructure modules (hatched boxes) are required to be separate in ecoinvent 2000 and are also constant for any waste.

The functional unit of all modules except infrastructure is 'kilogram'. In section A of Fig. 2.7 the unit is 'kilogram' of wet waste, in section B 'kilogram' of slag, in section C 'kilogram' of residual material. Infrastructure modules are inventoried per facility/plant.

## 2.2.2 System boundaries for hazardous waste incineration

The system boundaries for hazardous waste incineration are analogous to the boundaries for municipal waste incineration. Waste transportation to the HWI must be inventoried with the waste-producing activity.

<sup>10</sup> Apart from transport of waste.

<sup>11</sup> The reason for this data structure is data module clarity and efficiency. Otherwise the ecoinvent database would be inflated with several modules per waste material (e.g. 'disposal, slag from polyethylene incineration, to slag compartment' etc.) of which *only one* would be meant to be requested by the LCI user.

The incineration of the waste leads to direct air and water emissions as well as land use burdens. Indirect burdens originate from the consumption of ancillary materials, energy and infrastructure materials. The solid remains from incineration are landfilled. Bottom ash, fly ash and scrubber sludge are solidified with cement and landfilled in a residual material landfill. The data modules devised in this inventory contain all the burdens from the HWI process and the landfill. The burdens associated with the HWI (area A in Fig. 2.8) are inventoried in chapter 4 'Life Cycle Inventory of Hazardous Waste Incineration' starting on page 51. The burdens associated with the residual material landfill (area B in Fig. 2.8) are inventoried in part III on landfills of this report.

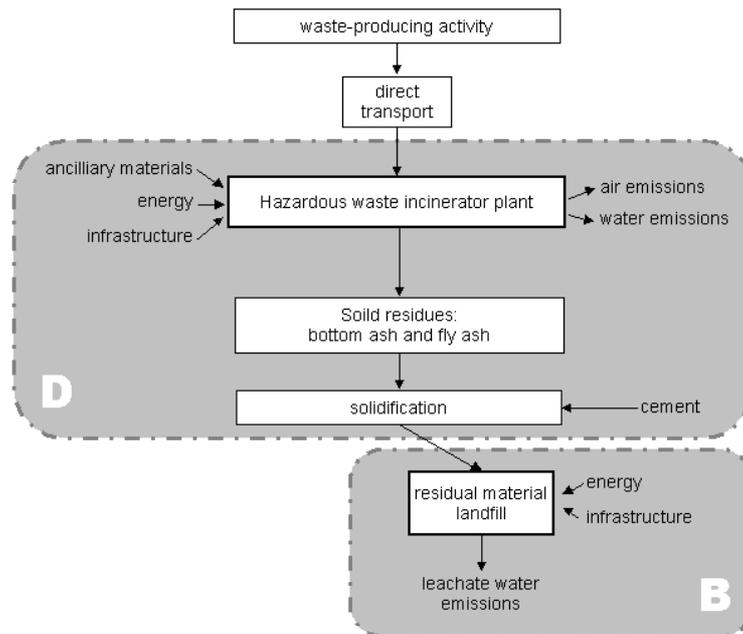


Fig. 2.8 System boundaries for the hazardous waste incineration process chain

The name of the data module to be requested by waste-producing activities has the structure '*disposal, XY, w% water, to municipal incineration*', where XY is the name of the waste, w% gives the water content of the waste at the time it is incinerated. The functional unit is 'kilogram' (wet waste).

## 2.3 Waste-specific burdens from incineration

Waste incinerators burn municipal waste, which is delivered to them. This waste composition is an average mixture of different waste fractions and of different waste products. Under average operation conditions the emissions from a MSWI can be determined. These emissions are attributable to components present in the incinerated waste, e.g. the lead emissions to air, to water and to solid outputs are caused by waste products containing lead. In LCA of products, we are interested in the specific burdens that are caused by the product when it is disposed. So the question is "How can we obtain *waste-specific* burdens from information on *average waste*?".

One quick solution here would be to inventory the burdens from the incineration of *average* waste as a proxy for the *waste-specific* burdens. This is an unsuitable solution, as waste fractions have distinctly different compositions and the burdens from a specific waste could be greatly under- or overestimated.

The solution employed in this study is to make the following assumptions and simplifications:

1. The waste-specific emissions are dependent on the composition of the specific waste. That is, the input determines the output.

2. The relevant features of the waste composition can be described by an analysis of composition of chemical elements (base analysis).
3. The waste-specific emissions are also dependent on the characteristics of the incinerator plant.
4. The relevant features of the characteristics of the incinerator plant can be described by an analysis of the flows of chemical elements through the incinerator (transfer coefficients).

If we know

- a) the elemental composition of a certain waste, and
- b) the flows of chemical elements through the incinerator,

the waste-specific burdens can be determined.

The elemental composition of waste fractions (a) can be determined or estimated for every waste product. The flows of chemical elements through the incinerator (b) can be determined for every incinerator technology based on the average waste feed, i.e. average operation conditions. The elemental flow characteristics of a certain element are expressed as so called *transfer coefficients*. Transfer coefficients (TK) for a certain chemical element describe the relation between the mass of this element in a certain output flow and the mass of this element in the waste input. Transfer coefficients must be derived for *every chemical element under consideration* and for *every output product* of the waste incinerator.

$$\text{Eq. 2.1} \quad \text{transfer coefficient for element } i \text{ to output } k = \frac{\text{mass of element } i \text{ in output } k}{\text{mass of element } i \text{ in average waste input}} = TK_k(i)$$

Eq. 2.1 shows a general formula to derive transfer coefficients from average operation of a plant. The masses involved must of course correspond to each other and e.g. relate to the same duration of operation time.

This procedure can be described as multi-input allocation. In LCA the case of a multi-output allocation is more common, i.e. a process generates several output products and the burdens from the process must be allocated to the various output products<sup>12</sup>. This situation is shown in Fig. 2.9 on the left.

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<sup>12</sup> In multi-output allocation usually an allocation key is used to value the output products. Let B be the total process emission of a substance X to air. If  $a_i$  is the content of the valued property in the output product  $i$  and A is the total process output of the valued property, then only the share  $a_i / A$  of the total emission B is allocated to the product  $i$ . I.e. the emission  $b_i$  allocated to the output product  $i$  is  $b_i = B * a_i / A$ .

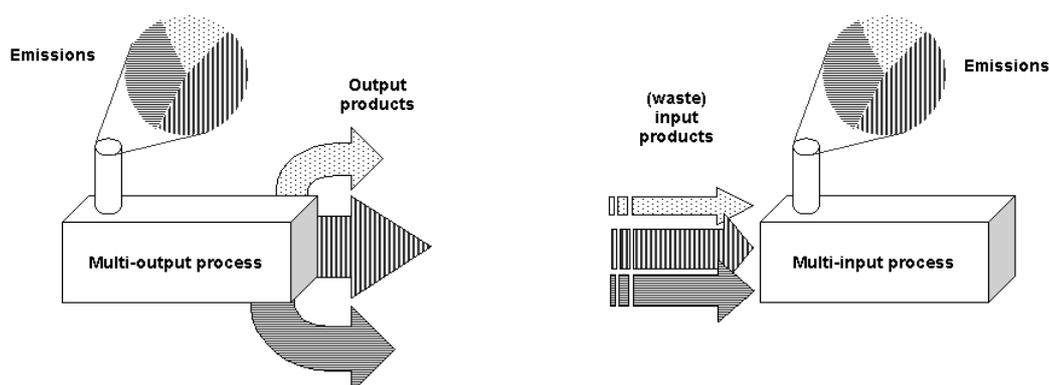


Fig. 2.9 Simplified scheme of the allocation of emissions in *multi-output* allocation (left) and *multi-input* allocation (right)

In waste disposal we have a similar situation: a mixture of material *inputs* is processed by incineration and we want to allocate the burdens to the various waste inputs (cf. Fig. 2.9 on the right). More precisely, we are conducting a separate allocation *for every chemical element*: For example the total average cadmium emissions to air of the incinerator are allocated to all cadmium-bearing wastes<sup>13</sup>.

However, this procedure of determining waste-specific burdens using transfer coefficients determined from average operation assumes that the chemical elements of the specific waste behave identically to the chemical elements of the average waste. In many cases, this is not a poor assumption. In certain cases however further distinction is necessary. For example, finely dispersed lead in a plastic waste will behave differently from lead in a bulk glass material. For the incinerator disposal systems an additional distinction is made between burnable and inert materials, which will have an effect on the applied transfer coefficients (cf. chapter 3.2.1 'Differentiation of burnability of waste materials' on page 26).

Differences in speciation of chemical elements might also be relevant (e.g. metallic zinc  $Zn^0$  vs. oxidised zinc  $Zn^{+II}$ ). However, usually there is no information on speciation available, and thus all wastes are treated as if their speciation of elements corresponds to the speciation encountered in average waste.

## 2.4 Necessary waste data

### 2.4.1 Elemental composition

To inventorise waste-specific burdens, the most important information within this model is the elemental composition of the waste. An extensive vector<sup>14</sup> of elemental concentrations in the waste can be specified by the user in the Excel calculation sheets (Tab. 2.5). This data is given in kilogram chemical element per kilogram wet waste. All composition data refers to the *wet composition* of the

<sup>13</sup> For example, let B be the cadmium emissions to air in average operation of a MSWI, and A is the total cadmium content of the *average* waste input. The transfer coefficients for cadmium to air  $TK_{air}(Cd)$  is then calculated as  $B/A$ . To calculate the cadmium emissions to air from a *specific* waste fraction *i*, this transfer coefficient is multiplied by the cadmium content of this waste fraction, which is  $a_i$ . So the cadmium emissions to air  $b_i$  (which are attributed to the waste fraction *i*) are then  $b_i = a_i * TK_{air}(Cd) = B * a_i / A$ . This expression is exactly analogous to the formula used in multi-output allocation (cf. footnote 12 on page 13).

<sup>14</sup> The order of the vector is as follows: major elements, other non-metals, halogens, heavy metals, prominent ash elements. This order is as arbitrary and subjective as any other encountered in literature and is adapted from the data referenced in (Zimmermann et al. 1996).

waste. The corresponding water content can be specified for information purposes, although it has little consequence later in the calculations, except for the GSD of the lower heating value. The water content can also influence the heating value, but only if no literature values for the heating values are given and heating values extrapolated from composition are used. The water content of the waste will be appearing in the name of the created exchange data module.

### 2.4.2 Heating values

For energy balance calculations, the upper and lower heating value must be specified in Megajoule per kilogram wet waste. The lower heating value should be consistent with the specified water content. Please note that the upper heating value derived from *dry mass* decreases if specified for a wet mass<sup>15</sup>. As assisting information, the sheet separately calculates the upper and lower heating values according to Michel (1938), which can be adopted by the user, if no literature data on the heating values is available.

The heating value of the waste determines how much usable energy can be obtained from the waste. In this study allocation between the waste disposal service and the energy production service of an MSWI plant is 100% on the waste disposal service. I.e. the energy production is free of any burdens and there is no reduced burden for high energetic waste. So the heating value only influences the waste heat emissions of the MSWI. Note that wet wastes can have negative lower heating values, which are accepted as input and lead to an increase of MSWI-internal energy consumption.

The uncertainty of the upper and lower heating values are mainly dependent on the elements that influence the heating values. The GSD for upper and lower heating values are derived from the GSD of elemental concentrations of O, H, C, S, N and water in the waste, i.e. the elements that contribute to the heating value according to (Michel 1938). Since H, C, S, and N add to the heating value and O and water decreases the lower heating value, the formula for subtraction is used to calculate the GSD value. I.e. the GSD of the heating values are waste-specific.

### 2.4.3 Share of magnetic iron

The incinerator model heeds magnetic separation of iron from MSWI slag according to the present Swiss technology mix. To calculate waste-specific descraping (and accordingly a reduction in slag mass), the user can specify, what percentage *of the given iron content* is magnetic iron. The GSD of that parameter is assumed to be 103%.

### 2.4.4 Biogenic vs. fossil carbon

Within the ecoinvent 2000 project, CO<sub>2</sub> and CH<sub>4</sub> air emissions are separated into emissions originating from biomass (paper, cardboard, vegetable waste, wastewater sludge) and from fossil material (plastics, mineral oil, solvents). The user must specify, what percentage of the given carbon content is biogenic. The GSD of that parameter is assumed to be 100%.

### 2.4.5 Degradation rate in sanitary landfill

For disposal in municipal landfills, the user must additionally specify the degradation rate during the first 100 years (See part III on landfills).

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<sup>15</sup> If  $H_{u,dry}$  is the upper heating value and  $w\%$  is the water content in the wet waste in kg/kg, then the upper heating value for the wet waste is  $H_{u,wet} = H_{u,dry} * (1-w\%)$ .

## 2.4.6 Solidification for wastes to residual material landfill

For *direct* disposal in residual material landfills – i.e. not via MSWI –, the user can specify if the waste is solidified with cement before deposition (See part III on landfills).

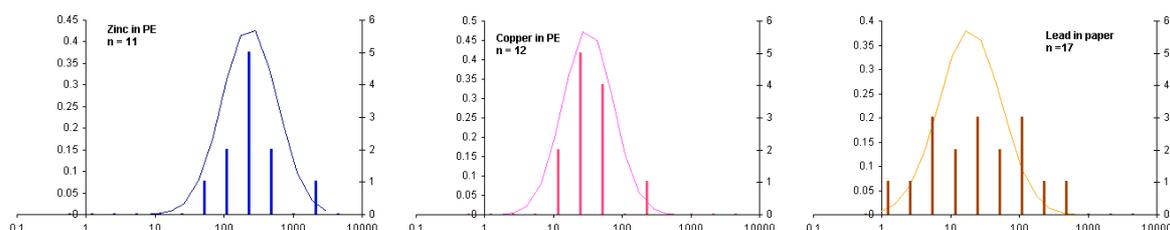
**Tab. 2.5 User-definable parameters for the waste composition and waste characterisation**

User-definable Parameter		Given as...
Unique name for the waste		Text
upper heating value	Ho	MJ/kg wet waste
lower heating value	Hu	MJ/kg wet waste
Water content	H2O	kg/kg wet waste
Oxygen (without O from H2O)	O	kg/kg wet waste
Hydrogen (without H from H2O)	H	kg/kg wet waste
Carbon (enter share of biogenic C below)	C	kg/kg wet waste
Sulfur	S	kg/kg wet waste
Nitrogen	N	kg/kg wet waste
Phosphor	P	kg/kg wet waste
Boron	B	kg/kg wet waste
Chlorine	Cl	kg/kg wet waste
Bromine	Br	kg/kg wet waste
Fluorine	F	kg/kg wet waste
Iodine	I	kg/kg wet waste
Silver	Ag	kg/kg wet waste
Arsenic	As	kg/kg wet waste
Barium	Ba	kg/kg wet waste
Cadmium	Cd	kg/kg wet waste
Cobalt	Co	kg/kg wet waste
Chromium	Cr	kg/kg wet waste
Copper	Cu	kg/kg wet waste
Mercury	Hg	kg/kg wet waste
Manganese	Mn	kg/kg wet waste
Molybdenum	Mo	kg/kg wet waste
Nickel	Ni	kg/kg wet waste
Lead	Pb	kg/kg wet waste
Antimony	Sb	kg/kg wet waste
Selenium	Se	kg/kg wet waste
Tin	Sn	kg/kg wet waste
Vanadium	V	kg/kg wet waste
Zinc	Zn	kg/kg wet waste
Beryllium	Be	kg/kg wet waste
Scandium	Sc	kg/kg wet waste
Strontium	Sr	kg/kg wet waste
Titanium	Ti	kg/kg wet waste
Thallium	Tl	kg/kg wet waste
Tungsten	W	kg/kg wet waste
Silicon	Si	kg/kg wet waste
Iron (share of metallic iron cf. below)	Fe	kg/kg wet waste
Calcium	Ca	kg/kg wet waste
Aluminium	Al	kg/kg wet waste
Potassium	K	kg/kg wet waste
Magnesium	Mg	kg/kg wet waste
Sodium	Na	kg/kg wet waste
Share of iron in waste that is metallic/recyclable		w-%
Share of carbon in waste that is biogenic		w-%
Degradability of waste in a municipal landfill within 100 years		w-%
Solidification of waste to residual landfill with cement? (Yes/No)		Yes/No

## 2.5 Uncertainty of waste composition data

In this report elemental waste composition data is used to calculate waste-specific emissions and treatment expenditures. Elemental waste composition data for various wastes is available from several literature sources (cf. part I). Compositional variation and uncertainty in waste measurements is typically high – especially for trace elements.

From existing measurements data uncertainty can be calculated. It is assumed in the following that the probability distribution of all composition data is lognormal. This can be justified by the distribution for some elements in materials where several measurements are available, cf. Fig. 2.10.



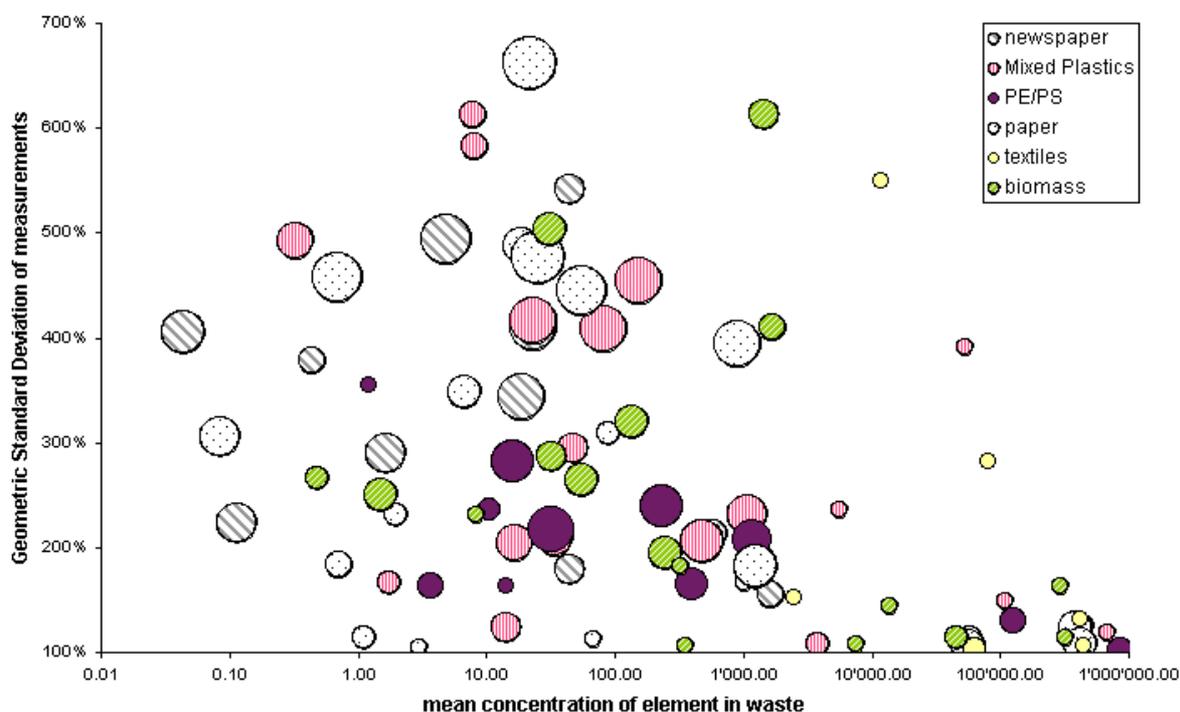
**Fig. 2.10** Histogram of measured composition data for some elemental concentrations in different waste fractions. The bell-shaped line represents a fitted *lognormal* distribution, which appears *symmetrical* due to the logarithmic x-axis

In Fig. 2.11 the calculated geometric standard deviation GSD of measurements for plastics, paper, cardboard, biomass<sup>16</sup> is plotted against the elemental concentration of the element under consideration. Waste measurements are taken from a large range of publications<sup>17</sup>. A trend is observable that the GSD increases with decreasing concentration. This is not surprising, as elements that constitute a major part of the waste (e.g. carbon in paper) will vary less than trace elements (e.g. lead in paper).

One way to obtain uncertainties for elemental waste composition data would be to simply use the data uncertainty calculated from existing measurements. This, however, is only feasible, if several measurements are available. If only one measurement is available, data uncertainty calculations from that data are not possible. In such cases uncertainty needs to be estimated. Instead of using the *non-specific* Pedigree approach devised for ecoinvent 2000 (Frischknecht et al. 2003a) a quantitative scheme is developed here to calculate generic uncertainty in elemental concentration measurements in waste, which is *based on such measurements*.

<sup>16</sup> These are waste types where measurements for a lot of elements are available.

<sup>17</sup> Publications used are (Barrage et al. 1995, Baumann et al. 1993, Bildingmaier 1990, Brahmms et al. 1989, BUWAL 1995, Domalski et al. 1987, Feess-Dörr et al. 1991, Frankenhäuser et al. 1995, Franssen et al. 1990, Grünwald et al. 1988, Haber et al. 1990, Hamm et al. 1986, IFEU 1991, IFEU 1992, Kaiser 1975, Keilen et al. 1977, Knezevic 1988, Mark et al. 1994a, Mark et al. 1995, Maystre et al. 1994, Rasp 1988, Salami et al. 1992, Stumpf 1994, Thalmann et al. 1982, Tillman 1991, Tuminski 1988, and Wünschmann et al. 1995).



**Fig. 2.11** Geometric standard deviation GSD of elemental measurements in various waste fractions vs. the mean concentration of the element in the waste in ppm. The bubble area indicates the number of measurements for that element/waste combination. The smallest bubble corresponds to two measurements, the largest to 17.

The prerequisites for this scheme are:

1. The mechanisms that govern the type of distribution of elements – especially traces – in materials are the same for all waste types. Hence, the observed trend of GSD vs. concentration from measurements in plastics, paper, biomass waste can be transferred to all types of waste.
2. The mean value of the GSD values for a certain concentration range reported in literature is a good estimate for an *unknown* GSD value in that concentration range.

The advantage of using this scheme is that uncertainty data can be obtained based on a broad sample for various materials and therefore minimises erratic results that would result if only the few available measurements for a specific chemical element in the waste material under consideration were used. A possible disadvantage might be that the first assumption of universal trace distribution mechanics might not be true for all materials considered. There might be materials where the concentration of certain elements is well determined by specific processes and the uncertainty might be overestimated with this procedure.

To estimate the data uncertainty of elemental concentrations in waste, the observed trend of Fig. 2.11 can be quantified. To quantify the trend, the data displayed in Fig. 2.11 is *averaged* over discrete concentration ranges. The weighted mean of GSD values from measurements vs. elemental concentration across all concentrations ranges are shown in Fig. 2.12 as hatched lines. The weighted mean is determined by weighing data points from a high number of measurements higher in the mean than measurements with only few points. The weighted average GSD value within the range can be determined as geometric mean or arithmetic mean. It is assumed that the distribution of the GSD values is lognormal. The mean of GSD values can then be determined as geometric mean. A weighted geometric mean of values  $GSD_i$  based on  $n_i$  measurements is determined by Eq. 2.2.

$$\text{Eq. 2.2} \quad \overline{GSD} = T \sqrt[T]{\prod_i (GSD_i)^{n_i}} \quad \text{with } T = \sum_i n_i$$

where

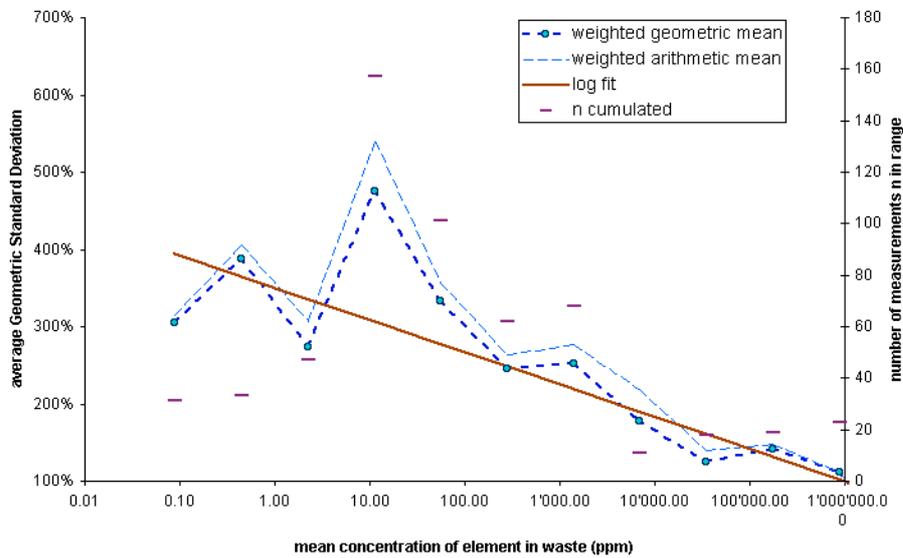
$GSD_i$  GSD value from one source

$n_i$  number of data points to derive  $GSD_i$

$\overline{GSD}$  is the mean GSD value

$T$  is the T-th root

$\prod_i GSD_i$  is the product of all  $x_i$



**Fig. 2.12 GSD values for measurements of elemental concentrations vs. the elemental concentration. Averaged GSD values from measurements are shown as hatched lines. The approximation for generic estimation of GSD values is shown as a solid line**

The trend of increasing GSD values with decreasing concentrations as detected in Fig. 2.11 becomes obvious in Fig. 2.12 (hatched lines). However, this trend seems to be discontinued for concentrations below 10 ppm (0.010%) where the GSD oscillates in the range of 280%-500%<sup>18</sup>. The trend is approximated by a logarithmic function that is drawn as a solid line in Fig. 2.12. The fit has the following form:

$$\text{Eq. 2.3} \quad GSD = N \cdot \ln(c) + 1 \quad \text{with } N = -0.181$$

where

$c$  concentration of element in kg/kg.

<sup>18</sup> This trend is maybe not significant as the number of measurement points in the range below 10 ppm is decreasing with decreasing concentration.

The approximation heads the reasonable boundary condition that the GSD approaches 100% for high concentrations<sup>19</sup>. On the other hand, for values of  $c$  below 0.1 ppm a constant GSD value of 400% is assigned. **This function will be used to calculate GSD values for elemental concentrations in waste compositions as a function of that elemental concentration itself.**

*Corollary:* It is reasonable to postulate, that the calculated uncertainty should be small enough, so that the probability of the concentration being larger than 100% – an impossibility – is negligible. The upper (97.5%) confidence boundary value  $u$  for a lognormal distribution is given by  $m/GSD^2$ . (where  $m$  is the mean value). It is assumed that this upper boundary shall not be bigger than 100%, i.e.  $m/GSD^2 = 1$ . Thus, for a mean value  $m$ , the GSD must not be above  $\sqrt{1/m}$ . This condition is fulfilled for the fitted function  $GSD = N \cdot \ln(c) + 1$  using the chosen values of  $N$ ; i.e.  $N \cdot \ln(m) + 1$  is always smaller than  $\sqrt{1/m}$ .

## 2.6 Emission speciation and category

### 2.6.1 Air and water emissions

As emissions are calculated from *elemental* waste compositions it is necessary to assign chemical substances to certain types of air and water emissions. Tab. 2.6 shows speciation of direct air and water emissions from a MSWI.

Tab. 2.6 Speciation of waste-specific air and water emissions from MSWI

Element	Air emission	Water emission
C	CO <sub>2</sub>	TOC, DOC, BOD, COD
S	Sulphate dioxide SO <sub>2</sub>	Sulphate SO <sub>4</sub> <sup>2-</sup>
N	Nitrogen oxide NO <sub>x</sub>	Nitrate NO <sub>3</sub> <sup>-</sup>
P	(P)	Phosphate PO <sub>4</sub> <sup>3-</sup>
F	Hydrogen fluoride HF	(F)
Cl	Hydrogen chloride HCl	(Cl)
Br	Hydrogen bromide HBr	(Br)

Carbon emissions to water are inventoried *simultaneously* as TOC, DOC, BOD, and COD in the ecoinvent database (Frischknecht et al. 2003a). For the MSWI effluent the carbon emissions are interpreted as total organic carbon TOC. It is safe to assume that all this carbon is dissolved. The majority of any particulate carbon would be transferred to the scrubber sludge during wastewater treatment. Hence the dissolved organic carbon DOC is set equal to the TOC value. No information on COD and BOD concentrations in MSWI is available. It is assumed that the COD is the oxygen demand for complete oxidation of the carbon ( $C + O_2 \Rightarrow CO_2$ ). Accordingly, the COD value can be calculated from the TOC value by multiplying with a factor of 2.29<sup>20</sup>. The BOD value is set equal to the COD value.

As shown in Fig. 2.13 MSWI are usually close to urban centres and located in the densely populated lowland regions of Switzerland. Air emissions are therefore designated to occur in the emission category 'air, high population density', i.e. in urban areas.

<sup>19</sup> I.e. the uncertainty for a hypothetical waste composed completely of one single element ( $m=1$ ) is very small and the GSD is 100%. In the approximation formula GSD for  $c=1$  equals  $N \cdot \ln(1) + 1 = -0.181 \cdot 0 + 100\% = 100\%$ .

<sup>20</sup> Two moles of oxygen atoms are needed to oxidise one mol of carbon atoms:  $2 \cdot 16 / 14 = 2.29$ .

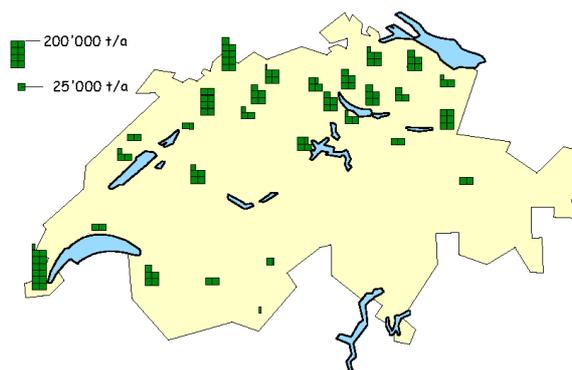


Fig. 2.13 Location and capacities of Swiss MSWI plants in 1999

Virtually all Swiss MSWI plants treat their scrubber liquids. The purified water is discharged to local rivers. An additional wastewater treatment is not necessary. Water emissions are therefore designated to occur in the emission category 'water, river'.

## 2.6.2 Oxidation of solid outputs

Similar to air emissions, the multiplication of the waste composition with the transfer coefficients gives only raw data on the solid outputs. The solid outputs (slag, residual material) are mostly oxidised. The theoretical amount of oxygen that needs to be present in the solid output, can be calculated from the elemental composition of metals. Usually there is too little oxygen in the raw composition calculated with transfer coefficients. This is understandable, as transfer coefficients refer only to the fate of waste elements. During incineration air oxygen is taken up, which leads to a slight weight increase in the solid outputs. This weight increase influences the transport weight during the transport to landfills and the cement needed for solidification of residual materials. The assumed oxidised substances and resulting weight increases per element are shown in the appendix in Tab. A 2.

In slag only a part of the iron and aluminium are oxidised. Only 50% of the metallic iron is removed depending on the MSWI technology mix (see chapter 2.8 'Allocation of iron scrap production' on page 24). The share of magnetic iron in waste is a waste-specific parameter (see chapter 2.4 'Necessary waste data' on page 14). The metallic iron that is *not removed by magnetic separation*, is not included in the calculation of air oxygen uptake. This iron remains as unoxidised metal in the slag. Similarly, about 34% of the aluminium in slag is metallic (Zeltner & Lichtensteiger 2002) and will not be included in the calculation of oxygen uptake.

For the residual waste material, complete oxidation of all elements is assumed.

The resulting weight increases as compared to raw calculated outputs are 1% for slag and 18% for residual waste material (in the case of average MSW).

## 2.7 Allocation of energy production

Besides disposing of waste materials, MSWI plants also produce energy in the form of thermal heat and/or electricity. Since both functions – waste disposal and energy production – are valuable, an allocation problem arises: How shall the burdens and expenditures created during waste disposal be allocated to these two functions? I.e. how much of the total burden of the waste disposal process shall be allocated to the waste disposal function of a MSWI?

Avoiding allocation by adapting the system boundaries is not feasible, as the initial liberation of chemical energy and the waste disposal occur at the same instance with the incineration of the waste. An allocation according to a fair casual or physical allocation key seems not feasible, as none of the

quantitative features of energy (energy or exergy content, or temperature level for heat) can describe a desired feature of the function of waste disposal<sup>21</sup>.

Allocation according to economical value is feasible for the functions 'waste disposal' (disposal fee) and the function 'energy generation' (energy price). The MSWI plants in Zürich had total earnings of 139 million CHF in 2000. Of that, 2.5 million CHF (1.8%) were net revenues from electricity sales and 17.6 million CHF (12.7%) of district heat sales<sup>22</sup>. This latter figure refers to the net revenues from district heat delivered to consumers. The district heat grid contains heat from waste but also heat from fossil auxiliary boilers to cover peak consumption during winter. The revenue figure above refers to that augmented amount of energy including fossil energy. In the year 2000 the share of fossil energy in the district heat was 37%. Accordingly, the district heat revenue that originates *from the waste alone* – which interests here – is merely 11 million CHF in 2000 or 8% of the total revenue. In 1999 the share of fossil energy in the district heat was 62% and district heat revenues that originated from the waste alone were only 3.5%, while 1.3% were from electricity sales. Over the two years 2000/1999 the average net revenue from total energy utilisation (heat and electricity) was 7.3%<sup>23</sup>. Energy production is therefore only a modest and certainly not dominant contribution to total revenue.

Is the goal of MSWIs energy production as well as waste disposal? Or is the energy production merely an optional side product of waste disposal by incineration? MSWI operators emphasise the fact that they consider MSWIs as 'combined heat/power plants fuelled by waste' (Kehricht-Heiz-Kraftwerk), also for reasons of public image. They operate MSWIs as to deliver heat when it is most needed and schedule partial load operation times or revisions during summer<sup>24</sup>. Energy utilisation from MSWIs is considered environmentally benign, as it helps to replace fossil energy carriers and put otherwise wasted energy to good use. Vigorous air pollution control makes waste competitive with commercial fuels and furnaces in respect to air emissions. Municipalities therefore often encourage energy utilisation from MSWIs. For example, in certain Zürich city areas it is compulsory to use the MSWI district heat for residential heating. Heating with other energy sources in these areas is prohibited (Zürich 2001). In fact, the Swiss Waste Ordinance prescribes that waste incinerators shall be built and operated so heat is utilised (TVA 2000:Art 38).

On the other hand energy production is clearly not the major goal of MSWIs, but rather a by-product. Waste has been burned in MSWIs formerly without energy utilisation. The tariffs for district heat are usually matched to fossil energy prices and are therefore not indicating market value of district heat. Indeed there is no market situation for district heat, even more if no other thermal energy source is allowed in designated district heating areas (cf. above). A market value of district heat would probably be lower than expressed above, as this energy form is unpopular. The Swiss MSWI practice (cf. Fig. 2.4) and municipal concepts (e.g. Zürich 2001) favour the production of electricity over heat. Continuing district heat production is probably mainly motivated by the large investment costs in its infrastructure<sup>25</sup>. But also electricity production is not particularly attractive as the electricity market is currently saturated and additional electricity is not very valuable. E.g. Zürich MSWIs sell their electric energy for 3.7 Rappen per kWh (ERZ 2001). The energy ordinance mentions the possibility of energy utilisation in MSWIs, but expressly refrains from setting minimal efficiency targets for MSWIs, unlike for other CHPPs (section 1-h in EnV 2002). That waste treatment is of primary importance over energy production can be illustrated by an imaginary situation where energy production would in some

<sup>21</sup> System expansion is not an option since the sensible functional unit for a service module of waste disposal within the ecoinvent 2000 database is '1 kilogram' and not '1 kilogram + x MJ electricity + y MJ heat'.

<sup>22</sup> Personal communications with Martin Leman, AVUT Consulting, Zürich in January and March 2002, own calculations and (ERZ 2001)

<sup>23</sup> Extrapolating Zürich tariffs to all Swiss MSWIs, the share of the net revenue from total energy utilisation was 11% in 2000.

<sup>24</sup> Personal communications with Martin Leman, AVUT Consulting, Zürich in January and March 2002 and (ERZ 2001)

<sup>25</sup> E.g. 40% of the investment volume of the Zürich disposal services is district heating infrastructure (ERZ 2001:68)

way compromise the waste treatment, which would lead to modification in the energy production and not adaptations in the waste treatment goals.

Mainly based on the fact that electricity utilisation from waste creates only a small part of the total yearly net income of a MSWI (roughly 5 – 10%, cf. above) it is decided that the full burden of the waste disposal process is allocated to the waste disposal function and 0% to the energy produced. This is in accordance with the former ETH waste inventories (Zimmermann et al. 1996, Frischknecht et al. 1996). This also means that energy from MSWIs is not burdened by air emissions or resource consumption, and that the *consumption* of such energy is without environmental burden (apart from district heat infrastructure and operation expenditures like peak boilers). This choice corresponds well with the popular understanding, that waste heat utilisation is an environmentally advantageous energy source.

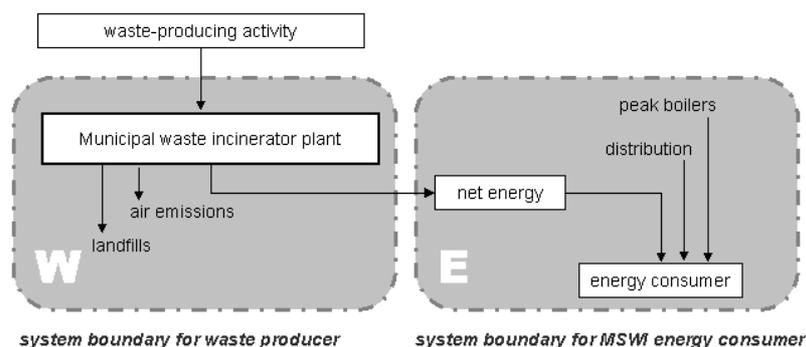


Fig. 2.14 System boundaries for waste disposal (W, left) and energy-from-waste consumption (E, right)

Within the ecoinvent 2000 project this means that the consumer of MSWI-generated energy is rewarded in the LCI by absent air emissions and resource use. On the other side, a producer of high-calorific wastes cannot profit from any reduced burdens in the waste disposal of his/her waste product. It seems fair that the actual *consumer* of MSWI-generated energy (who had special investments) shall profit from the MSWI efforts to utilise energy and not the *deliverer* of waste, who does usually make no special provisions to make the energy gain from his/her waste optimal<sup>26</sup>. The system boundaries for waste disposal (W) and energy-from-waste consumption (E) are shown in Fig. 2.14 (more detailed scheme of (W) in Fig. 2.6 on page 10).

**All burdens of waste incineration and subsequent processes are allocated to the function 'waste disposal'. Generated heat or electrical energy is free of any burden.**

<sup>26</sup> Another – quite disturbing – *ex post* reason to refrain from energy allocation, is that any allocation proportional to the energy produced from a particular waste fails to comply with the 100% rule, that a mixture of wastes  $M$  should be able to be represented by the weighted sum of its composing waste fractions  $m_i$  (with  $M = \sum_i [w_i \cdot m_i]$ ). I.e. if an allocation of the burdens proportional to the energy produced from the waste is introduced, it follows that inert wastes have no reduction of burdens, while high-calorific wastes do. It can be shown that in this case the allocated burden of a *mixed waste*  $A$  does not equate the weighted sum of the allocated burdens  $a_i$  of its *composing waste fractions*, i.e.  $A$  is not equal to  $\sum_i (w_i \cdot a_i)$ . I.e. we would get different LCIs if we were to inventorise allocated burdens for an average municipal waste mix *as a whole* or the sum of the allocated burdens for the *respective shares* of the individual waste fractions.

However for sensitivity analyses one process module (for average municipal waste) is formulated as multi-output process, where the user can change allocation factors. Suggested alternative Allocation factors are shown in Tab. 2.7.

**Tab. 2.7 Alternative allocation factors for sensitivity analysis**

	Suggested alternative allocation factors based on revenue	Outputs in average operation	request of Multi-Output-Process (MOP) per unit of Single Output
electricity	1.55% (1.3–1.8%)	0.2798 kWh <sup>1</sup>	0.0554 MOP for 1 kWh
heat	5.75% (3.5–8%)	2.164 MJ <sup>1</sup>	0.0266 MOP for 1 MJ
disposal function	92.7% (90.2%–95.2%)	1 kg disposed	0.927 MOP for 1 kg

<sup>1</sup> Based on a lower heating value of 11.74 MJ/kg waste and a water content in waste of 22.9 w-%

## 2.8 Allocation of iron scrap production

Approximately 50% of the MSW is burnt in MSWI with magnetic iron scrap separation. For the average Swiss MSWI technology mix, 26 kg scrap per ton of slag is separated. Like in chapter 2.7 'Allocation of energy production' it is debatable, whether burdens from incineration shall be reduced, because a recyclable material – and hence some value – is produced.

In Switzerland MSWI iron scrap is recycled in electric arc furnaces (EAF). The input to these EAFs is 100% scrap. MSWI iron scrap can only be added in small percentages due to high contents of copper and tin. Cleaner scrap from other sources is needed to dilute those contaminants. MSWI iron scrap is accepted to EAFs as a no value commodity, i.e. price is zero<sup>27</sup>.

Accordingly, there is little motivation to allocate burdens from the incineration to the no-value iron scrap from MSWIs. However, wastes with separable iron will profit from magnetic separation because less landfill volume is occupied. The chief motivation for iron recycling from the view of the MSWI operator is to save slag landfill costs, rather than to increase income.

## 2.9 Allocation of ancillary materials

Ancillary materials used in MSWI are used after the incineration step to treat the flue gas and scrubber liquids. Consumption figures of these materials are available for average operation of the plant. The consumption of some ancillary materials is caused by certain chemical elements in the waste. Therefore the use of ancillary materials can be allocated to the presence of these causing agents<sup>28</sup>. In a waste-specific inventory consumption of the ancillary materials is dependent on the presence and extent of these causing agents in the waste.

<sup>27</sup> Personal communication with Christoph Zeltner, Stahlwerk Gerlafingen, Switzerland, November 14, 2001.

<sup>28</sup> No common technical term for these causing agents, i.e. the flows to which burdens are allocated to exist in LCA terminology. In multi-output-allocation usually the word 'co-product' is used for the allocation recipient. However, in the multi-input-allocation performed here the corresponding term would be 'co-educt', which seems at least as unclear as using 'co-product' for input flows. In a personal communication with Wanda Philips, graduate of English language, University of Zürich, in December 21, 2002, the word 'allocation recipient' or 'allocation receiver' was deemed most suitable. New expressions are imaginable (allocant, allo-getter), but seem more confusing than clarifying.

## 2.10 Process-specific burdens

Apart from waste-specific emissions which can be attributed to some waste property (i.e. elemental content), there are also emissions from MSWI which are not attributable to some waste property. For example carbon monoxide, dioxins, and thermal nitrogen oxides are less dependent on waste input but more on the operating conditions like temperature, flow velocities and oxygen concentrations. These emissions are evenly attributed to the waste input on a mass basis; i.e. each kilogram of waste, regardless of its composition, will be attributed a certain constant amount of these emissions. In contrast to waste-specific emissions described above, these emissions are called *process-specific emissions*.

## 3 Life Cycle Inventory of Municipal Waste Incineration

### 3.1 Assessed technology

As pointed out in chapter 2.1.1 'Municipal solid waste incinerator' on page 2, the average MSWI in Switzerland has an electrostatic precipitator and a wet flue gas scrubber system. The few incinerator lines with no electrostatic precipitator or a semi-dry scrubber system are neglected in this report. There is variation in the extent and type of NO<sub>x</sub> abatement installations. In this report the various types of NO<sub>x</sub> abatement are discerned and a weighted average mix of the technologies is inventoried.

### 3.2 Waste-specific emissions

As described in chapter 2.3 'Waste-specific burdens from incineration' on page 12, the waste-specific burdens are derived from multiplication of the waste composition and the transfer coefficients for different chemical elements and several MSWI-outputs. There are 5 relevant outputs: 1.) emissions to air, 2.) emissions to water, 3.) slag (bottom ash), 4.) fly ash and precipitator ash 5.) waste scrubber sludge. The emissions to air and to water are the direct emissions of the MSWI. Slags and ashes are landfilled and lead to indirect emissions downstream of the process chain. The outputs of the MSWI are shown in Fig. 3.1. The different chemical elements considered in this report are listed in chapter 2.4 'Necessary waste data' on page 14.

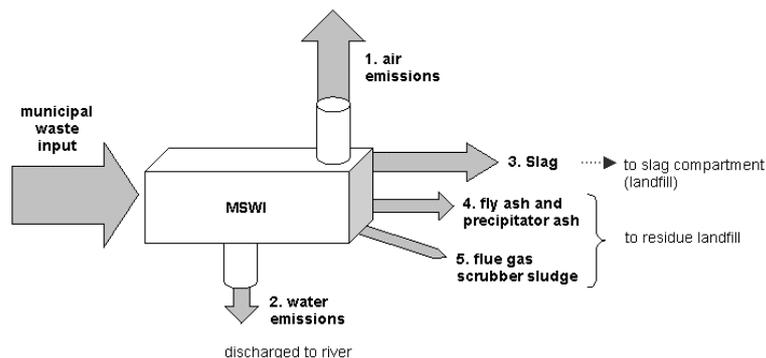


Fig. 3.1 The five material outputs of the MSWI considered in this report. Compare to Fig. 2.1 on page 3

#### 3.2.1 Differentiation of burnability of waste materials

The transfer coefficients for average waste can be obtained from measurements in literature (Hellweg 2000, Morf et al. 1997, Belevi 1998, Zimmermann et al. 1996). However, the application of *average* transfer coefficients can be inappropriate for the assessment of certain specific waste fractions. Inert materials like glass, ceramics or bulk metals are completely transferred to the slag. Application of average transfer coefficients would lead to obviously flawed results for these wastes. Therefore, waste products are differentiated as *burnable* or *inert* (i.e. non-burnable) wastes.

Inert wastes are assumed to be transferred completely to the slag, i.e. TK to slag for all elements of inert wastes is 100%; TK to all other outputs is 0%:

$$TK_{\text{Inert, slag}} = 100\% ; TK_{\text{Inert, } \neq \text{slag}} = 0\%$$

For burnable wastes a new set of transfer coefficients,  $TK_{\text{burn,k}}$ , is applied. The two sets of transfer coefficients,  $TK_{\text{burn,k}}$  and  $TK_{\text{Inert,k}}$ , must be consistent with the average transfer coefficients in such a way that the weighted sum of the outputs from all burnable or inert waste fractions present in average

waste, must result in the outputs as would be obtained from this average waste and the transfer coefficients for average waste, expressed by the following formula.

$$\text{Eq. 3.1} \quad O_{\text{avg},k} = I_{\text{avg}} \cdot TK_{\text{avg},k} = I_{\text{inert}} \cdot f_{\text{inert}} \cdot TK_{\text{inert},k} + I_{\text{burn}} \cdot f_{\text{burn}} \cdot TK_{\text{burn},k}$$

And accordingly:

$$\text{Eq. 3.2} \quad TK_{\text{burn},k} = \frac{O_{\text{avg},k} - (I_{\text{inert}} \cdot f_{\text{inert}} \cdot TK_{\text{inert},k})}{I_{\text{burn}} \cdot f_{\text{burn}}} = \frac{(I_{\text{avg}} \cdot TK_{\text{avg},k}) - (I_{\text{inert}} \cdot f_{\text{inert}} \cdot TK_{\text{inert},k})}{I_{\text{burn}} \cdot f_{\text{burn}}}$$

where:

- $O_{\text{avg},k}$  Average output  $k$  from average waste ( $k$  = flue gas, slag etc.) [g/kg waste]
- $I_{\text{avg}}$  Average waste input composition vector [g/kg waste]
- $TK_{\text{avg},k}$  Transfer coefficients for average waste and output  $k$  [-]
- $I_{\text{inert}}$  Composition of inert waste in average waste [g/kg waste]
- $f_{\text{inert}}$  Share of inert waste in average waste by weight, w-%
- $TK_{\text{inert},k}$  Transfer coefficients for inert waste and output  $k$  [-]
- $I_{\text{burn}}$  Composition of burnable waste in average waste [g/kg waste]
- $f_{\text{burn}}$  Share of burnable waste in average waste by weight, w-%
- $TK_{\text{burn},k}$  Transfer coefficients for burnable waste and output  $k$  [-]

For this study the  $TK_{\text{burn}}$  calculated by Hellweg (2000) are applied<sup>29</sup>. Compositions of different waste fractions in average waste are obtained from literature (Tillmann 1991, Mayestre et al. 1994, Zimmermann et al. 1996:B.25ff., BUWAL 1990, Finnveden 2001). Due to the large variations in waste and waste fraction composition data it is necessary to adjust the waste fraction data, so that the weighted sum of all waste fraction compositions is in accordance with the composition of average waste. An average waste composition from (Belevi 1998) was used as a reference.

Thus, all parameters to calculate  $TK_{\text{burn},k}$  are known. The compositions of waste fractions used for the calculation of  $TK_{\text{burn}}$  are shown in the appendix in Tab. A 1. All inert wastes are identified by a lower heating value ( $H_u$ ) of zero.

In the calculation tool it is possible to enter mixtures of burnable and inert wastes, e.g. 70% metal parts from an appliance as inert waste and 30% plastic parts as burnable waste. The inert waste is calculated with the transfer coefficients for inert waste and the burnable waste with the transfer coefficients for burnable waste.

### 3.2.2 Additional transfer coefficients

#### Proxies from coal power plants for B, Be, Sc, Sr, Ti, Tl, W

For some elements transfer coefficients for MSWI could not be calculated due to the lack of data. Some of the gaps (B, Be, Sc, Sr, Ti, Tl, W) are filled with data from modern coal power plants described in part VI of (Frischknecht et al. 1996). Coal power plants are in some ways similar to MSWIs. After incineration of the fuel, sulfur dioxide is removed from the flue gas in a wet scrubber

<sup>29</sup> Personal communication with Stefanie Hellweg, Safety and Environmental Technology Group of the ETH Zürich, Switzerland, June 28, 2002.

(flue-gas desulfurisation FGD with lime-based materials, not with NaOH like in a MSWI). The resulting gypsum product is separated and an effluent produced (corresponds to the scrubber sludge and water emissions in a MSWI). Particulate matter in the flue gas is removed with filters (corresponds to precipitator ash). Also, the coal incinerated is a burnable material and the transfer coefficients derived fit therefore well together with the notion of transfer coefficients for burnable waste  $TK_{burn,k}$  described above. However, these are only proxy estimates and should be regarded with care, if these elements influence the LCIA result significantly.

Transfer coefficients to air of coal power plants are listed in (Frischknecht et al. 1996:VI.179). The values for hard coal with FGD and dust filter efficiency of 99.9% are used. The difference of these figures to the transfer coefficients for hard coal *without FGD* (but an equal dust filter efficiency of 99.9%) are taken to be the transfer coefficients to scrubber sludge.

In (Frischknecht et al. 1996:VI.97) concentrations of traces from FGD effluent are listed in mg/l. The value for boron of 66 mg/l, derived from German coal power plants, is of interest here. Per kilogram German coal, 0.085 litres of REA effluent are produced<sup>30</sup>. German coal contains 37.23 ppm boron (Frischknecht et al. 1996:VI.107). Hence the transfer coefficient to water for boron is 15.1%.

In (Frischknecht et al. 1996:VI.116) compositions of hard coal and boiler ash, retained fly ash, and emitted fly ash for Dutch hard coal power plants are listed. Per Terajoule coal input Dutch plants produce 4400 kilograms of retained ash (Frischknecht et al. 1996:VI.117). The upper heating value of Dutch hard coal is 26.37 MJ/kg (Frischknecht et al. 1996:VI.173). Hence per kilogram of Dutch hard coal 116 grams of ash are retained. Using the composition data on the hard coal (input) and the retained ash (output) transfer coefficients can be calculated.

For Sc, Sr, and Ti the transfer coefficient to boiler ash was near (slightly above) 100%. For this reason it was decided that the transfer coefficient to boiler ash in the coal power plant shall represent the transfer coefficient to MSWI bottom ash (= MSWI slag). A transfer coefficient to slag of 100% minus the TKs to air, water and sludge was used for these three elements. These are elements that are essentially not volatile during incineration.

For B, Be, Tl, and W the transfer coefficient to boiler ash was significantly smaller than 100% (38.4%, 69.6%, 34.8%, and 75.1% respectively). These transfer coefficients to boiler ash in the coal power plant are used as the transfer coefficients to MSWI bottom ash (slag) for these four elements. The remainder to 100% (heeding the TKs to air, water and sludge) was set as transfer coefficient to fly ash (precipitator ash). These are volatile elements during incineration, but are captured later in the dust filter.

### **Educated guesses for silver**

No transfer coefficients for silver are known. The transfer coefficients for silver are derived from four neighbours of silver in the periodic table of the elements Cu, Zn, Cd, and Ni. With 1234°K silver has a moderate elemental melting temperature and can be expected to be somewhat volatile in the MSWI, which influences e.g. the transfer coefficients to fly ashes. To derive a weighing factor for the four neighbours of silver, the inverted absolute difference between the melting temperature of the neighbour element and the melting temperature of silver is used ( $= 1/|T_{melt,X} - T_{melt,Ag}|$ ), i.e. the closer the melting temperature of the neighbour element is to the melting temperature of silver, the larger the weight for the mean. Melting temperatures of the neighbour are 1357°K, 692°K, 594°K, and 1726°K respectively, resulting in weighting factors of 59.9%, 13.6%, 11.5%, 15.0%. So, Silver is assumed to

<sup>30</sup> Derived from 20 grams of SO<sub>2</sub> in raw gas per kg coal, a sulfur removal rate of 85% in REA, and 5 liters of effluent per kg SO<sub>2</sub> removed (Frischknecht et al. 1996:VI.95f.).

behave largely similar to copper, which seems to make sense. The transfer coefficient to slag is 61.5% (copper 80%).

### **Educated guesses for iodine**

No transfer coefficients for iodine are known. From the calculation of Dutch coals above, based on (Frischknecht et al. 1996:VI.116), it can be calculated that less than 5% of the iodine in coal is transferred to boiler ash. Iodine therefore seems not to be as ash-bound as bromine or fluorine, that feature transfer to slag and fly ashes of over 90%.

The remaining halogen, where transfer coefficient data is available, chlorine, is volatile. For example, only 7% are transferred to slag, but over 90% to water. This seems compatible with the observed characteristics of iodine during coal incineration. Therefore the transfer coefficients of chlorine are used for iodine without modification.

### **3.2.3 Synopsis of transfer coefficients**

The transfer coefficients for burnable waste are shown in Tab. 3.1. These figures can be changed waste-specifically, depending on the content of inert, unburnable material in the waste under consideration. Inert material increases the transfer coefficient to slag, depending on the relative share of the element in an inert matrix.

**Tab. 3.1 Transfer coefficients in the MSWI for burnable waste. For inert wastes the transfer coefficient to slag is 1000g/kg for all elements. See text for sources and explanation.**

Transfer coefficients	slag	boiler ash	ESP ash	scrubber sludge	water emissions	air emissions
Burnable waste	g/kg	g/kg	g/kg	g/kg	g/kg	g/kg
H <sub>2</sub> O	0	0	0	0	0	1000
O	69.8	2.04	10.2	1.02	0	917
H	0	0	0	0	0	1000
C	7.55	0	3.44	0.0101	0.0101	989
S	554	0	298	74.6	71.4	2.13
N	10	0	0	0	1	989
P	880	25	94	0	0	1
B	384	0	166	180	151	120
Cl	71.3	0	13	6.48	909	0.0108
Br	110	10	877	0	0	3
F	615	0	308	21	56	0.5
I	71.3	0	13	6.48	909	0.0108
Ag	615	5.32	274	106	0.0729	0.013
As	550	30	381	39	0.1	0.0000102
Ba	887	22	90	0	0	1
Cd	3.27	0	369	627	0.441	0.0551
Co	850	10	120	20	0.01	0.0000318
Cr	455	31.9	446	63.7	3.19	0.0000739
Cu	801	0	185	14.8	0	0.00738
Hg	5.74	2.09	366	615	10.5	0.0000345
Mn	860	10	120	10	0.01	0.00000545
Mo	867	21	110	0	0	2
Ni	901	35.5	63.5	0	0	0.0000432
Pb	66.4	0	848	85.4	0.0186	0.0371
Sb	16.1	18.2	822	144	0.182	0.00000389
Se	80.3	130	719	70.7	0.118	0.00000503
Sn	496	19.9	451	31.8	0.0133	1.33
V	890	10	90	10	0.01	0.1
Zn	3.29	0	817	180	0.163	0.0163
Be	696	0	294	9	0	1
Sc	999	0	0	0.5	0	0.5
Sr	999	0	0	0.9	0	0.1
Ti	999	0	0	0	0	1
Tl	348	0	650	1	0	1
W	751	0	249	0	0	0
Si	919	25.6	53.5	0	0	2.33
Fe	899	0	93.5	6.68	0.334	0.0334
Ca	862	36.7	100	0	0	1.67
Al	853	35.9	109	0	0	1.56
K	668	47.2	281	0	0	3.01
Mg	917	20.7	60.6	0	0	1.38
Na	614	94.1	282	0	0	9.41

### 3.2.4 Uncertainty estimates for transfer coefficients of MSWI

Transfer coefficients are obtained from measurements of the output flows of an element  $e$  divided by the measurements of the input flows ( $TK_e = \text{OUT}_e / \text{IN}_e$ ). In the LCI model calculation performed, the waste-specific output of the MSWI is calculated by multiplying the input waste composition by the transfer coefficients for the respective elements. The uncertainty of the output can be obtained from the uncertainties of the factors: input composition and transfer coefficients. However, since the data for the transfer coefficients is actually obtained from data on input and output, these uncertainties are not independent. Standard deviations and ranges of transfer coefficients from data of different MSWI plants are listed in (Erichsen & Hauschild 2000). However, this data should not be used as an uncertainty measure for transfer coefficients, as this data already includes uncertainty of the waste

input. To avoid double-counting of the uncertainty of the waste input the uncertainty of the transfer coefficients 'as such' needs to be evaluated.

The uncertainty of the transfer coefficients can be understood as the *system-inherent* uncertainty of the incinerator that occurs when a hypothetical waste with *perfectly defined* composition is incinerated. This uncertainty can occur e.g. through stochastic processes in the incineration chamber. Conditions in the incinerator are heterogeneous in certain respects and can influence minor pathways of elements. For example a flake of ash might be trapped in the slag, while an identical flake might be carried away in the air stream and go to the electrostatic precipitator ash. For transfer coefficients of minor pathways these stochastic processes cause high variance, while transfer coefficients of major pathways are hardly affected by them.

To estimate the system-inherent uncertainty of the transfer coefficients, it is assumed here that the GSD for large transfer coefficients is close to 1, i.e. major pathways of elements have a low uncertainty. For example silicon is transferred to slag with a transfer coefficient of over 90% and the uncertainty for that is rather low; i.e. it is extremely unlikely there is a MSWI plant with an average silicon-to-slag TK of 50% instead of 90%. On the other hand minor pathways are less well defined and the uncertainty is higher for small transfer coefficients. It is assumed that the GSD is not above 1.4 even for minor pathways; i.e. there is a 95%-confidence range of the TK value that is maximally between 50% and 200% of the mean value of the TK, even for minor pathways<sup>31</sup>.

The GSD value is derived from the logarithmic formula below. The type of equation is adopted from the uncertainty calculations performed for waste compositions (c.f. chapter 2.5 'Uncertainty of waste composition data' on page 17).

$$\text{Eq. 3.3} \quad GSD_{TK} = N \cdot \ln(m_{TK}) + 1 \quad \text{with } N = -0.022, \text{ TK in [kg/kg]}$$

The value of  $N$  is chosen so that the  $GSD^2$  value for the smallest  $m_{TK}$  values (metals to air), are 200%. I.e. at most the transfer coefficient is assumed to double or half of the mean value. For larger  $m_{TK}$  the  $GSD^2$  value decreases and approaches 100% for  $m_{TK} = 1$ .

The uncertainty of elements in output flows of the MSWI is a combination of the uncertainty in the waste and of the uncertainty of the transfer coefficients.

### 3.2.5 Waste-specific air emissions

Waste-specific air emissions are calculated by multiplication of waste composition data and transfer coefficients for air. Nitrogen containing air emissions are calculated using the species-profile derived in chapter 3.2.9 'Waste-specific burdens from fuel-NOx' on page 38.

#### Uncertainty estimates for waste-specific air emissions

Waste-specific air emissions are calculated by multiplication of waste composition data and transfer coefficients for air. The resulting GSD for air emissions can be obtained from the GSD of the factors by using the formula for multiplication of values. Uncertainty estimates for waste composition data are discussed in chapter 2.5 'Uncertainty of waste composition data' on page 17. Uncertainty estimates for

<sup>31</sup> If the GSD is 140%, then  $GSD^2$  is 200%. The 2.5%-confidence value is  $\text{mean}/GSD^2 = \text{mean} \cdot 50\%$ . The 97.5%-confidence value is  $\text{mean} \cdot GSD^2 = \text{mean} \cdot 200\%$ . The Pedigree approach used to determine generic  $GSD^2$  values within theecoinvent 2000 project suggests a basic uncertainty value of  $GSD^2 = 500\%$  for heavy metals to air. However, this basic uncertainty value mainly contains the uncertainty in the *compositions* of the burned material. The value for the transfer coefficient alone should therefore be smaller. The choices made here are compatible with the Pedigree approach.

transfer coefficients are discussed in chapter 3.2.4 'Uncertainty estimates for transfer coefficients of MSWI' on page 30.

Emissions of carbon dioxide, carbon monoxide and methane are discerned according to the fossil or biogenic origin of the carbon in the incinerated waste. E.g. fossil methane is inventoried separately from biogenic methane. The user must define the share of carbon in waste that is biogenic. Strictly, this share is a uncertain figure and this uncertainty influences the emissions of biogenic and fossil emissions. In most cases however, the uncertainty of the biogenic share will be small, i.e. the share will be 0% (plastics) or 100% (biomass products) and the uncertainty will be negligible. The uncertainty of the share of biogenic carbon in waste is neglected.

### **3.2.6 Waste-specific water emissions**

Waste-specific water emissions are calculated by multiplication of waste composition data and transfer coefficients for water. These relate to the flows after the wastewater treatment. The emissions are emitted into river water.

#### **Uncertainty estimates for waste-specific air emissions**

Waste-specific water emissions are calculated by multiplication of waste composition data and transfer coefficients for water. The resulting GSD for water emissions can be obtained from the GSD of the factors by using the formula for multiplication of values. Uncertainty estimates for waste composition data are discussed in chapter 2.5 'Uncertainty of waste composition data' on page 17. Uncertainty estimates for transfer coefficients are discussed in chapter 3.2.4 'Uncertainty estimates for transfer coefficients of MSWI' on page 30.

### **3.2.7 Waste-specific solid outputs**

The waste-specific solid outputs are calculated by multiplication of waste composition data and transfer coefficients for the several solid outputs (slag, boiler ash, precipitator ash, scrubber sludge). The slag is landfilled in a slag compartment (cf. part III). The boiler ash, precipitator ash, and scrubber sludge are combined, solidified with cement and landfilled in a residual material landfill (cf. part III). The oxygen balance is adjusted to match the oxygen content of oxidised elements (See chapter 2.6.2 'Oxidation of solid outputs' on page 21.).

The GSD values for the several solid outputs are obtained from the GSD of the waste composition data and transfer coefficients by using the formula for multiplication of values. Uncertainty estimates for waste composition data are discussed in chapter 2.5 'Uncertainty of waste composition data' on page 17. Uncertainty estimates for transfer coefficients are discussed in chapter 3.2.4 'Uncertainty estimates for transfer coefficients of MSWI' on page 30. The uncertainty of the elements in the residual material is obtained from by the formula for addition (Wilkinson-Fenton approximation) from the mean and GSDs of the three contributing solid outputs.

### **3.2.8 Waste-specific ancillary materials**

Several ancillary materials are used in the MSWI for the flue gas scrubber system and the waste water treatment. Tab. 3.2 shows the input of ancillary materials during average MSWI operation, based on 7 Swiss plants (unreferenced values in Zimmermann et al. 1996:B.114).

Tab. 3.2 Input of ancillary materials during average MSWI operation

Solution	Concentration	Input of diluted solution g/kg waste
NaOH	30%	7
CaO	95%	6
HCl	30%	0.04
FeCl <sub>3</sub>	40%	0.1
TMT 15	15%	0.1
Polyelectrolyte	1	0.02
Ammonia for DeNO <sub>x</sub>	25%	2

1 Dilution in use unknown

2 cf. chapter 3.3.1 'Fuel-NO<sub>x</sub> and thermal NO<sub>x</sub>' on page 40

In accordance with chapter 2.9 'Allocation of ancillary materials' on page 24 the consumption of ancillary materials is allocated to the causing agents in the waste as far as possible. In the following it is discussed what kind of agents cause the consumption of ancillary materials. Specific consumption factors are devised from average operation figures that can be used to calculate waste-specific consumption of ancillary materials. These specific consumption factors are shown in Tab. 3.3 on page 36.

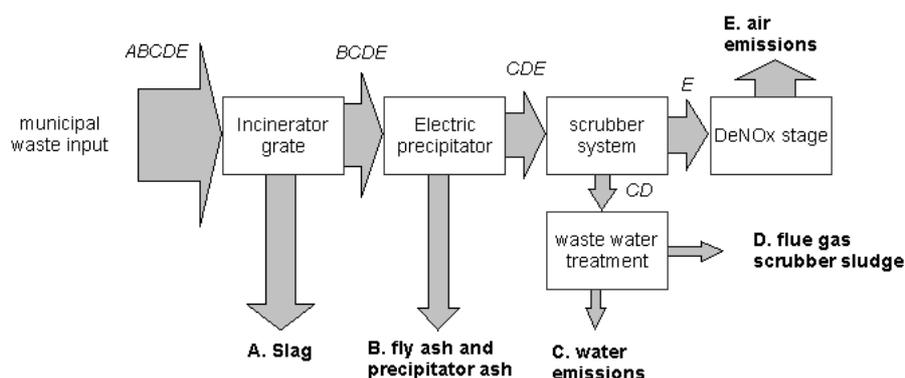


Fig. 3.2 Material throughput in a MSWI (not to scale)

### Sodium Hydroxide (NaOH) for SO<sub>2</sub> removal.

An aqueous solution of NaOH is used in the second, alkaline stage of the wet scrubber to transfer SO<sub>2</sub> to the scrubber liquid. Although the use of NaOH increases the transfer rates of other acids such as HCl and HF as well, the prime motivation of using NaOH is the removal of SO<sub>2</sub>. Most of the other acids are already removed in the *first* stage of the wet scrubber using only water. However, excess NaOH can also help to remove other acids.

A stoichiometric share of NaOH is allocated to the sulphur removed in the scrubber system for neutralising SO<sub>2</sub>. Two moles of NaOH are necessary to transfer SO<sub>2</sub> in air to sulfate (SO<sub>4</sub><sup>2-</sup>) in water. The excess NaOH is allocated to the acid produced in the scrubber by nitrogen (as HNO<sub>3</sub>), phosphorus (as H<sub>3</sub>PO<sub>4</sub>), and the halogens chlorine, bromine and fluorine (as HCl, HBr, HF).

The sulphur removed in the scrubber system (described by flow CD in Fig. 3.2) is all the sulphur in the waste except the amount of sulphur that was removed to slag (flow A), to precipitator ash (flow B) and to air (flow E).

### **Burnt lime (CaO) for neutralising acids**

The scrubber liquids are transferred to a wastewater treatment facility. Acids present in the scrubber liquids must be neutralised. An 95% aqueous solution of calcium hydroxide (Ca(OH)<sub>2</sub>) is used. This solution is a mixture of burnt lime (CaO) and water.

Approximately 95% of the acid load in scrubber liquids during average operation of the MSWI is hydrochloric acid (HCl). Since the aim of this report is to inventory several different waste compositions and not just average waste, HCl can not be seen as the sole causing agent for CaO consumption, but *all* acids in the untreated wastewater together. Acidifying elements in the untreated wastewater are nitrogen (as HNO<sub>3</sub>), phosphorus (as (H<sub>3</sub>PO<sub>4</sub>), and the halogens chlorine, bromine and fluorine (as HCl, HBr, HF)<sup>32</sup>.

### **Additional consumption factors for iodine**

No iodine measurements are available for the average municipal waste. The level of iodine in average municipal waste is probably very low. However, iodine leads to acid production from iodic acid (HIO<sub>3</sub>) in the scrubber, which needs to be neutralised. For the *average* waste, this acid production is negligible. However, for a *specific* waste rich in iodine the acid production can be significant. In order to assign correct NaOH and CaO consumption for such a waste, additional consumption factors are derived from the other halogens, heeding the different atomic weights. I.e. NaOH consumption for bromine (atomic weight = 79.9 g/mol) is calculated to be 0.47 kg per kg Br in the scrubber. Accordingly, the NaOH consumption for iodine (atomic weight = 126.9 g/mol) is 0.3 kg per kg I in the scrubber (= 0.47\*79.9/126.9). Similarly, a CaO consumption factor for iodine is calculated. See Tab. 3.3.

### **Hydrochloric acid (HCl) for pH control**

Small amounts of hydrochloric acid are used in the wastewater treatment for pH control. The consumption of hydrochloric acid is allocated to the total mass of waste elements transferred to the untreated wastewater (flow CD in Fig. 3.2).

### **Iron chloride (FeCl<sub>3</sub>) for precipitation of heavy metals**

Iron chloride is used to precipitate heavy metals during wastewater treatment. Only small amounts are used. To obtain some waste-specificity, the consumption of iron chloride is allocated to the total mass of heavy metals present in the untreated wastewater.

### **TMT15 for precipitation of mercury and cadmium**

TMT15 is a commercial name for a 15% aqueous solution of sodium trimercapto-S-triazine (Na<sub>3</sub>C<sub>3</sub>N<sub>3</sub>S<sub>3</sub>). TMT15 is used specifically to remove mercury (Hg) and cadmium (Cd) from the scrubber liquid. Other heavy metals are removed as chlorides or hydroxides and do not need a special agent to be precipitated. Accordingly the consumption of TMT15 is allocated to the presence of mercury and cadmium in the untreated wastewater. Allocation is calculated based on the moles of mercury and cadmium present in the untreated wastewater.

<sup>32</sup> As a consequence the incineration of e.g. polyvinylchloride PVC is burdened with a CaO consumption through the formation of hydrogen chloride HCl and the incineration of polyvinylfluoride PVF through the formation of the acid hydrogen fluoride HF.

**Polyelectrolyte solution for flocculation**

Minor amounts of polyelectrolyte solution (essentially an aqueous solution of inorganic salts) are used for flocculation in the wastewater treatment. The consumption of hydrochloric acid is allocated to the total mass of waste elements transferred to the untreated wastewater.

**Ammonia and natural gas for DeNO<sub>x</sub> stage**

The consumption of ammonia and natural gas for the abatement of NO<sub>x</sub> is discussed in the chapter 3.3.1 'Fuel-NO<sub>x</sub> and thermal NO<sub>x</sub>' on page 40.

**Cement for ash solidification**

The boiler ash, precipitator ash and scrubber sludge are solidified with cement before being landfilled in a residual material landfill. A mixture of (residue + cement + water) of (50% + 20% + 30%) is assumed. The addition of cement and water is headed by a module 'solidifying cement in residual material landfill'. This increases the cumulated landfilled mass for residual material.

**SCR catalyst material**

First patented in 1959, the first successful commercial NO<sub>x</sub> SCR system was built by Hitachi Zosen in 1975. Since then, the technology has become well established world wide. Most catalysts consist of a mixture of vanadium (V<sub>2</sub>O<sub>5</sub>, active catalyst) and titanium (TiO<sub>2</sub>, used to disperse and support the vanadium). However, the final catalyst composition can consist of many active metals and support materials to meet specific requirements in each SCR installation. Base metal catalysts may also contain molybdenum or tungsten. The V<sub>2</sub>O<sub>5</sub> composition typically ranges between 1 and 5 w-% depending upon the content of the catalyst poison SO<sub>2</sub> in the flue gas. Tungsten trioxide (WO<sub>3</sub>) might be added as a co-catalyst/promoter in the cases where additional catalyst activity is needed. The catalyst shape is usually a honeycomb shaped metal or ceramic plate. The catalyst blocks are held in place by a stainless steel T-bar grid. The shape is aimed at minimising clogging from particulate matter. The best temperature range for SCR catalyst activity and selectivity is from 300° to 400°C. Ammonia is injected downstream from the combustion zone and upstream of the catalyst.

The catalyst undergoes slow deactivation over time. SCR catalyst poisons are arsenic, sodium and potassium (Khodayari 2001). Clogging or plugging – especially in high-dust SCR installations, i.e. upstream of the ESP – limits the useful lifetime of the catalyst. High sulphur fuels can increase formation of adhesive ammonium bisulphate from SCR ammonia and SO<sub>3</sub>, which can be created from SO<sub>2</sub> on the SCR catalyst. The thickness and hence the space velocity of the honeycomb blocks is typically specified for a three year lifetime. As the activity of the catalyst decreases, the amount of excess ammonia required for the desired conversion increases. An ammonia slip of 3 mg/Nm<sup>3</sup> is typical.

It is concluded that the useful lifetime of SCR catalysts is actually not dependent on the nitrogen load but rather on the position of the catalyst (high-dust or low-dust). However, it is clear that the motivation to install a SCR catalyst is to reduce NO<sub>x</sub> levels in the flue gas. Accordingly, the SCR material consumption can be allocated to the NO<sub>x</sub> in flue gas. The SCR materials are therefore headed in chapter 3.3.1 'Fuel-NO<sub>x</sub> and thermal NO<sub>x</sub>' on page 40.

Tab. 3.3 Synopsis of the allocation of auxiliary material consumption to various allocation recipients.

Allocated material	Allocation recipient (in scrubber)	Allocation key	Mass of allocation recipient in scrubber during average operation Grams	Specific consumption factors of auxiliary materials Kg per kg allocation recipient in scrubber
NaOH	S	mol	0.435	2.51
	N	acid production	0.01	2.69
	P	acid production	0.0288	2.43
	Cl	acid production	5.25	1.06
	Br	acid production	0.00104	0.47
	F	acid production	0.012	1.98
	I	acid production	- <sup>1</sup>	0.3
CaO	N	acid production	0.01	0.48
	P	acid production	0.0288	0.43
	Cl	acid production	5.25	0.19
	Br	acid production	0.00104	0.08
	F	acid production	0.012	0.35
	I	acid production	- <sup>1</sup>	0.053
	HCl	all elements	mass	6.3
FeCl <sub>3</sub>	all heavy metals	mass	0.069	0.58
TMT 15	Hg	mol	0.00244	3.55
	Cd	mol	0.001	6.34
Polyelectrolyte	all elements	mass	6.3	0.0032

<sup>1</sup> consumption factors for iodine calculated from other halogens heeding different atomic weights.

### Transportation of auxiliary materials

No information on the transportation modes and distances is available. Generic transportation distances to Switzerland are assumed for all materials (see ecoinvent 2000 methodology in Frischknecht et al. 2003a). For aqueous solutions the solvent water is transported as well. The transportation distances used are shown in Tab. 3.4. For lorries an average load factor of 50% is already heeded in the requested data modules. The inventoried transportation service is the product of 'distance × mass'.

**Tab. 3.4 Generic transportation modes and distances for auxiliary materials**

Material	Solution concentration	Transportation distance train in km	Transportation distance lorry 28t in km
NaOH	30%	600	50
CaO	95%	100	20
HCl	30%	200	50
FeCl <sub>3</sub>	40%	600	50
TMT 15	15%	600	50
Polyelectrolyte	1	600	50
Cement for solidification	–	100	20
Ammonia in DeNO <sub>x</sub>	25%	600	50
TiO <sub>2</sub> for SCR catalyst	–	200	100
V <sub>2</sub> O <sub>5</sub> for SCR catalyst	–	200	100

1 dilution in use unknown. 100% used for calculation.

### Uncertainty estimates for auxiliary material demands

The GSD<sup>2</sup> value for the *specific consumption figures* of auxiliary material inputs is shown in Tab. 3.5. The same value is used for all auxiliary materials. Since the waste-specific material input depends on the amount of chemical elements reaching the scrubber, the final GSD values of the inventoried material exchanges also depend on the GSD of the concentration of chemical elements in the waste and the GSD of transfer coefficients.

**Tab. 3.5 GSD<sup>2</sup> values for specific consumption figures of auxiliary material inputs for MSWI**

Exchange	GSD <sup>2</sup> value	Pedigree codes	Comment
Auxiliary material	124%	(1,3,3,1,1,5)	Basic uncertainty of 105%; unreferenced values in Zimmermann et al. 1996, pages B-97 and B-116, checked with current figures of MSWI Buchs (GEKAL 2002)

### Uncertainty estimates for cement consumption for ash solidification

The fly ash and scrubber sludge are solidified with cement before being landfilled in a residual material landfill. A mixture of residue+cement+water of 50%+20%+30% is assumed, i.e. per kilogram of residue 400 grams of cement and 600 grams of water are needed. The GSD values for these specific inputs of cement and water per mass of residue are estimated to be 110%. The waste-specific mass of residue has a waste-specific uncertainty. The waste-specific demand of cement and water is obtained by multiplication of the specific input figures and the waste-specific mass of residue. The GSD values for the cement and water demand are obtained accordingly.

Uncertainties in the contribution of heavy metals from cement to the solidified ash residue is discussed along with the mean values of heavy metals in cement in section 'Uncertainty estimates for auxiliary material demands' on page 37.

### Uncertainty estimates for transportation of auxiliary materials

The needed transportation service is the product of distance and mass. The GSD value of the transportation service, depends on the GSD values for the masses of the auxiliary materials, which in turn depend on waste composition. Thus, no constant GSD value for the transportation services can be given. This issue is further discussed in part I of this report.

### 3.2.9 Waste-specific burdens from fuel-NO<sub>x</sub>

MSWIs produce NO<sub>x</sub> and other nitrogen containing air emissions. As outlined in chapter 3.3.1 'Fuel-NO<sub>x</sub> and thermal NO<sub>x</sub>' on page 40, only a part of the produced NO<sub>x</sub> can be allocated to the nitrogen contained in the waste. In this report, it is assumed that 50% of all NO<sub>x</sub> is formed from nitrogen from waste (fuel-NO<sub>x</sub>) and 50% from nitrogen in combustion air (thermal-NO<sub>x</sub> from N<sub>2</sub>). The emissions and burdens that are allocated to the nitrogen in waste (or more precisely the nitrogen in waste that is transferred to the *raw gas*) are shown in Tab. 3.6. These are the factors used in the model to determine burdens from waste-specific nitrogen. The calculation of these figures and their uncertainty is explained in chapter 3.3.1 'Fuel-NO<sub>x</sub> and thermal NO<sub>x</sub>' on page 40. Please note, that the emission factors in Tab. 3.6 are *not* transfer coefficients, but are given in grams of *emitted compound* per kilogram of nitrogen in raw gas and hence do not add up to 1000 grams.

In Tab. 3.6 the emission factor for N<sub>2</sub> per kilogram of nitrogen in raw gas is calculated by difference from the nitrogen-containing emissions<sup>33</sup>. The emission of elemental nitrogen to air (N<sub>2</sub>) is not inventoried in the ecoinvent database. The factor is displayed in Tab. 3.6 to show that most of the air-bound nitrogen (i.e. more than 98%) is converted to harmless elemental nitrogen. For vanadium pentoxide (SCR catalyst material) no data module is available in the ecoinvent 2000 database. It is heeded with chromium trioxide as proxy, since these two metals are sometimes mined together and since they are neighbouring elements in the periodic table (V<sub>2</sub>O<sub>5</sub> has 44% oxygen, while CrO<sub>3</sub> has 48% oxygen).

**Tab. 3.6 Waste-specific emissions from fuel-NO<sub>x</sub> and abatement of fuel-NO<sub>x</sub> per kg of N in raw gas. Emissions factors with a grey background are for emissions that occur only waste-specifically.**

Waste-specific exchanges per kg N in raw gas		Type of NO <sub>x</sub> abatement technology				Burden per kg N in raw gas	GSD
		without DeNO <sub>x</sub>	SNCR	SCR-high dust	SCR-low dust	Weighted mix	See text page 42
Share in technology mix		13.8%	29.4%	32.2%	24.6%	100%	
<b>Emissions to air</b>							
Elemental nitrogen N <sub>2</sub> #	g					<b>982.62</b>	#
Nitrogen oxides NO <sub>x</sub>	g	106.06	20.30	20.30	20.30	<b>32.14</b>	207%
Ammonia NH <sub>3</sub>	g	0.12	0.91	0.91	0.91	<b>0.80</b>	207%
Nitrous oxide N <sub>2</sub> O	g	3.03	6.06	3.64	3.64	<b>4.27</b>	207%
Cyanide CN	g	0.91	0.91	0.91	0.91	<b>0.91</b>	207%
<b>Material inputs</b>							
Ammonia (100%)	g	-	75.76	50.51	50.51	<b>50.96</b>	201%
Natural gas	MJ	-	-	-	20.20	<b>4.97</b>	201%
TiO <sub>2</sub> for SCR catalyst	g	-	-	3.416	1.46	<b>1.46</b>	247%
V <sub>2</sub> O <sub>5</sub> for SCR catalyst <sup>1</sup>	g	-	-	0.0697	0.0299	<b>0.03</b>	247%

# not inventoried in ecoinvent database.

1 not inventoried in ecoinvent database. Chromium trioxide was inventoried as a proxy.

### 3.2.10 Waste heat emissions

The energy content of the waste is liberated in the MSWI. Some of the energy is converted to thermal and electric energy (see the section 'Energy production' on page 6). Within the ecoinvent 2000 project, waste heat emissions of electricity are inventoried in the electricity-consuming process, while the full

<sup>33</sup> I.e. this is not a *measurement* of N<sub>2</sub> in the flue gas, but a calculation referring to the fate of waste-specific nitrogen *alone*. A measurement of N<sub>2</sub> in the flue gas would include a lot of N<sub>2</sub> from combustion air.

waste heat emissions of heat sources are inventoried in the heat-producing process. To avoid double counting no waste heat emissions must be inventoried in the electricity production process. Accordingly, the waste heat generated in MSWI is the energy content of the waste *excluding the amount of energy leaving the plant as net produced electricity*<sup>34</sup>. See also Fig. 3.3. No distinction is made between waste heat from renewable sources and non-renewable sources (Frischknecht et al. 2003a).

It is estimated from air and water throughputs that 83% of all waste heat is emitted to air and 17% to water. The GSD for these percentages is estimated to be small (102%).

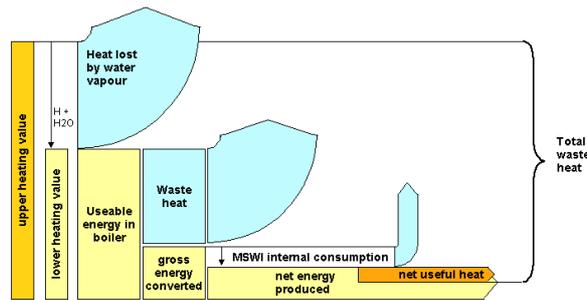


Fig. 3.3 Scheme of waste heat balance in the MSWI.

### 3.2.11 Electricity demand for magnetic iron separation

The separation of iron from slag is inventoried as a waste-specific energy demand. Average MSWI slag contains about 10 w-% iron (Stark 1993, Goetz 1989, EKESA 1992, Rey 1992). This figure is influenced by tin collection. About 60% of the iron in slag are in magnetic form (Zeltner 1992). For the magnetic descraping of slag about 2.5 kWh/t average slag are needed (unreferenced value in Menard & Zimmermann 1995:VII.4). A specific electricity demand of 0.042 kWh<sub>el</sub>/kg of iron removed is calculated (presumed that the magnetic iron is removed quantitatively from slag by this procedure).

In the input interface for the waste composition data, the share of magnetic iron in the specified waste can be entered. This figure will be used to determine the waste-specific mass of magnetic iron in slag, which will be removed and recycled (see also chapter 2.8 'Allocation of iron scrap production' on page 24). Not all Swiss MSWIs feature a magnetic separation. Hence, in the average technology mix for MSWIs only about 50% of the magnetic iron is removed (see chapter 'Recycling efforts' on page 6).

The electricity demand will be added to the average process-specific electricity demand of 0.144 kWh electricity per kilogram waste (see chapter 3.3.8 'Energy consumption' on page 48). Strictly this is a double counting, since the average energy for descraping is already contained in the process-specific energy demand. In reality the contribution from descraping is small (<0.3%), so for the average waste this is negligible. For a waste of 100% magnetic iron the energy demand is increased by 0.021 kWh<sub>el</sub>/kg (+14%).

<sup>34</sup> Wastes with low heating values are not able to produce any net electricity in the MSWI. For the processing of such wastes an additional energy source is necessary. The electricity demand is inventoried with the module 'electricity from MSWI' (with default allocation of 0% of average MSWI burdens). This electricity demand leads to a waste heat emission, which is added to any waste heat from the waste itself.

### 3.3 Process-specific burdens

Process-specific burdens indicate emissions and expenditures that are *constant* for every waste composition. See chapter 2.10 'Process-specific burdens' on page 25.

#### 3.3.1 Fuel-NO<sub>x</sub> and thermal NO<sub>x</sub>

When waste is incinerated, formation of NO<sub>x</sub> can occur by three different types of reactions:

1. By oxidation of the nitrogen present in the waste material (generally called 'fuel-NO<sub>x</sub>', equivalent to 'waste-specific NO<sub>x</sub>' in this report).
2. By direct oxidation of the elemental nitrogen present as N<sub>2</sub> in the combustion air (generally called 'thermal-NO<sub>x</sub>', equivalent to 'process-specific NO<sub>x</sub>' in this report).
3. By indirect oxidation of N<sub>2</sub> present in combustion air via reaction with free hydrocarbon radicals (CH·) formed during the combustion process followed by oxidation of the resulting nitrogen atoms or nitrogen radicals (so called 'prompt-NO<sub>x</sub>'). Prompt-NO<sub>x</sub> plays only a marginal role in controlled incineration.

In usual combustion processes, thermal-NO<sub>x</sub> formation occurs above 1200°C. Since the combustion temperature in MSWI is usually below 1000°C, thermal-NO<sub>x</sub> formation seems unlikely. However due to the inhomogeneous nature of the incinerated waste and varying operation conditions MSW experts expect that thermal-NO<sub>x</sub> can account for up to 70% of the total formed NO<sub>x</sub> in the raw gas (Zimmermann et al. 1996:B.88). Kremer et al. (1998) even consider an average thermal-NO<sub>x</sub> share of 100%, i.e. that the NO<sub>x</sub> concentration in the flue gas is completely independent of waste nitrogen input. Other sources consider a share of 20 to 30% of thermal NO<sub>x</sub> as typical for MSWI (CORINAIR 1999). For this report a share of 50% of thermal-NO<sub>x</sub> and share of 50% fuel-NO<sub>x</sub> is assumed.

The raw data for nitrogen-related burdens (emissions and input materials for NO<sub>x</sub> abatement) are shown in Tab. 3.7 (Zimmermann et al. 1996:B.89 based on BUWAL 1995a and <sup>35</sup>). Materials consumption for the DeNO<sub>x</sub> SCR catalyst (high-dust and low-dust) were estimated from catalyst geometry, catalyst lifetimes of 1.5 years (3.5 years for low-dust, respectively) and a V<sub>2</sub>O<sub>5</sub> concentration of 2%, based on descriptions of MSWI SCR facilities in Hinton (2000) and SKC (2000).

Compared to a plant without DeNO<sub>x</sub> installation, the different types of DeNO<sub>x</sub> technologies (SNCR, SCR) reduce the NO<sub>x</sub> emissions by 81%. In contrast, ammonia emissions increase for DeNO<sub>x</sub> technologies, due to ammonia slip from the ammonia addition for NO<sub>x</sub> reduction. Also N<sub>2</sub>O emissions increase, formed by the reducing conditions in the DeNO<sub>x</sub> stage. Cyanide emissions are unaffected by DeNO<sub>x</sub> technologies. All DeNO<sub>x</sub> technologies require an 25% aqueous ammonia solution<sup>36</sup> as a reaction partner in the NO<sub>x</sub> reduction:



SCR high-dust DeNO<sub>x</sub> facilities also need a natural gas input to heat up the flue gas to DeNO<sub>x</sub> operation temperature after the wet scrubber.

<sup>35</sup> Personal communication with M. Brunner, Technikum Rapperswil, in March 1996

<sup>36</sup> Sometimes also referred to as NH<sub>4</sub>OH = NH<sub>3</sub> + H<sub>2</sub>O, which however would represent a 49% aqueous ammonia solution.

**Tab. 3.7 Raw data for nitrogen-related burdens for different NO<sub>x</sub>-abatement technologies. Total burdens per kilogram of waste for average operation and average waste**

	per kg waste	Type of NO <sub>x</sub> abatement technology			
		without DeNO <sub>x</sub>	SNCR	SCR-high dust	SCR-low dust
<b>Emissions</b>					
Nitrogen oxides NO <sub>x</sub>	g	2.1	0.402	0.402	0.402
Ammonia NH <sub>3</sub>	g	0.0012	0.018	0.018	0.018
Nitrous oxide N <sub>2</sub> O	g	0.03	0.06	0.036	0.036
Cyanide CN	g	0.009	0.009	0.009	0.009
<b>Material inputs</b>					
Ammonia (100%)	g	–	1.5	1	1
Natural gas	MJ	–	–	–	0.4
TiO <sub>2</sub> for SCR catalyst	g	–	–	0.0676	0.029
V <sub>2</sub> O <sub>5</sub> for SCR catalyst	g	–	–	0.00138	0.000591

For emissions of cyanide CN and nitrous oxide N<sub>2</sub>O carbon- and nitrogen-containing precursors are necessary, which originate from waste nitrogen. Hence, N<sub>2</sub>O emissions and CN emissions are completely allocated to waste nitrogen. Ammonia emissions in plants without any DeNO<sub>x</sub> facility are completely allocated to waste nitrogen. All emissions with a grey background in Tab. 3.8 are completely allocated to waste nitrogen. All other emissions and burdens are allocated between fuel-NO<sub>x</sub> (nitrogen from waste) and thermal-NO<sub>x</sub> (process-specific emission). As pointed out above, a share of 50% of thermal-NO<sub>x</sub> and share of 50% fuel-NO<sub>x</sub> is assumed. This results in the process-specific emissions (from allocation to thermal-NO<sub>x</sub>) as shown in Tab. 3.8. Each kilogram of waste will bear these process-specific emissions irrespective of its composition.

The *transportation* of the consumed process-specific masses of ammonia and SCR materials for DeNO<sub>x</sub> (Tab. 3.7) is inventoried in the process-specific module using the same transportation distances and solution concentrations as for the waste-specific auxiliary materials (cf. section 'Transportation of auxiliary materials' on page 36).

**Tab. 3.8 Process-specific emissions from thermal-NO<sub>x</sub> and abatement of thermal-NO<sub>x</sub> per kg of waste**

	per kg waste	Type of NO <sub>x</sub> abatement technology				Burden per kg waste	GSD
		without DeNO <sub>x</sub>	SNCR	SCR-high dust	SCR-low dust		
Share in technology mix		13.8%	29.4%	32.2%	24.6%	100%	
<b>Emissions</b>							
Nitrogen oxides NO <sub>x</sub>	g	1.05	0.201	0.201	0.201	<b>0.318</b>	150%
Ammonia NH <sub>3</sub>	g	0	0.009	0.009	0.009	<b>0.00776</b>	150%
Nitrous oxide N <sub>2</sub> O	g	0	0	0	0	<b>0</b>	150%
Cyanide CN	g	0	0	0	0	<b>0</b>	150%
<b>Material inputs</b>							
Ammonia (1.00)	g	–	0.75	0.5	0.5	<b>0.5045</b>	142%
Natural gas	MJ	–	–	–	0.2	<b>0.0492</b>	142%
TiO <sub>2</sub> for SCR catalyst	g	–	–	0.0338	0.0145	<b>0.0144</b>	195%
V <sub>2</sub> O <sub>5</sub> for SCR catalyst <sup>1</sup>	g	–	–	0.00069	0.000296	<b>0.000295</b>	195%

<sup>1</sup> not inventoried in ecoinvent database. Chromium trioxide was inventoried as a proxy.

The difference of emissions and burdens between Tab. 3.7 (= all emissions) and Tab. 3.8 (= emissions allocated to thermal-NO<sub>x</sub>) are allocated to fuel-NO<sub>x</sub>, i.e. to the nitrogen contained in the waste. To be more precise, these burdens are allocated to the nitrogen in waste that is transferred to the raw gas and actually can form NO<sub>x</sub> air emissions at all. In average operation with average MSW 98.9% of the nitrogen in waste is transferred to the raw gas<sup>37</sup>. The average nitrogen content in MSW is 10 grams N per kg of MSW<sup>38</sup>. The fuel-NO<sub>x</sub> is therefore caused on average by 9.89 grams of N in the raw gas. From that information, emission factors and burden factors *per kg N in the raw gas* are calculated (cf. Tab. 3.6 on page 38). The factors are then used to calculate waste-specific, non-average burdens and emissions based on the nitrogen content of the particular waste under consideration and the amount of nitrogen transferred to the raw gas<sup>39</sup>.

### Uncertainty estimates for nitrogen emissions and DeNO<sub>x</sub> burdens.

The uncertainty of the *process-specific* nitrogen emission figures given in Tab. 3.8 depend on several values. Emissions and material inputs were taken from (Zimmermann et al. 1996) and are based on literature data from Switzerland and Germany. The uncertainty for these figures is shown as GSD<sup>2</sup> in Tab. 3.9. However, these GSD<sup>2</sup> values cannot be applied directly to the figures in Tab. 3.6 and Tab. 3.8. Other uncertain figures are involved to calculate the mean values in those tables.

Tab. 3.9 GSD<sup>2</sup> values for raw data on emissions and material inputs for DeNO<sub>x</sub> facilities

Exchange	GSD <sup>2</sup> value	Pedigree codes	Comment
N-Emissions	152%	(1,3,3,2,1,2)	Basic uncertainty of 150%; literature data from Switzerland and Germany
Ammonia and natural gas input	113%	(1,3,3,2,1,2)	Basic uncertainty of 105%; literature data from Switzerland and Germany
SCR materials	314%	(4,5,2,n.a.,1,5)	Basic uncertainty of 300%; estimate from SCR geometry, operation, lifetimes.

The first uncertain assumption is the share of thermal-NO<sub>x</sub> of the total NO<sub>x</sub>. A mean value of 50% was assumed. As pointed out in chapter 3.3.1 'Fuel-NO<sub>x</sub> and thermal NO<sub>x</sub>' there is considerable uncertainty on this figure. However, the value cannot possibly exceed 100%. The maximum possible GSD value is therefore  $\sqrt{100\%/50\%} = 141\%$ , which is used here, considering the substantial differences in literature. The combination of this value with the values in Tab. 3.9 give the GSD values for the *process-specific* burdens in Tab. 3.8; i.e. 150% for emissions, 142% for ammonia and natural gas input, and 195% for SCR materials.

For the *waste-specific* nitrogen-related burdens in Tab. 3.6 on page 38 still another uncertain value is used in the calculation: the average amount of nitrogen in average waste. A value of 10 grams N per kg of MSW is used. The GSD for that figure was determined according to chapter 2.5 'Uncertainty of waste composition data' on page 17 to be 183%. The combination of this value with the values in Tab.

<sup>37</sup> The nitrogen in waste transferred to the raw gas is given by the amount of nitrogen which is not transferred to slag (100% – TK<sub>N,slag</sub>).

<sup>38</sup> Newer sources give a value of 3.1 g/kg (Hellweg 2000:61). Erichsen & Hauschild 2000 cite a range of 2 of 13 g/kg. The original value of 10 g/kg from (Zimmermann et al. 1996:B.89) is used here since the raw data given in Tab. 3.7 were not changed. (Corinair 1999) gives a emission factor of 1.8g NO<sub>x</sub> / kg MSW for modern plants without DeNO<sub>x</sub>, which corresponds well with the value of 2.1 g NO<sub>x</sub> / kg MSW in Tab. 3.7.

<sup>39</sup> Remember that there are inert wastes that are completely transferred to slag. Nitrogen contained in such wastes will not be transferred to the raw gas and will accordingly not lead to any nitrogen emissions or DeNO<sub>x</sub> efforts.

3.9 and the uncertainty of the share of fuel-NO<sub>x</sub> give the GSD values for the waste-specific burdens in Tab. 3.8; i.e. 207% for emissions, 201% for ammonia and natural gas input, and 247% for SCR materials. To obtain a GSD value, the component GSD values are combined *as if* the respective mean values were *multiplied*. This is mathematically not exact, as the mean values are obtained by *division* (with the nitrogen content in average waste), not by multiplication. However, no exact formula to estimate uncertainty values for divisions is available.

To calculate the inventoried exchanges for any specific waste, an additional uncertain figure is needed: the amount of nitrogen in raw gas for that specific waste. The uncertainty for that figure is dependent on the uncertainty of the waste composition and the uncertainty of the transfer coefficient. These figures change for every waste and it is not possible to give final GSD figures for the waste-specific nitrogen-related burdens. The inventoried exchanges are obtained by multiplication of the values in Tab. 3.9 with the amount of nitrogen in raw gas. The final GSD is obtained by using the formula for multiplication.

Other uncertain data used in these calculations are the flue gas volume per kilogram of average waste and the transfer coefficient for nitrogen to raw gas for average waste. The introduced uncertainties of these figures are comparatively small and need not to be discussed here.

### 3.3.2 Volatile organic compounds VOC

Influencing factors for the level of VOC in the flue gas are temperature, residence time, mixing with secondary combustion air and combustion air control. A few VOC species can be determined and are inventoried specifically. Other VOC species are inventoried with the sum parameter NMVOC.

For incinerator lines older than 1985 an average value of 30 mg NMVOC/Nm<sup>3</sup> flue gas is assumed, for lines younger than 1985 an average value of 3.5 mg NMVOC /Nm<sup>3</sup> flue gas is assumed (Zimmermann 1996:B.91). With 27% of all incinerator lines being older than 1985 and an average flue gas volume of 6 Nm<sup>3</sup>/kg waste, a process-specific NMVOC emission of 64 mg per kg waste is calculated.

Some measurements of VOC species from Swiss and German MSWI are available<sup>40</sup>. Based on this data, process specific emissions factors were calculated with plants with and without SCR stage (Zimmermann 1996:B.92). The differentiated NMVOC species in Tab. 3.10 are 1.28 mg/kg waste and must be subtracted from the total NMVOC emission factor of 64 mg per kg waste derived above to avoid double counting. The process-specific, undifferentiated NMVOC emission is 62.65 mg per kg waste.

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<sup>40</sup> Personal communication with M. Brunner, Technikum Rapperswil in March 1996.

Tab. 3.10 Emission factors for VOC species

	SNCR or without DeNOx	with SCR	Technology mix
	µg/kg waste	µg/kg waste	µg/kg waste
Methane	7200	5760	<b>6382</b>
Benzene	480	384	<b>425</b>
Toluene	960	768	<b>851</b>
Pentachlorobenzene	3.27	1.79	<b>2.43</b>
Hexachlorobenzene	1.32	0.70	<b>0.968</b>
Pentachlorophenol	0.534	0.0615	<b>0.266</b>
Benzo(a)pyrene	0.01205	0.0081	<b>0.0098</b>
Sum of differentiated NMVOC	–	–	<b>1280</b>
Sum of total NMVOC	–	–	<b>63'930</b>
Sum of undifferentiated NMVOC	–	–	<b>62'650</b>

In the ecoinvent database methane emissions from fossil carbon and biogenic carbon are discerned. Since wastes with varying contents of fossil and biogenic carbon are inventoried here, the amounts of fossil and biogenic methane will change as well, e.g. a plastic waste will lead to fossil methane and a paper waste to biogenic methane. So, this emission cannot be placed in the data module for (constant) process-specific emissions but must be inventoried differently for each waste. Emissions of methane are hence inventoried with the *waste-specific* air emissions. The sum of fossil and biogenic methane for any waste will however remain constant for a given MSWI technology mix.

### 3.3.3 Dioxins and furans

Dioxins and furans can be formed and destroyed at high temperatures in the combustion zone. But according to numerical calculations the formation is unlikely with good mixing of air and fuel. Experiments indicate that formation of dioxins and furans in the combustion zone is due to the presence of large, not completely burned particles.

The level of dioxins and furans present in the incoming waste is very low. Therefore, the incomplete destruction of dioxins in the waste can not account for more than a small fraction of the dioxins typically present in the flue gas (Erichsen & Hauschild 2000).

During cooling of the flue gas after the combustion zone, a catalytic formation of dioxins can take place on fly ash particles from which the dioxin may later evaporate. The catalytic formation can follow two different mechanisms: *de novo synthesis* and *synthesis from precursors*. In *de novo* synthesis dioxins are formed from carbonaceous particulate matter and inorganic halogens in the fly ash, catalysed by metal ions, primarily Cu<sup>II+</sup> ions. *De novo* synthesis is fastest at temperatures between 250°C and 350°C. At temperatures lower than 250-300°C, dioxins are predominantly retained on the fly ash. In *synthesis from precursors* dioxins and furans are formed on the fly ash by surface-catalysed reactions of precursors through a three step mechanism

1. Production of Cl<sub>2</sub> from a metal-catalysed reaction of HCl and oxygen (Deacon Process).
2. Chlorinating of aromatic rings through substitution reactions with Cl<sub>2</sub>.
3. Formation of dual ring structures by a second, metal-catalysed reaction.

The formation of dioxin requires the presence of carbon, inorganic chlorine and metal ions of various elements (e.g. Cu, Fe, Pb or Ni). All of these elements will be present in the normal waste in such large amounts that the exact extent of their presence is not a limiting factor in dioxin formation. Several studies have shown that there is no correlation of dioxin formation and the chlorine input in the waste (Mark et al. 1994b, Rigo et al. 1995, Kanters & Louw 1993). Dioxin formation is more dependent on process specific parameters like carbon content of fly ashes, residence time of ashes in a 300–400°C temperature range, oxygen content etc. A decrease of the chlorine content in the waste will have no perceivable effect on dioxin formation. Accordingly, dioxin emissions are treated as *process-specific* emissions in this report. Figures from (Zimmermann et al. 1996:B.94) are adopted based on (BUWAL 1995a) and <sup>41</sup>.

**Tab. 3.11 Emission factors for dioxins (TEQ = Toxicity equivalents)**

	ng TEQ per Nm <sup>3</sup> of flue gas	ng TEQ per kg waste (with 6 Nm <sup>3</sup> flue gas per kg waste)
Without DeNOx	6	<b>36</b>
With DeNOx	0.5	<b>3</b>

Dioxins have probably been the major cause for the bad public reputation of MSWI as excessive air polluting facilities. Indeed, around 1980 MSWIs were responsible for about 75% of all dioxin emissions in Switzerland. However, Switzerland has little heavy industry facilities so it is easy to reach a high rank. Today, however, the situation has changed drastically. MSWIs contribute but a small part in the total dioxin emissions. The biggest single polluter today are households, by open fires and illegal burning of waste (Studer 1999).

Regarding the cumulated burden of waste disposal (LCIA results), the dioxin emissions from a modern MSWI usually play only a minor role. More important hazards to health are originating from leachate emissions from landfilled incinerator remains.

<sup>41</sup> Personal communication with M. Brunner, Technikum Rapperswil in March 1996.

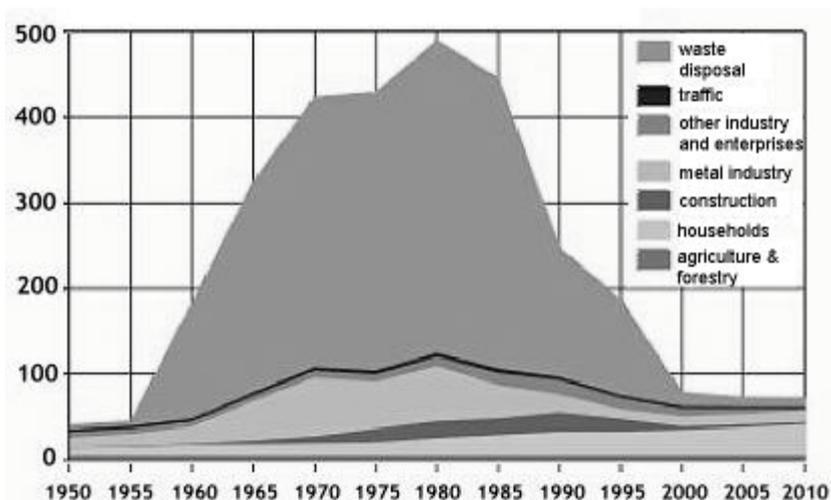


Fig. 3.4 Dioxin emissions in I-TEQ grams in Switzerland from 1950 to 2010 (BUWAL 1999)

### 3.3.4 Particulate Matter PM

The level of particulate matter in the flue gas is mainly determined by the flue gas treatment, especially the electrostatic precipitator. Since virtually all plants in Switzerland are fitted with an ESP, only one emission factor is devised for all Swiss plant technologies. An average value of 1 mg TSP/Nm<sup>3</sup> is assumed based on literature sources (Zimmermann et al. 1996:B.120, Zürcher et al. 2001). The permitted threshold limit value in Switzerland is 10 mg TSP/Nm<sup>3</sup> (LRV 1985). With an average flue gas volume of 6 Nm<sup>3</sup>/kg waste a process-specific TSP emission of 6 mg/kg waste is calculated. Data in the CEPMEIP database on PM for the source code 510000 (incineration of municipal solid waste) suggests that *all* particulate matter from MSWI is below 2.5µm (CEPMEIP 2002). Measurements for 4 Swiss MSWIs show that over 99% of the PM<sub>10</sub> is PM<sub>2.5</sub> (Zürcher et al. 2001). It is concluded here that of the total TSP, 100% is PM<sub>10</sub> and 99.5% is PM<sub>2.5</sub>. The emission factors in Tab. 3.12 are inventoried.

Tab. 3.12 Emissions factors for different size fractions of particulate matter

		Size fraction	Emission factor mg/kg waste
Fine particles	PM <sub>2.5</sub>	99.5%	<b>5.97</b>
Coarse matter	PM <sub>2.5-10</sub>	0.5%	<b>0.03</b>
Large particles	PM <sub>&gt;10</sub>	0	<b>0</b>
Total suspended particles	TSP	100%	6

### 3.3.5 Carbon monoxide CO

The absence of carbon monoxide is an indicator of optimal incineration management. In the usual range of oxygen content (9 – 11%) CO concentrations are independent of excess air. With oxygen contents in combustion air below 6% and above 12% carbon monoxide is formed by incomplete oxidation. Decisive factors for the level of CO in the flue gas are residence time, temperature and degree of mixing with secondary combustion air. Fluctuations of CO are usually caused by inhomogeneous fuel. There is no correlation between the carbon content in the waste and the amount of CO emitted. Therefore, carbon monoxide is treated as a process-specific emission in this report.

However, in the ecoinvent database carbon monoxide emissions from fossil carbon and biogenic carbon are discerned. Since wastes with varying contents of fossil and biogenic carbon are inventoried here, the amounts of fossil and biogenic CO will change as well, e.g. a plastic waste will lead to fossil CO and a paper waste to biogenic CO. So, this emission cannot be placed in the data module for (constant) process-specific emissions but must be inventoried differently for each waste. Emissions of carbon monoxide are hence inventoried with the *waste-specific* air emissions. The sum of fossil and biogenic CO for any waste will however remain constant for a given MSWI technology mix.

Carbon monoxide levels are not influenced by the flue gas treatment. However, older incinerators have higher CO emission levels than more recent incinerator types. For incinerator lines older than 1985 an average value of 70 mg CO/Nm<sup>3</sup> flue gas is assumed, for lines younger than 1985 an average value of 25 mg CO/Nm<sup>3</sup> flue gas is assumed (Zimmermann 1996:B.90). From Fig. 2.3 it can be estimated that approximately 27% of all incinerator lines (by capacity) are older than 1985. With an average flue gas volume of 6 Nm<sup>3</sup>/kg waste a process-specific CO emission of 223 mg/ kg waste is calculated for the Swiss mixture of plants.

A CO emission factor of 223 mg/ kg waste is inventoried as a process-specific emission for each waste. According to the ecoinvent 2000 methodology, carbon in CO emissions must be subtracted from carbon in CO<sub>2</sub> emissions to maintain the carbon balance (Frischknecht et al. 2003a). Accordingly, CO<sub>2</sub> emissions from the waste are reduced in order to maintain carbon material balance. In the case of carbon-free wastes (e.g. glass, metals) a CO emission is inventoried, although no carbon is present in the waste. Strictly the requirement for an accurate carbon balance would result in negative CO<sub>2</sub> emissions in this case as compensation. However this makes no sense, and a CO<sub>2</sub> emission of zero is inventoried. I.e. for these types of waste the carbon balance is not strictly correct. With a carbon content of less than 484 mg per kg (0.0484 w-%) waste, not enough carbon is present to account for the inventoried upper confidence value of the CO emissions.

### 3.3.6 Uncertainty estimates for process-specific air emissions

Tab. 3.13 Uncertainty estimates for process-specific air emissions

Exchange	GSD <sup>2</sup> value	Pedigree codes	Comment
Dioxins, Benzo(a)pyrene	305%	(2,3,4,2,1,2)	Basic uncertainty of 300%; confidential data from 5 plants in Switzerland and Germany
Benzene, Toluene, Pentachlorobenzene, Hexachlorobenzene, Pentachlorophenol	157%	(2,3,4,2,1,2)	Basic uncertainty of 150%; confidential data from 5 plants in Switzerland and Germany
other NMVOC species	151%	(1,3,2,2,1,2)	Basic uncertainty of 150%; studies from Switzerland and Germany
NOx	152%	(1,3,3,2,1,2)	Basic uncertainty of 150%; literature data from Switzerland and Germany adapted to plant DeNOx technology mix
Ammonia	152%	(1,3,3,2,1,2)	Basic uncertainty of 150%; literature data from Switzerland and Germany adapted to plant DeNOx technology mix
Particulates <2.5	302%	(1,3,3,2,1,2)	Basic uncertainty of 300%; from total PM value from Swiss and German plants + PM fractions from measurements on Swiss plants
Particulates 2.5-10 and particulates >10	202%	(1,3,3,2,1,2)	Basic uncertainty of 200%; from total PM value from Swiss and German plants + PM fractions from measurements on Swiss plants

### 3.3.7 Water consumption

The MSWI uses water for quenching the slag, for the wet scrubber and for process steam. Since virtually all Swiss MSWI are equipped with these, a single water consumption value is devised for all plants. An allocation to the generated slag and flue gas masses would be possible, but regarding the minor importance of the water consumption a process-specific assignment per kilogram waste is chosen. A value of 1 litre water per kilogram waste is used (Zimmermann et al. 1996:B.104) and assumed to be potable tap water.

### 3.3.8 Energy consumption

All 28 Swiss MSWI utilise the energy contained in the waste to produce useful heat and/or electricity. On average, the gross heat generation efficiency is 24.4% and the gross electricity generation efficiency is 13% (BUWAL 2001f). Data on the internal consumption is available from the same source. An average internal consumption of 0.839 MJ heat per kilogram waste is calculated<sup>42</sup>. The average internal consumption of electricity is 0.144 kWh electricity per kilogram waste<sup>43</sup>.

If a specific waste (e.g. glass in MSWI) has not enough heat content to produce its own process energy, the process energy is provided from the surplus of average waste incineration. Since the

<sup>42</sup> Seven plants not reporting any heat consumption were excluded from this calculation.

<sup>43</sup> Two plants not reporting any electricity consumption were excluded from this calculation.

allocation on energy is 0%, this energy is available for free. To keep track of the total energy consumption of such wastes the additional consumed energy of low-calorific wastes is inventoried with two empty modules for heat and electricity.

It might be interesting to note, that there is quite some variation amongst the Swiss MSWI plants regarding energy consumption. The internal electricity consumption ranges between 0.1 and 0.342 kWh/kg waste with a GSD<sup>2</sup> value of 168%; internal heat consumption ranges between 0.005 and 3.69 MJ<sub>th</sub>/kg waste with a GSD<sup>2</sup> value of over 4000% (sic). However the goal of this study is to inventory the average Swiss MSWI plant, accordingly the average values are used. Uncertainty of energy consumption figures is derived from the Pedigree approach.

Energy statistics of MSWI sometimes also feature additional fossil energy consumption; e.g. 179 GWh/a or 0.23 MJ/kg waste for 2000 (BUWAL 2001g). This energy input is fossil energy from auxiliary boilers to maintain heat levels for the district heating system. This input has no association with the waste treatment service function of MSWI plants, but must be allocated to the heat generation and distribution of the district heating system. As the goal of this report is the waste treatment service function of MSWI, this energy input is not considered.

### 3.3.9 Infrastructure

The infrastructure data module contains the building and maintenance expenditures for the whole plant in kg and the land occupation and transformation figures per plant (not per kg waste). With an average feed of 100'000 tons of MSW per year, and an operational time of 40 years,  $2.5 \cdot 10^{-10}$  plant units are inventoried per kg waste.

#### Building materials

Building materials for building and maintenance of a MSWI plant of 100'000 t/a capacity were determined from two sources (Mauch 1992) and <sup>44</sup>. Material flows for construction, maintenance and demolition are shown in Tab. 3.14.

Tab. 3.14 Building materials for MSWI infrastructure

MSWI infrastructure	g/kg waste	Tons per MSWI life
High alloyed steel	0.02	<b>80</b>
Low alloyed steel	0.98	<b>3920</b>
Concrete cement	1.25	<b>4990</b>
Concrete gravel	8.75	<b>35'000</b>
Bitumen	0.2	<b>800</b>
Sand	2.5	<b>10'000</b>
Reinforced concrete dismantling (including 3% reinforcement steel)	10.3	<b>41'200</b>
Bitumen to MSWI	0.2	<b>800</b>
Non-reinforced concrete dismantling	2.5	<b>10'000</b>
Steel to recycling #	0.7	<b>2800</b>

# cut-off to recycling. Not inventoried in ecoinvent database.

<sup>44</sup> Personal communication with W. Stücheli, based on a licentiate work, March 15, 1996.

### Land occupation and transformation

A medium-sized MSWI occupies an estimated area of 3000 m<sup>2</sup> (e.g. 54 m by 54 m). Construction, operation and demolition time are estimated to be 4, 40 and 1 year, respectively. Typically, MSWIs are built within or close to industrial areas to obtain small heat distribution distances. Some MSWIs are also close to residential areas. A generic type of transformed land is not devised here, and land transformation is inventoried as 'from unknown land use type'. The occupied area is categorised as built-up industrial area. Recultivation after demolition is not likely, rather reuse as industrial or residential area. However, land transformation after demolition is inventoried as 'to unknown land use type'. Per plant, a land occupation of 120'000 m<sup>2</sup>a as built-up industrial area, and 15'000 m<sup>2</sup>a as 'construction site' is inventoried.

### Uncertainty estimates for land occupation and transformation

Tab. 3.15 GSD<sup>2</sup> values for specific consumption figures of land use exchanges for MSWI

Exchange	GSD <sup>2</sup> value	Pedigree codes	Comment
land transformation MSWI	214%	(4,5,1,1,1,5)	Basic uncertainty of 200%; estimate on land area and production volume
land occupation MSWI	167%	(4,5,1,1,1,5)	Basic uncertainty of 150%; estimate on land area, production volume and building, operation and demolition time

## 3.4 Unit process inventories

The unit process inventories (i.e. non-cumulated data) of all waste materials to MSWI is a very large table of 180 columns and 200 rows. A table representation in print is not very sensible, as many cells contain no data and some comment cells contain long text entries. The inventoried data for waste disposal processes is therefore contained in Excel tables on the CD-ROM. These workbooks contain the list of exchanges in ECOSPOLD format. Tab. 3.16 gives an overview of the file names and their contents.

Tab. 3.16 Names of files with MSWI inventory data and their contents.

Excel file	Contents
13_MSWI_LCI.xls	Waste disposal inventories
13_proc_MSWI_LCI.xls	Process-specific burdens
13_infra_MSWI_LCI.xls	Incinerator infrastructure

## 4 Life Cycle Inventory of Hazardous Waste Incineration

The hazardous waste incineration (HWI) process is in large parts similar to the calculations performed for the MSWI (chapter 3 'Life Cycle Inventory of Municipal Waste Incineration' on page 26). Major differences are:

3. In the HWI additional fuel oil is used for wastes with low heating value.
4. Only average transfer coefficients are available for HWI. No differentiation is made between inert and burnable wastes as for MSWI (see chapter 3.2.1 'Differentiation of burnability of waste materials' on page 26).
5. In the HWI one single solid output is produced, which is solidified and disposed in a residual material landfill.

### 4.1 Waste-specific burdens

#### 4.1.1 Transfer coefficients

Transfer coefficients of HWIs for C, N, S, P, Cl, Br, I, F, Cd, Co, Cu, Hg, Ni, Pb, Zn, and Fe are available from two sources (UVB 1988, Jahn 2002). The following procedure is applied. Priority is given to data from Jahn (2002), as this is more recent data (1997-2001) from a waste solvent incineration plant. Next, data from (UVB 1988) is used, which is older planning stage data. Several chemical elements remain unaccounted for. The remaining data gaps are filled with transfer coefficient data derived for MSWI. Regarding the elevated incineration temperature of HWIs compared to MSWIs, the transfer coefficients for burnable waste are applied (cf. chapter 3.2.1 'Differentiation of burnability of waste materials' on page 26).

The derived transfer coefficients are used as average transfer coefficients. No additional differentiation depending on the burnability of the waste as in the MSWI calculation is made.

**Tab. 4.1 Transfer coefficients for hazardous waste incineration, rounded to three digits**

Element	Source	Air emissions	Water emissions	Residual waste
		g/kg	g/kg	g/kg
H2O	MSWI	1000	0	0
O	MSWI	917	0	83
H	MSWI	1000	0	0
C	UVB 1988	1000	0	0
S	Jahn 2002	0.6	957	42.8
N <sup>1</sup>	UVB 1988	46.3	79.3	0
P	Jahn 2002	0.7	15.3	984
B	MSWI	120	151	729
Cl	Jahn 2002	0.3	998	1.7
Br	Jahn 2002	0.2	998	1.7
F	UVB 1988	0.5	266	734
I	Jahn 2002	0.2	998	2.1
Ag	MSWI	0.013	0.0729	1000
As	MSWI	0.0000102	0.1	1000
Ba	MSWI	1	0	999
Cd	UVB 1988	8.99	102	889
Co	Jahn 2002	0.7	999	0
Cr	MSWI	0.0000739	3.19	997
Cu	Jahn 2002	0.7	42.9	956
Hg	UVB 1988	40.2	106	853
Mn	MSWI	0.00000545	0.01	1000
Mo	MSWI	2	0	998
Ni	Jahn 2002	0.7	205	795
Pb	UVB 1988	3.31	115	881
Sb	MSWI	0.00000389	0.182	1000
Se	MSWI	0.00000503	0.118	1000
Sn	MSWI	1.33	0.0133	999
V	MSWI	0.1	0.01	1000
Zn	Jahn 2002	0.7	13.4	986
Be	MSWI	1	0	999
Sc	MSWI	0.5	0	1000
Sr	MSWI	0.1	0	1000
Ti	MSWI	1	0	999
Tl	MSWI	1	0	999
W	MSWI	0	0	1000
Si	MSWI	2.33	0	998
Fe	Jahn 2002	0.7	803	196
Ca	MSWI	1.67	0	998
Al	MSWI	1.56	0	998
K	MSWI	3.01	0	997
Mg	MSWI	1.38	0	999
Na	MSWI	9.41	0	991

1 The difference to 1000g/kg is emitted as elemental nitrogen (N<sub>2</sub>) to air, which is not inventoried in ecoinvent.

### Uncertainty estimates of transfer coefficients of HWI

The uncertainty of the transfer coefficients can be understood as the system-inherent uncertainty of the incinerator that occurs when a hypothetical waste with perfectly defined composition is incinerated. The same uncertainty formula as for MSWI transfer coefficients is applied (cf. chapter 3.2.4 'Uncertainty estimates for transfer coefficients of MSWI' on page 30).

$$\text{Eq. 4.1} \quad GSD_{TK} = N \cdot \ln(m_{TK}) + 1 \quad \text{with } N = -0.022, \text{ TK in [kg/kg]}$$

### Speciation of emissions

The same speciation of emissions as described in chapter 2.6.1 'Air and water emissions' on page 20 is used.

### 4.1.2 Fuel oil consumption

For the thermal decomposition of complex organic substances a constant temperature of 1200°C is necessary. To achieve the necessary temperature additional light fuel oil is incinerated along with the waste. In the inventory, the fuel oil consumption is allocated waste-specifically, i.e. wastes with high heating value need little or no additional fuel oil, as they are essentially able to sustain the desired incineration temperature themselves. On the other hand, wastes with small or no own heating value increase fuel oil consumption compared to average waste.

Tab. 4.2 Energy input to hazardous waste incinerators

Plant		Valorec Basel 1992	HWI Hamburg 1998	Valorec Schweizerhalle 2000	This study
Source		(Ciba 1994)	(Lahl 2001)	(Jahn 2002)	
Data type		Planning stage	Operation	Operation	
Burnt wastes		waste solvents	solid/liquid	waste solvents	<b>solid/liquid</b>
Heating value of waste	MJ/kg	17	15	21.7	variable
Fuel oil input	kg/kg waste	0.593	0.038	0.031	variable
Heating value of fuel oil <sup>1</sup>	MJ/kg	40.6	42.7	42.7	42.7
Total heat input per kg waste	MJ/kg	41.06	16.62	23	<b>23</b>
Share from fuel oil	%	58.6%	9.8%	9.0%	variable

1 Valorec Basel 1992 uses heavy (high-sulfur) fuel oil, while the other plants use light fuel oil.

From the average operation of the incinerator, it is possible to derive the total incinerator energy input from waste and additional fuel<sup>45</sup>. From literature values, a total incinerator energy input between 17 and 41 MJ per kilogram waste can be expected (See Tab. 4.2). The data from (Ciba 1994) shows a much higher share of fuel oil heat in the total incinerator energy input than the other two plants (60% instead of 10%). This can probably be explained by the fact that this data is older, that incineration efficiency has increased and the fact that (Ciba 1994) gives merely planning stage data. For this study a value of 23 MJ per kilogram waste is chosen, which corresponds to the value from Valorec Schweizerhalle for 2000 (Jahn 2002). If a specific waste has a lower heating value of less than 23 MJ/kg, the difference will be complemented with light fuel oil. This can be formalised with the following equation.

$$\text{Eq. 4.2} \quad m_{oil} [kg] = \begin{cases} 0 & ; \text{if } H_{u,waste} > 23 \text{ MJ/kg} \\ (23 - H_{u,waste}) / H_{u,oil} & ; \text{if } H_{u,waste} < 23 \text{ MJ/kg} \end{cases}$$

The fuel oil will be inventoried as 'fuel oil in industrial furnace' which already contains the direct emissions from fuel oil incineration (Jungbluth 2003b). To avoid double counting the emissions from the incineration of the average fuel oil input (Tab. 4.3) will be subtracted from the inventoried process-specific air emissions (cf. NMVOC, Dioxins, PM, CO, thermal NO<sub>x</sub> below).

<sup>45</sup> Natural gas inputs are not included in this balance, as natural gas is used after the wet scrubber to reheat the flue gas for the DeNO<sub>x</sub> process and is therefore not participating in the incineration.

**Tab. 4.3 Emissions from the incineration of the average light fuel oil input**

Direct emissions from fuel oil incineration		Emissions from 0.031 kg oil / kg waste
NO <sub>x</sub>	g	0.066
NM VOC	g	0.00264
PM total	g	0.000132
Dioxins TEQ	g	5.94E-13
CO	g	0.0079

The GSD uncertainty of the 23 MJ/kg total energy input is estimated to be 153% with the Pedigree approach (see Tab. 4.4).

**Tab. 4.4 GSD<sup>2</sup> values for total energy input for HWI**

Data	GSD <sup>2</sup> value	Pedigree codes	Comment
Energy input of HWI	153%	(1,3,3,2,4,1)	Basic uncertainty of 105%; median of three plants in Switzerland and Germany.

### 4.1.3 Auxiliary material consumption

Several auxiliary materials are used in the incinerator. These are similar to the ones used in a MSWI. Most of them are used in the wet scrubber and wastewater treatment facilities of the plant. Sodium hydroxide is used to neutralise acid forming gases from sulfur, nitrogen and halogens in the wet scrubber. HCl is used to adjust pH. Ammonia is used in the DeNO<sub>x</sub> stage. TMT15 is used to precipitate heavy metals. Iron chloride and polyelectrolyte is used to precipitate metals. Calcium chloride is used to precipitate phosphates. Hydrogen peroxide is used for sulfite oxidation.

**Tab. 4.5 Auxiliary material consumption of the waste solvent incinerator Valorec, Schweizerhalle, Switzerland. Based on operation data 1998 – 2001 (Jahn 2000, Jahn 2002)**

Material	Concentration	Input kg per ton of waste solvent
Water	–	4300
NaOH	30%	128
HCl	32%	7.8
Ammonia NH <sub>3</sub>	25%	5.9
CaCl <sub>2</sub>	77%	4.5
Polyelectrolyte	<sup>1</sup>	0.02
TMT15	15%	0.13
FeCl <sub>3</sub>	40%	0.03
H <sub>2</sub> O <sub>2</sub>	35%	0.082

<sup>1</sup> Dilution in use unknown

The auxiliary materials in Tab. 4.5 should be allocated to the corresponding causing elements in the waste to obtain waste-specific auxiliary material consumption factors. However, the metal composition of the waste solvent is incompletely known (Jahn 2000 gives values for Zn, Ni, Co, Cu and Fe). To circumvent data gaps, which would lead to unjustifiably increased auxiliary material consumption factors, the figures from MSWI are adapted. Since the same thermal NO<sub>x</sub> share as in the MSWI is used, the figures correspond to the correct fuel-NO<sub>x</sub> level and do not need to be corrected (chapter 4.1.4 'Fuel-NO<sub>x</sub> vs. thermal NO<sub>x</sub>' on page 56).

For data adapted from MSWI technology the same uncertainty factors are used, but the pedigree indicator value for 'technological correlation' is set to 4, signifying 'data on related processes or materials but different technology', since the HWI does not use a grate incinerator.

Calcium chloride and hydrogen peroxide are not used in the MSWI process. These materials are heeded additionally. Calcium chloride is used to precipitate phosphates in the scrubber water treatment. In the average waste solvent 0.1% phosphorous is present (Jahn 2000). Of that amount, 99.93% is transferred to the scrubber. The 4.5 kg 77% CaCl<sub>2</sub> per ton waste correspond to 0.003465 kg 100% CaCl<sub>2</sub> per kilogram waste. Thus, per kilogram phosphorus in the scrubber, 3.47 kg 100% CaCl<sub>2</sub> are used

Hydrogen peroxide oxidises sulfite in the scrubber water treatment. In the average waste solvent 0.642% sulfur is present (Jahn 2000). Of that amount, 99.94% is transferred to the scrubber. The 0.082 kg 35% H<sub>2</sub>O<sub>2</sub> per ton waste correspond to 0.0000288 kg 100% H<sub>2</sub>O<sub>2</sub> per kilogram waste. Thus, per kilogram sulfur in the scrubber 4.5 g 100% H<sub>2</sub>O<sub>2</sub> are used<sup>46</sup>.

### Transportation of auxiliary materials

No information on the transportation modes and distances is available. Generic transportation distances to Switzerland are assumed for all materials (see ecoinvent 2000 methodology in Frischknecht et al. 2003a). For aqueous solutions the solvent water is transported as well. The transportation distances used are shown in Tab. 4.6. For lorries an average load factor of 50% is already heeded in the requested data modules. The inventoried transportation service is the product of 'distance × mass'.

Tab. 4.6 Generic transportation modes and distances for auxiliary materials

Material	Concentration	Transportation distance train in km	Transportation distance lorry 28t in km
NaOH	30%	600	50
HCl	32%	200	50
Ammonia NH <sub>3</sub>	25%	600	50
CaCl <sub>2</sub>	77%	600	50
Polyelectrolyte	1	600	50
TMT15	15%	600	50
FeCl <sub>3</sub>	40%	600	50
H <sub>2</sub> O <sub>2</sub>	35%	600	50

1 Dilution in use unknown

### Uncertainty estimates for transportation of auxiliary materials

The needed transportation service is the product of distance and mass. The GSD value of the transportation service, depends on the GSD values for the masses of the auxiliary materials, which in turn depend on waste composition. Thus, no constant GSD value for the transportation services can be given. This issue is further discussed in part I of this report.

<sup>46</sup> Not necessarily all sulfur in the waste water is sulfite. The share of sulfite is not known and does not necessarily need to be known. For the waste-specific allocation, the final H<sub>2</sub>O<sub>2</sub> consumption is the same, if consumption factors were specific to *all* sulfur in wastewater or to the sulfite alone. No waste-specific sulfite share is known, so an average sulfite share would have to be assumed anyway, if a detailed calculation were to be performed. This average sulfite share drops out of the calculation for the final H<sub>2</sub>O<sub>2</sub> consumption, and a reference to all sulfur in wastewater can be made.

#### 4.1.4 Fuel-NO<sub>x</sub> vs. thermal NO<sub>x</sub>

The emission level of NO<sub>x</sub> in the waste solvent incinerator Schweizerhalle is 0.52 g/kg waste, which is similar to a MSWI with DeNO<sub>x</sub> (0.4 g/kg, see Tab. 3.7). The level of thermal NO<sub>x</sub> is influenced by incineration temperature and homogeneity of fuel. Both, temperature and homogeneity are higher in HWI than MSWI. The level of fuel-NO<sub>x</sub> is influenced by nitrogen content in the waste. The nitrogen content in the waste solvents incinerated at Valorec, Schweizerhalle is 1% per wet mass. In average MSW the nitrogen content is equal (1% in Zimmermann et al. 1996) or somewhat lower (0.3% in current study). No further information on thermal NO<sub>x</sub> share was available from the plant operators (Jahn 2002). Hence the same share as in MSWI (50% thermal NO<sub>x</sub>) is used. The GSD for this figure is assumed to be 141% (like MSWI).

#### 4.1.5 Solid residues

The raw composition of the solid residues is calculated from the waste composition and the transfer coefficient for residues. The data are augmented with the necessary oxygen from oxidation of elements (see chapter 2.6.2 'Oxidation of solid outputs' on page 21). The residue is solidified with cement and disposed in a residual landfill (see part III on landfills).

The GSD values for the several solid outputs are obtained from the GSD of the waste composition data and transfer coefficients by using the formula for multiplication of values. Uncertainty estimates for waste composition data are discussed in chapter 2.5 'Uncertainty of waste composition data' on page 17. Uncertainty estimates for transfer coefficients are discussed on page 52.

### 4.2 Process-specific burdens

The discerning of process-specific emissions is identical to the MSWI. Thermal NO<sub>x</sub>, NMVOCs, dioxins, particulate matter PM, carbon monoxide are inventoried as process-specific emissions. For further explanations see chapter 3.3 'Process-specific burdens' on page 40.

#### 4.2.1 Thermal NO<sub>x</sub>

As discussed in chapter 4.1.4 'Fuel-NO<sub>x</sub> vs. thermal NO<sub>x</sub>' on page 56, a share of 50% thermal NO<sub>x</sub> is assumed. Out of the average 0.52g NO<sub>x</sub>/kg waste, 0.26g are assigned as process-specific NO<sub>x</sub> emission. Out of that, 0.0794 g can be assigned to the average fuel oil input (Tab. 4.3 on page 54). The difference of 0.18 g is inventoried as process-specific emission for each kg of waste.

#### 4.2.2 Non-methane volatile organic compounds NMVOC

For the waste solvent incinerator Valorec, Schweizerhalle an NMVOC emission factor of 3 mg/kg waste is reported (Jahn 2002). This is much lower than the NMVOC emission factor for MSWI (64 mg/kg), which can probably be explained by higher incineration temperatures and more homogenous waste. The GSD<sup>2</sup> value is estimated to be 158% with the Pedigree approach.

During average operation an estimated amount of 3.35 g of NMVOC are emitted from the fuel oil input alone (Tab. 4.3 on page 54). It can be concluded from that the NMVOC emissions originate dominantly from the fuel oil incineration. Since the emissions of NMVOC are already inventoried with the fuel oil consumption, the inventoried process-specific amount of NMVOC is zero.

### 4.2.3 Dioxins and furans

No information about dioxins and furans are available. The value of 3 ng TEQ per kilogram waste for MSWI with DeNO<sub>x</sub> is adapted. Out of that, a minor amount of 0.0006 ng is assigned to the average fuel oil input (Tab. 4.3 on page 54). The difference of 2.9994 ng is inventoried as process-specific emission for each kg of waste. The GSD<sup>2</sup> value is estimated to be 158% with the Pedigree approach.

### 4.2.4 Particulate Matter PM

For the waste solvent incinerator Valorec, Schweizerhalle a PM emission factor of 40 mg/kg waste is reported (Jahn 2002). This is larger than the value for MSWI (6 mg/kg), which can probably be explained by high-calorific fuels and consequently finer and highly volatilised ash particles. Out of the 40 mg/kg, a minor amount of 0.132 mg is assigned to the average fuel oil input (Tab. 4.3 on page 54). The difference of 39.87 mg is inventoried as process-specific emission for each kg of waste.

The same particle size distribution as for MSWI is assumed, i.e. 100% is PM<sub>10</sub> and 99.5% is PM<sub>2.5</sub>.

Tab. 4.7 Emissions factors for different size fractions of particulate matter

		Size fraction	Emission factor mg/kg waste
Fine particles	PM <sub>2.5</sub>	99.5%	<b>39.8</b>
Coarse matter	PM <sub>2.5-10</sub>	0.5%	<b>0.2</b>
Large particles	PM <sub>&gt;10</sub>	0	<b>0</b>
Total suspended particles	TSP	100%	40

### 4.2.5 Carbon monoxide CO

For the waste solvent incinerator Valorec, Schweizerhalle, a CO emission factor of 20 mg/kg waste is reported (Jahn 2002). This is much lower than the CO emission factor for MSWI (223 mg/kg), which can be explained by higher incineration temperatures and more homogenous waste. Out of the 20 mg/kg, an amount of 15.88 mg is assigned to the average fuel oil input (Tab. 4.3 on page 54). The difference of 4.12 mg is inventoried as process-specific emission for each kg of waste.

Carbon monoxide emissions from fossil carbon and biogenic carbon are inventoried separately in the ecoinvent database (see also chapter 3.3.5 'Carbon monoxide CO' on page 46). Since hazardous waste can contain biogenic carbon (e.g. preserved wood) or fossil carbon (e.g. organic solvents) The process-specific emission cannot be placed in the data module for (constant) process-specific emissions, but must be inventoried differently for each waste. Emissions of carbon monoxide are hence inventoried with the *waste-specific* air emissions.

Tab. 4.8 GSD<sup>2</sup> values for process-specific emissions of the HWI

Data	GSD <sup>2</sup> value	Pedigree codes	Comment
Polyaromatics in HWI	328%	(2,3,4,2,4,2)	Basic uncertainty of 300%; Adapted from MSWI from confidential data from 5 plants in Switzerland and Germany
Process emissions: PM10	206%	(1,4,1,1,3,1)	Basic uncertainty of 200%; from total PM value from 1 Swiss plant for waste solvent incineration + PM fractions from measurements in MSWI
Process emissions: PM2.5	306%	(1,4,1,1,3,1)	Basic uncertainty of 300%; from total PM value from 1 Swiss plant for waste solvent incineration + PM fractions from measurements in MSWI
CO in HWI	507%	(1,4,1,1,3,1)	Basic uncertainty of 500%; 1 plant for waste solvent incineration in Switzerland

#### 4.2.6 Water consumption

For the waste solvent incinerator Valorec, Schweizerhalle, a water consumption of 4.3 litres/kg waste is reported (Jahn 2002). This is groundwater used as process water for the wet scrubber. The water for the steam production is not included here (approx. 9 litres/kg waste)<sup>47</sup>. The electricity demand to pump the groundwater is assumed to be included in the total electricity demand, and estimated to be 0.2% to 1.2% of the total demand.

### 4.3 Energy balance and waste heat

As explained in chapter 4.1.2 'Fuel oil consumption' on page 53, the HWI needs to maintain a combustion temperature of 1200°C. This is achieved by adding light fuel oil to the waste solvent. After combustion, process steam and electricity are produced with a gross efficiency of 74.4% and 10% respectively (Jahn 2002). Internal consumption of electricity is 0.287 kWh/kg waste (Jahn 2002). The resulting net electrical efficiency is 5.5%. The fuel oil addition makes sure that each specific waste is incinerated with *at least* a combustion energy of 23 MJ. Hence, there is *always* enough energy for the production of the internal electricity consumption even for low calorific wastes. No external energy source – besides the fuel oil input – needs to be inventoried.

<sup>47</sup> This water is attributed to the process steam grid or the district heat grid and must be inventoried there.

**Tab. 4.9 Energy balance of the waste solvent incinerator Valorec, Schweizerhalle, based on the years 1998–2001 (Jahn 2002)**

Energy flows	MJ/kg waste	Efficiency
<b>Input</b>		
Waste solvent	21.7	
Light fuel oil	1.34	
Total input	23.04	
<b>Gross output</b>		
Process steam	17.14	74.4%
Electricity	2.3	10%
Waste heat	3.6	15.6%
<b>Self consumption</b>		
Electricity	1.033	4.5%
in kWh/kg waste	0.287	

The waste heat of the plant is the total energy input minus the energy contained in the produced net electricity. Of that waste heat, the waste heat from the fuel oil input is already heeded in the data module for fuel oil demand. To avoid double counting, the inventoried waste heat in the HWI module is therefore the total energy input (23 MJ/kg waste) minus the produced net electricity (1.27 MJ/kg waste) minus the fuel oil input. The fuel oil input varies according to the heating value of each specific waste. Thus, the waste heat emissions and its uncertainty is dependent on the heating value of the waste. The waste heat can be assumed to be distributed to air and water essentially like in a MSWI. A share of 83% waste heat to air and 17% to water is assumed (see chapter 3.2.10 'Waste heat emissions' on page 38). The uncertainty of these figures assumed to be small (GSD =102%).

The uncertainty of the heating values of the waste is estimated from the waste composition as explained in chapter 2.4.2 'Heating values' on page 15.

Oil/water emulsions with high water content are usually dehydrated before use. This is achieved by ultra-filtration or thermal evaporation. If wastes are not dehydrated at the production site only thermal evaporation is feasible (Lehmann 1990). The energy demand for the separate thermal evaporation is approximately equal to the energy needed to evaporate the water in the incinerator kiln. The latter energy consumption is heeded waste-specifically. No further energy demand for separate evaporation is inventoried.

### 4.3.1 Allocation of energy production

HWIs have the double function of disposing wastes and generating energy. Like for MSWIs, the question arises if and how the burdens from the process shall be split up and allocated to either of these functions (cf. also chapter 2.7 'Allocation of energy production' on page 21). A difference to MSWI is, however, that each kilogram of waste leads to a total energy input of 23 MJ, irrespective of the heating value of the waste.

No information on the monetary revenues of the HWI Schweizerhalle is available, as the generated energy is consumed internally in the chemicals industry and does not enter the open market. Typical market prices could be applied. As the total energy input per kg waste is larger than in a MSWI, larger revenues can be expected from the incineration in a HWI. Nevertheless, full allocation of all burdens to the waste disposal function of the HWI is performed here. This is mainly motivated by the fact, that the waste treatment (i.e. the disposal function of the HWI) requires high combustion temperatures in the first place. In a hypothetical situation, a decrease in heat demand in the chemical industry would *not* lead to a decrease in fuel oil consumption in the HWI, nor to a decrease in produced energy from the HWI. However, a decrease in waste would lead to a decrease in produced energy from the HWI.

The treatment of the hazardous waste requires directly the induced energy flows. It is reasonable to exploit this source to produce useful energy instead of emitting it as waste heat. A full allocation of all burdens to the disposal function of the HWI leads to energy from HWI that is virtually free of any burdens. This agrees with the understanding that waste heat utilisation is a relatively environmentally advantageous energy source. It also seems fair that the *consumer* of the generated energy is rewarded and not the *producer* of the waste, who does usually make no special provisions to make the energy gain from his/her waste optimal.

### 4.3.2 Infrastructure

No information on the infrastructure is available. The infrastructure is expected to be of minor importance in the overall process. As a proxy the infrastructure of the MSWI is adapted, but scaled by a global factor as HWIs have usually a smaller annual throughput than MSWIs. A throughput of 20'000 tons per year is assumed, which is the average size of a Swiss HWI. The infrastructure expenditures are adapted with a factor 0.36<sup>48</sup>.

Operation time is assumed to be 40 years. Accordingly each kg of waste requires  $1.25 \cdot 10^{-9}$  units of a complete plant. Unit Process Inventories

The unit process inventories (i.e. non-cumulated data) of all waste materials to HWI is a very large table of 50 columns and 170 rows. A table representation in print is not very sensible, as many cells contain no data and some comment cells contain long text entries. The inventoried data is therefore contained in Excel tables on the CD-ROM. These workbooks contain the list of exchanges in ECOSPOLD format. Tab. 4.10 gives an overview of the file names and their contents.

**Tab. 4.10 Names of files with HWI inventory data and their contents.**

Excel file	Contents
13_HWI_LCI.xls	Waste disposal inventories
13_proc_HWI_LCI.xls	Process-specific burdens
13_infra_HWI_LCI.xls	Incinerator infrastructure

<sup>48</sup> Assuming that 20% of the infrastructure is not scale dependent, and the remaining 80% are scaled down according to waste masses incinerated from 100kt/a to 20kt/a. This might be an underestimation, as hazardous waste bears more pollutants and demands more treatment than municipal waste.

## 5 Life Cycle Inventory of Open Burning

Open burning is the uncontrolled incineration of wastes outside a designed facility. Open burning was inventoried in (Doka 2000) as a (illegal) disposal option of burnable building wastes. Estimated shares of this disposal path for burnable building wastes were significant around 1994 in Switzerland (BUWAL 1996). The situation has improved drastically since then and illegal disposal of building wastes is close to zero<sup>49</sup>.

As this type of disposal is not common anymore in Switzerland it is not inventoried in ecoinvent 2000. In other countries and special circumstances the inventory for open burning might be relevant. See also chapter 'Municipal waste generation and treatment in other countries' in part I and the section on landfill fires in part III. For such cases inventories must be calculated specifically. Transfer coefficients and process-specific emissions are referenced in (Doka 2000).

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<sup>49</sup> Personal communication with Kaarina Schenk, BUWAL section waste, of 16. April 2002.

## 6 Calculation manual

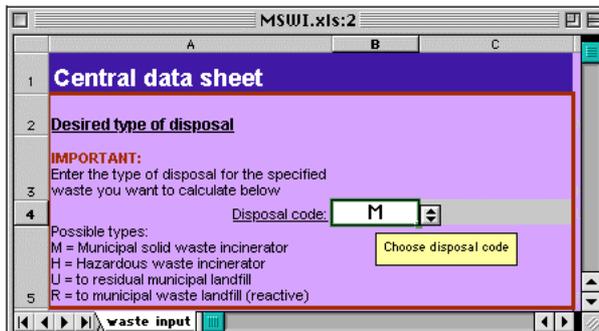
Together with this report calculation tools are distributed which allow the calculation of waste disposal inventories of user-specified wastes. The calculation tools are Excel spreadsheets and run on Microsoft Excel software. In the following a step-by-step guide to calculation of inventories of your wastes.

**Preliminary remark:** It is probably a good idea to keep a backup copy of the original Excel sheets. The sheets are cross-linked and can lose proper functionality by mishap. Be sure not to save the original sheets under different names. If you want to have different versions of the spreadsheets, make copies of the folder containing the spreadsheets.

### 6.1 For waste to incineration (MSWI or HWI)

#### Setting up

1. Open the Excel file '13\_MSWI.xls'. If you want to calculate waste to HWI open *both files* '13\_HWI.xls' and '13\_MSWI.xls'.
2. Do not update links to other sheets. There is no need. Choose 'No' in the dialog box.
3. On slower machines you might want to disable automatic calculation. Choose the menu command 'Extras/Preferences', choose 'calculation' and set the calculation to 'manual'.
4. In '13\_MSWI.xls' go to the sheet 'waste input' (see next picture). If you want to calculate an incineration in a MSWI, set the 'disposal code' in cell B4 to 'M'. If you want to calculate an incineration in a HWI, set the 'disposal code' in cell B4 to 'H'. This will set the residual landfill calculation sheet to use the correct residual material as an input.



5. Go to the waste composition database (located on the same sheet 'waste input') starting at cell K9. Examine the waste compositions listed there. If you find something suitable for you go to 'Choosing the waste fractions'. If not, you need to define a new waste composition.

#### Enter a new waste fraction

6. Find an empty column in the waste composition database in the sheet 'waste input'. **You can only enter data in the white cells.** In case you need to insert new, additional columns, make sure that the functionality of the formulas in columns H and I is retained, i.e. the SUMPRODUCTs must refer the *complete* database area. To do that, insert new columns *starting at the left of the last column* in the database (then copy the contents/formulas of an existing data column, to retain functionality). You can also overwrite existing columns.
7. In a free column, enter the flag for burnability of the waste in row 13 (burnable = 1, or inert = 0). If you have waste that is heterogeneous in burnability, e.g. bulk metal compounded with plastic,

you need to divide the waste up into an inert part (metal) and a burnable part (plastic). Similarly, for waste to sanitary landfill you need to discern waste fractions of different degradability.

8. Enter a name for the waste in row 15 of the free column.
9. Enter heating values, composition data and other characteristics in rows 18 to 66 of the free column. Use the correct units (MJ/kg and kg/kg). *All figures are per kilogram wet waste.* More information and guidance in chapter 2.4 'Necessary waste data' on page 14. An error message appears in row 14, if the entered composition is not reasonably close to 100%. Please note that unnecessary rounding to significant digits at this stage can lead to 'mass gaps', resulting in an error message. The upper and lower heating values according to (Michel 1938) are shown in row 70–73 (extrapolations on given data and theoretic values).
10. Save the sheet.

### Choosing the waste fraction(s)

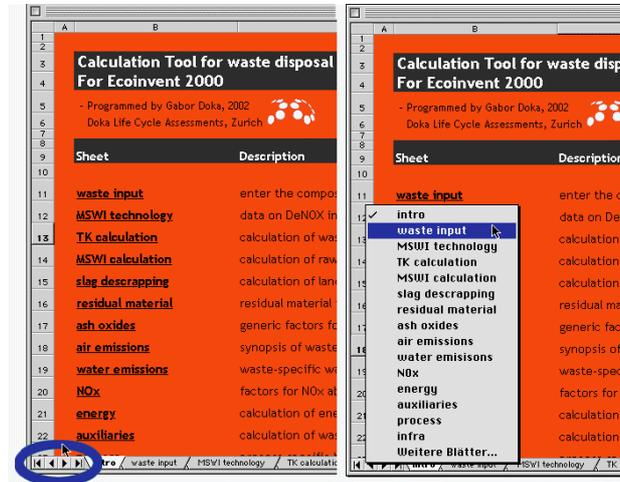
11. In row 12 of the sheet 'waste input' you can enter percentages for the waste fractions you want to be present in your waste. These shares make up the final waste for which you want to calculate LCI unit process data. Usually you just select one column/waste fraction with 100%. You can enter any combination of fractions as long as the sum total of all fractions is 100%<sup>50</sup>. The functional unit is fixed to 1 kg waste to MSWI. Check if the flag for burnability of the waste in row 13 is entered correctly (burnable = 1, or inert = 0).
12. Go to cell E15. Enter an identifying name in English for the waste you just defined. In cell E16 you can enter a German name. These names will be used to generate module names in the sheets 'X-Process' and 'X-Exchange'. The water content will be added to the name automatically. With the restriction that module names can't be longer than 80 characters, and that the MSWI disposal must have the format 'disposal, WASTE\_XY, WW% water, to municipal incineration' (German 'Entsorgung, WASTE\_XY, WW% Wasser, in Kehrlichtverbrennung'), the identifier name should not be longer than 30 to 33 characters (depending on WW%).
13. Force Excel to recalculate the workbook by pressing 'Alt = =' in Windows machines or '⌘ = =' in Mac machines or by choosing the menu command 'Extras/Preferences', 'calculation' and clicking on 'Calculate now'. Important: wait until calculation is complete. Excel's status bar comment in the lower left corner of your screen shows you the progress of the calculation.
14. Go to the sheets 'X-Process' and 'X-Exchange' from where you can copy the inventory data of the disposal of your waste in a MSWI in ECOSPOLD format. Paste the tables *as values* to another sheet (press '⌘-Shift-V' and choose 'values').

## 6.2 General Tips

The workbook comprises many different worksheets where calculations are performed. To inventory a waste disposal, only the sheet 'input' and the sheets 'X-Process', 'X-Exchange', 'X-Person' and 'X-Source' need to be visited by the user. The sheet 'input' contains waste composition data, the latter contain the resulting inventory data in ECOSPOLD format. The other sheets in the workbook can be ignored (but of course must be there for the calculation). To easily navigate between the sheets you can either:

<sup>50</sup> This scheme allows to inventorise burnable and inert waste, which have different transfer coefficients, in one single exchange module.

1. hide the sheets you do not need to see. Go to the corresponding sheet and run the macro 'hidesheet'<sup>51</sup> or
2. press CTRL and click on the sheet navigation arrows (four icons at the bottom left corner of the active sheet) and select the target sheet. See Fig. 6.1.



**Fig. 6.1** Quick access to sheets in large Excel workbooks by CTRL-click on the bottom left corner.

If you do not like the split windows you can remove split windows *on all sheets* by running the macro 'removesplits'.

<sup>51</sup> To make all hidden sheets visible again run the macro 'allVisibleSheet'.

## 7 Cumulative Results and Interpretation

### 7.1 Introduction

Selected LCI results and values for the cumulative energy demand are presented and discussed in this chapter. Please note that only a small part of the about 1000 elementary flows is presented here. The selection of the elementary flows shown in the tables is not based on their environmental relevance. It rather allows to show by examples the contributions of the different life cycle phases, or specific inputs from the technosphere to the selected elementary flows. Please refer to the ecoinvent database for the complete LCIs.

The shown selection is not suitable for a life cycle assessment of the analysed processes and products. Please use the data from the database for your own calculations, also because of possible minor deviations between the presented results and the database due to corrections and changes in background data used as inputs in the dataset of interest.

The ecoinvent database also contains life cycle impact assessment results. Assumptions and interpretations were necessary to match current LCIA methods with the ecoinvent inventory results. They are described in (Frischknecht et al. 2003c). It is strongly advised to read the respective chapters of the implementation report before applying LCIA results.

### 7.2 Results for municipal waste incineration

The Tab. 7.1 shows some arbitrary results of the cumulated inventory of waste disposal in municipal waste incinerator. Three wastes were chosen:

- Average municipal solid waste (MSW), which is a mixture of burnable and inert waste fractions
- Paper, which is a well burnable biomass waste
- Polyethylene (PE), which is a well burnable waste with fossil carbon.

The results only refer to the incineration of those wastes and all further downstream burdens, and not the *production* of the disposed materials. The results of the disposal of these same materials to sanitary landfill are discussed in part III.

Tab. 7.1 Selected LCI results and the cumulative energy demand for municipal waste incineration

Name			disposal, municipal solid waste, 22.9% water, to municipal incineration	disposal, paper, 11.2% water, to municipal incineration	disposal, polyethylene, 0.4% water, to municipal incineration	
Location			CH	CH	CH	
Unit			kg	kg	kg	
Infrastructure			0	0	0	
<b>LCIA results</b>			Unit			
	cumulative energy demand	non-renewable energy resources, fossil	MJ-Eq	0.3870	0.2720	0.2090
	cumulative energy demand	non-renewable energy resources, nuclear	MJ-Eq	0.0719	0.0344	0.0255
	cumulative energy demand	renewable energy resources, water	MJ-Eq	0.0141	0.0074	0.0054
	cumulative energy demand	renewable energy resources, wind, solar, geothermal	MJ-Eq	0.0037	0.0017	0.0013
	cumulative energy demand	renewable energy resources, biomass	MJ-Eq	0.0067	0.0036	0.0027
<b>LCI results</b>			Unit			
resource	Land occupation	total	m2a	0.00136	0.000734	0.000413
air	Carbon dioxide, fossil	total	kg	0.498	0.0176	3
air	NM VOC	total	kg	0.0000847	0.0000777	0.0000735
air	Nitrogen oxides	total	kg	0.000515	0.000498	0.000398
air	Sulphur dioxide	total	kg	0.0000482	0.0000311	0.0000209
air	Particulates, < 2.5 um	total	kg	0.0000224	0.0000174	0.0000134
water	BOD	total	kg	0.00602	0.0033	0.00669
soil	Cadmium	total	kg	4.24E-11	2.43E-11	1.59E-11
<b>Further LCI results</b>			Unit			
water	Cadmium, ion	total	kg	0.000000303	2.41E-08	0.000000362
water	Copper, ion	total	kg	0.00109	0.0000493	0.0000335
water	Lead	total	kg	0.000221	0.00000591	0.00000178
water	Zinc, ion	total	kg	0.000348	0.00000494	0.00000771

The three waste materials have different burnability. The unburnable incineration residues are bottom slag, scrubber sludge and filter dusts, which are landfilled. Prior to landfilling scrubber sludge and filter dusts are solidified with cement. This leads to higher burdens for unburnable wastes, as observable in the row 'land occupation' and in the rows of 'cumulative energy demand' CED.

Unburnable wastes also produce less energy in the incinerator. This increases the demand of energy sources for the treatment of unburnable waste materials. On average, the MSWI is able to generate enough energy for the waste processing (see chapter 3.3.8 'Energy consumption' on page 48). For unburnable materials the process energy is provided from the surplus energy of the average waste incineration. This internal energy consumption does not appear in the CED figures, not because its an internal consumption, but because counting CED during waste incineration would be double counting<sup>52</sup>.

The differences in the row 'carbon dioxide fossil, can be explained mostly by the waste composition. Most fossil CO<sub>2</sub> originates from the waste itself. MSW has a carbon content of 33.4% of which 60.4% is fossil. Paper has no fossil carbon, but polyethylene is a fossil, high-carbon waste (82 w%).

Emissions of NMVOC, NO<sub>x</sub>, SO<sub>x</sub>, and PM are similar. Differences can be explained by differences in composition (for NO<sub>x</sub> and SO<sub>x</sub>) and by increased burdens from treatment of flue gas.

Emissions to water are usually dominated by emissions from landfills (to river water and long-term to groundwater). These emissions reflect differences in *composition* and *landfill fate* of incineration residues. The incineration residues of MSW contain many bulk metals which are transferred to slag and landfilled. In contrast, paper and PE contain heavy metals only in traces. Due to the completely

<sup>52</sup> For example, the raw oil extraction used to produce 1 kilogram of plastic causes a fossil CED, which is assigned in the raw oil extraction step. If the heating value of that plastic would be counted again in a CED indicator during the incineration of that plastic, a double counting would occur.

burnable nature of these wastes, more metals are transferred to the flue gas residues and landfilled in a residual landfill.

### 7.2.1 Contribution analysis for municipal waste incineration

A common question in LCA is, which processes in the life cycle chain constitute the dominant burdens, i.e. which processes have most influence on the end result. This type of analysis is called contribution analysis. For simplicity, this discussion can be performed on the basis of fully aggregated LCIA data. In reports of other process chains in ecoinvent 2000 this contribution analysis is performed on basis of cumulative energy demand CED. In most waste disposal processes the input of commercial energy is of minor importance and not indicative of produced burdens. The major contribution is the burden from the waste material itself. Here a contribution analysis is performed based on Eco-indicator'99-HA-values<sup>53</sup>. Fig. 7.1 shows a contribution analysis for the three wastes already selected for Tab. 7.1. The contributions are grouped.

LT	Long-term emissions to water from landfills
ST water	Short-term emissions to water from MSWI and landfills
ST air	Short-term emissions to air from MSWI
Proc.	Process-specific burdens from MSWI (thermal NO <sub>x</sub> , NMVOC etc.) and landfills
auxiliaries	Auxiliary materials for flue gas scrubber, DeNO <sub>x</sub> , water treatment and solidification cement
transport	Lorry transport for auxiliary materials and of incineration remains to landfills
infra	Infrastructure for MSWI and landfills

**Tab. 7.2 Groups of contributors to total burden**

Dominant parts of the burden are made up of long-term emissions from landfills and short-term emissions to air from MSWI. For the analysed wastes the short-term emissions to air are dominated by CO<sub>2</sub> emissions. Other 'popular' air pollutants like NO<sub>x</sub>, SO<sub>x</sub>, NMVOC or CH<sub>4</sub> play a minor role, which reflects the extensive flue gas treatment performed in the incinerator. The burden from long-term emissions is determined by two factors 1) 'How much of the waste is transferred to the landfilled incineration remains?' and 2) 'What pollutants are contained in these remains?'

A significant part of average municipal waste is inert in combustion and will be transferred to the bottom ash (14 w%). Also burnable wastes contribute to bottom ash but also to fly ash and scrubber sludge. Per kg of average MSW 190 g of bottom ash are produced and 14 g of residual solid waste materials. These figures also contain oxygen added from oxidation processes. By comparison, incineration of 1 kg paper produces 80 + 12 g of solid remains, 1 kg PE produces 19 + 6 g of solid remains. A large part of the long-term burdens of MSW are determined by copper emissions. Average MSW contains 1200 ppm copper and 1090 ppm are released in landfills. For paper and PE also other elements like cadmium and arsenic become important.

Burdens from auxiliary materials reflect the expenditures in flue gas purification steps, like gas scrubbing, DeNO<sub>x</sub><sup>54</sup>, scrubber liquids treatment and cement for the solidification of flue gas treatment residues. Of these, solidification cement is the most important contributor followed by sodium

<sup>53</sup> It is pointed out in part III that CML'01 would be preferable for LCIA of disposal processes, because it has more characterisation values for the important landfill emissions. Eco-indicator'99 values only 8 metals (Cd, Cr VI, Cu, Pb, Hg, As, Ni, Zn), whereas e.g. CML'01 human toxicity scores value 18 elements (Cd, Cr VI, Cu, Pb, Hg, As, Ni, Zn, Ba, Co, Sb, F, Tl, Sn, Be, Se, V). However, the CML'01 method was not implemented in ecoinvent 2000 at the time of completion of this report (September 2003) and could not be used. It is assumed that results from Eco-indicator'99 are indicative enough to perform a contribution analysis.

<sup>54</sup> Only the expenditures for abatement of 'fuel NO<sub>x</sub>' is included here. Expenditures for abatement of 'thermal NO<sub>x</sub>' is included in the process-specific burdens from MSWI (cf. chapter 3.3.1 'Fuel-NO<sub>x</sub> and thermal NO<sub>x</sub>' on page 40).

hydroxide consumption (NaOH). Also important is consumption of quicklime (CaO). The burden from auxiliaries increases usually the more fly ashes and scrubber residues are produced. The exact amounts are determined by the requirements for treatment, which in turn are determined waste-specifically by the composition of the ash and residue.

Transport also reflects the amounts of produced unburnable residues, as it is used for transport of slag and residues to the landfills and the transport of auxiliary materials. Similarly the contribution 'Proc.' combines the process-specific burdens from MSWI and landfills. It is essentially dominated by the process-specific burdens from MSWI. These burdens are constant for every waste. The process-specific burdens from MSWI are dominated by the post-abatement emissions of thermal NO<sub>x</sub> (67%) and by the expenditures for the abatement of thermal NO<sub>x</sub> (22% = input of natural gas and ammonia, DeNOx catalyst materials, and the transportation for the latter two). Other important contributions in this module are emissions of particulates <2.5µm (8%), and dioxins (2%).

All these statements are based on the three examples presented here and valuation with Eco-indicator'99-HA. For other wastes with other compositions or other LCIA methods different inventory parts can become dominant.

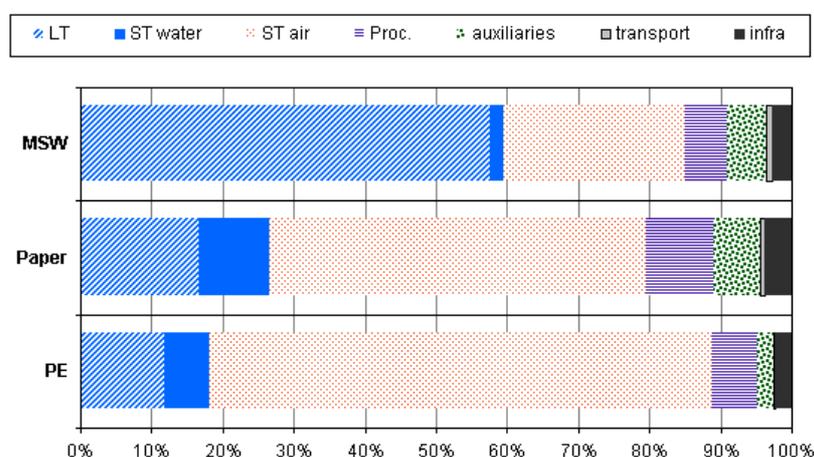


Fig. 7.1 Contribution analysis to environmental burden with Eco-indicator'99-HA for three wastes to MSWI

## 7.2.2 Typology of behaviour of elements in the MSWI

The transfer coefficients can be grouped into several types of elements. These types are not strictly defined here and the boundaries between the types are soft.

**Type 1:** A first type might be elements that are transferred with priority to the **purified flue gas**. These are the non-metals hydrogen, oxygen, nitrogen, carbon (as well as water) which are found in the first and second row of the periodic table of elements. Boron, another second row element, has an ambiguous role and is found in the flue gas to a considerable extent, but mostly in other outputs like slag. These first type elements can be considered volatile and are emitted as direct air emissions of the MSWI.

**Type 2:** A second type are elements that are inclined to be transferred to the **effluent water**. These are mainly the halogens chloride and iodide, and to some extent sulfur and fluoride, another halogen. These second type elements can be considered 'aqua-phil' and are emitted as direct water emissions of the MSWI. Boron also belongs to this second type to some extent, as well as to the first type.

**Type 3:** The vast majority of the remaining elements are mostly metals. They are dominantly transferred to the **solid outputs** of the MSWI – bottom ash, fly ash, ESP ash and scrubber sludge. This

third type of elements can be further distinguished depending on the *reluctance of transferring them to a solid phase*. Bottom ash never enters the raw gas stream and is the easiest to collect as a solid phase. Boiler ash is collected from the raw gas stream as 'fall-out' or fly ash in the steam boiler. The ESP ash is collected by *enforcing* the ash fall-out with an electrostatic field, which indicates the increasing adherence of the material to the air stream. Finally, scrubber sludge is obtained as a solid phase from the air stream only after an extensive series of cascading scrubbing stages.

Type 3 elements that are difficult to 'solidify' are e.g. mercury and cadmium, and to some extent also zinc, antimony, selenium, and silver. They are transferred to a considerable extent to the scrubber sludge. These are metals with low melting points compared to the incineration bed temperature and this subgroup can be referred to as 'volatile metals'.

Type 3 elements of 'medium' difficulty to transfer to a solid phase are e.g. thallium, bromine, and tin that are predominantly transferred to fly and ESP ashes. Again, these elements tend to have rather low melting points.

The last subgroup of type 3 elements are the ones that are predominately transferred to slag (bottom ash). Typical examples are silicon, magnesium, iron or manganese. These are usually elements with high or very high melting points.

In Fig. 7.2 the transfer coefficients in burnable waste are charted and grouped according to the typology presented above.

Belevi & Moench (2000) investigated mechanisms of the transfer of chemical elements to the raw gas in a MSW grate incinerator. Of the elements investigated by Belevi & Moench (2000) the following were found to be transferred to the raw gas dominantly by evaporation (including incineration/oxidation): C, Cl, S, F, Hg, Cd, Zn, Sb, Pb, Br, Sn, As, Cu, and Mo. I.e. these elements are likely to be in a *gas phase* at one point during the incineration process. This group corresponds mainly to the type 1 and 2 mentioned above including the more volatile part of type 3 (the lacking elements were not reported by Belevi & Moench). The following elements were found to be transferred to the raw gas dominantly by entrainment (German 'Verschleppen') i.e. blowing away of *soild* ash flakes: Cr, K, Co, Al, Mn, Ca, P, Ba, Fe, Ni, Mg, Sr, and Ti. This corresponds mainly to the less volatile part of type 3 elements mentioned above. Belevi & Moench (2000) find the elements F, Cu, Mo, Pb, Sn, Zn and Si, Fe, Co, Cr, Mn, Ni to be mostly present in bulky goods in waste, which reduces the likeliness of evaporation or entrainment to the raw gas and increases likeliness of transfer to the bottom ash.

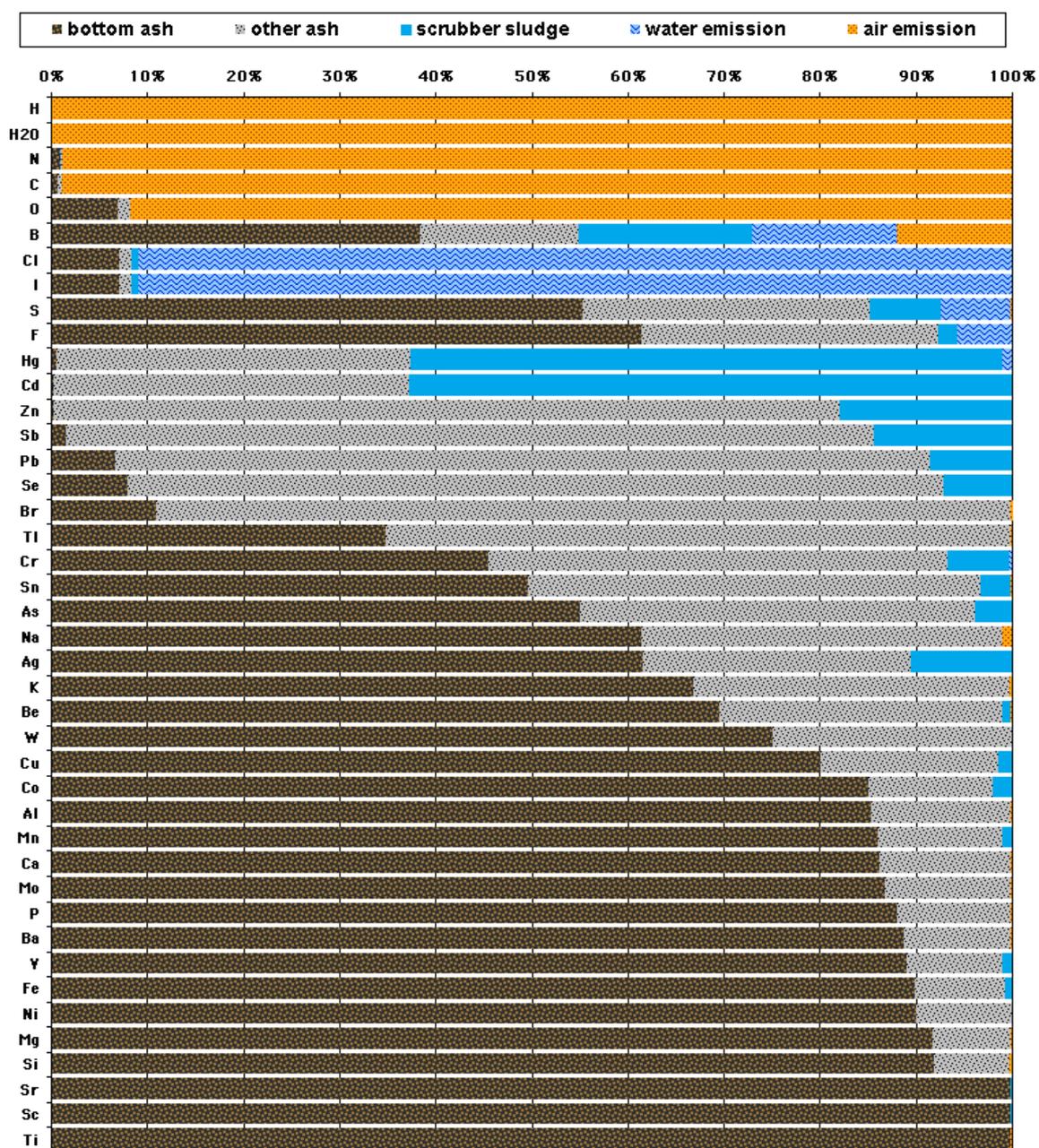


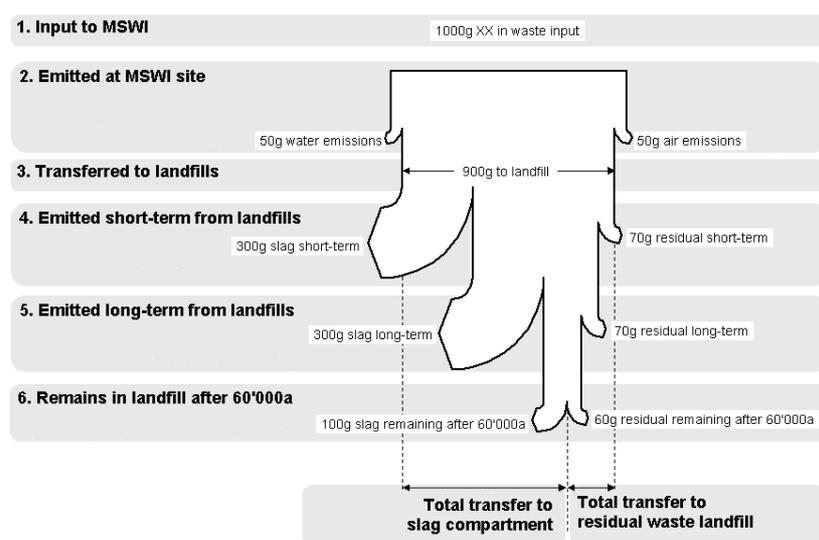
Fig. 7.2 MSWI transfer coefficients of elements in burnable waste. Grouped and arranged according to their behaviour. Easily emitted elements (type 1) at the top, easily solidified elements at the bottom (type 3). 'Other ash' is the sum of boiler ash and ESP ash

### 7.2.3 Flows along the MSWI process chain

The fate of chemical elements within the process chain of disposal can be depicted with the use of material flow diagrams, so called Sankey diagrams. In Sankey diagrams the width of arrows is

proportional to the represented material flux. The following Sankey diagrams<sup>55</sup> are 'flowing' from top-down, representing an approximate process timeline comprising following elements:

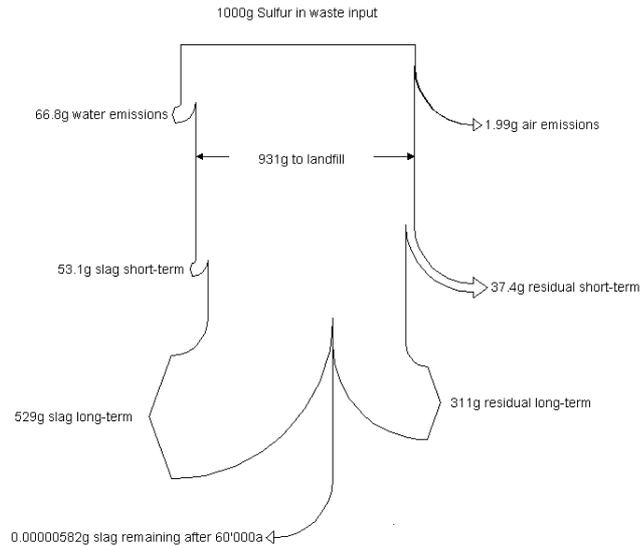
1. Input to MSWI (normalised to 1000 grams)
2. Direct emissions from the MSWI site to air and water
3. Transfer to landfills
4. Direct short-term emissions from the landfills (0–100 years after waste placement)
5. Direct long-term emissions from the landfills (100–60'000 years after waste placement)
6. Remaining material in landfill after 60'000 years



**Fig. 7.3 Legend for Sankey diagrams**

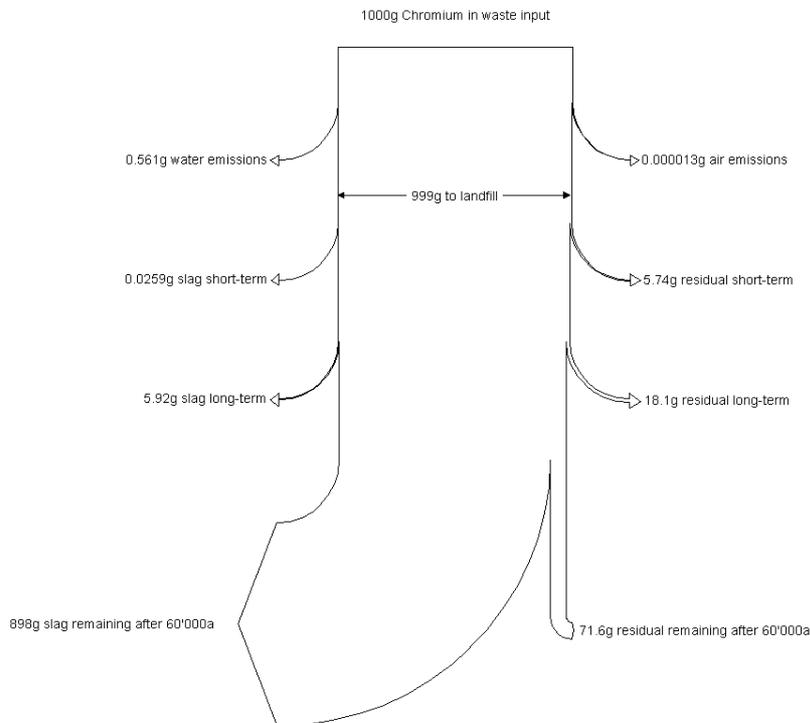
For sulfur (Fig. 7.4), the majority is transferred to landfills. A large part of sulfur (580g/kg) is contained in slag. A considerable part (4000g/kg) needs to be removed from the flue gas stream by scrubbing and is transferred to residual material landfill or emitted to MSWI effluent water. Only 0.2% are emitted to air as sulfur dioxide. In the landfill, most sulfur is weathered and emitted over the years.

<sup>55</sup> The Sankey diagrams were created with the Excel macro 'SankeyHelper v1.1' written by the author. Free download from <http://www.doka.ch/sankey.htm>.



**Fig. 7.4 Sankey diagram for sulfur in the full MSWI process chain**

Chromium (Fig. 7.5) is an element with very high melting point (1875°C for elemental Cr) and also chromium compounds have usually high melting points. During incineration most of chromium (900 g/kg) is transferred to the slag and landfilled in a slag compartment. Some chromium is entrained in the fly ash and transferred to the scrubber residues and ultimately to the residual material landfill (90g/kg). In both landfill types chromium emissions are relatively low. In the residual material landfill with better oxianion mobility chromium mobility is higher. After 60'000a the vast majority of the chromium (970g/kg) will remain in the landfills.



**Fig. 7.5 Sankey diagram for chromium in the full MSWI process chain**

Mercury (Fig. 7.6) has very low melting temperature as an element and mercury compounds also a volatile in the combustion environment of the MSWI. The majority of mercury is transferred to the



The results only refer to the *incineration* of those wastes and all further downstream burdens, and not the production of the disposed materials.

**Tab. 7.3 Selected LCI results and the cumulative energy demand for hazardous waste incineration**

Name			disposal, hazardous waste, 25% water, to hazardous waste incineration	disposal, solvents mixture, 16.5% water, to hazardous waste incineration	disposal, catalyst for EDC production, 0% water, to hazardous waste incineration	
Location			CH	CH	CH	
Unit			kg	kg	kg	
Infrastructure			0	0	0	
<b>LCIA results</b>			Unit			
	cumulative energy demand	non-renewable energy resources, fossil	MJ-Eq	12.3000	3.3100	32.5000
	cumulative energy demand	non-renewable energy resources, nuclear	MJ-Eq	2.0800	0.4960	1.1000
	cumulative energy demand	renewable energy resources, water	MJ-Eq	0.3820	0.0942	0.3090
	cumulative energy demand	renewable energy resources, wind, solar, geothermal	MJ-Eq	0.1090	0.0257	0.0393
	cumulative energy demand	renewable energy resources, biomass	MJ-Eq	0.1720	0.0457	0.1110
<b>LCI results</b>			Unit			
resource	Land occupation	total	m2a	0.0164	0.00451	0.025
air	Carbon dioxide, fossil	total	kg	1.8	1.33	2.35
air	NM VOC	total	kg	0.000437	0.000117	0.00135
air	Nitrogen oxides	total	kg	0.00164	0.000795	0.00325
air	Sulphur dioxide	total	kg	0.00164	0.00039	0.00314
air	Particulates, < 2.5 um	total	kg	0.00026	0.000108	0.00057
water	BOD	total	kg	0.00842	0.00816	0.00546
soil	Cadmium	total	kg	4.57E-10	1.15E-10	5.48E-10
<b>Further LCI results</b>			Unit			
water	Cadmium, ion	total	kg	7.52E-08	2.32E-08	0.0000011
water	Copper, ion	total	kg	0.0000233	0.00000921	0.00438
water	Lead	total	kg	0.0000025	0.00000444	0.00000134
water	Zinc, ion	total	kg	0.000162	0.0000276	0.0000871

Hazardous waste incinerators need a certain operating temperature to achieve the goals of complete destruction of toxic compounds. To achieve this operating temperature the heating value in the waste is utilised. For wastes with low heating value additional fuel oil is incinerated. This is reflected in the energy demands for the three wastes: the lower the heating value of the waste is, the larger the energy demand, which is dominated by fossil energy, e.g. the EDC catalyst which is essentially a silicate matrix and some copper has the highest energy consumption.

The unburnable incineration residues are solidified with cement and landfilled. This leads to higher burdens for unburnable fractions in wastes, as observable in the row 'land occupation'. The spent EDC catalyst is largely transferred to the landfill.

The differences in the row 'carbon dioxide fossil, can be explained mostly by the fuel oil and cement input, but less by waste composition. E.g. the spent EDC catalyst, which has no carbon content has the largest carbon dioxide emissions. Similar patterns apply for the other air emissions.

Emissions to water are usually dominated by emissions from residual landfill. These emissions reflect differences in composition of incineration residues. Spent catalyst contains 5.5% (55'000ppm) copper which is largely landfilled. Only a part of the copper is released from the landfill, but the large copper content leads to significantly larger copper emissions from the disposal of spent EDC catalyst compared to the other two wastes, which contain 267 ppm and 5.1 ppm Cu respectively.

## 8 Conclusions

The waste-specific modelling of waste treatment processes gives distinctly different LCI results for various waste materials. The burdens per waste are greatly influenced by composition. This justifies the waste-specific modelling performed here. In turn, waste-unspecific inventories, e.g. based on mass alone, but ignoring waste characteristics, should not be used.

It could also be shown that burdens from subsequent landfilling of incineration residues are important. Waste incineration assessments shall therefore not only inventory burdens at the incinerator site itself. The life-cycle approach must be followed through and generated burdens after of the incinerator must be heeded also. Again burdens from landfilling need to be assessed waste-specifically, i.e. must heed waste composition.

The approach followed here is limited to modelling the 'technosphere fate' of *chemical elements*. The behaviour of *specific compounds*, like dioxins, is too complex to model in a waste-specific manner, and measurements are generally lacking. If measured data is available, the behaviour of specific compounds of interest can be included in the inventory, however not with the present Excel tools.

### 8.1 Suggestions for further work

#### Fuel oil consumption in hazardous waste incineration

The fuel oil consumption in the hazardous waste incineration is calculated waste-specifically by heeding the lower heating value of the waste. I.e. for the incineration of low-calorific wastes fuel oil is added in the inventory. This seems like a simple, straightforward solution. In fact a problem occurs when looking at the incineration of *waste mixtures*. The 100%-rule for allocation states that combinations of process inventories must result in the same magnitude of exchanges as when these processes were inventoried together as a "black box". For the HWI this means that the inventory for incineration of average waste must result in the same exchanges as for the weighted mixture of the inventories of the incineration of its composing fractions. While this requirement holds true for mass balance in the HWI inventory, it does not for fuel oil input. The combination of processes can consume more fuel oil than reported for the average waste. It is possible to adjust the allocation procedure presented chapter 4.1.2 'Fuel oil consumption' on page 53, to correct this. However, it turns out that not only the average heating value, but the *probability distribution* of heating values of incinerated wastes needs to be known to arrive at accurate figures. This information is currently not available, but has a major influence on the allocated fuel oil input. In future updates, this shortcoming should be corrected.

#### Industrial wastewater treatment

Currently ecoinvent 2000 only has explicit wastewater treatment modules for *municipal* wastewater treatment plants. Wastewater treatment in *industrial* production processes is heeded implicitly in black box production inventories, or – where applicable – with the municipal WWTP model. Industrial wastewater differs from municipal wastewater in composition and treatment technologies. The MSWI model contains quite detailed information on treatment of scrubber liquids. In further studies it could be inspected, if this information is appropriate for certain kinds of industrial wastewater to purification.

## Abbreviations

a	annum = year, used in expressions like "kg/a" for "kilogram per year".
ASR	Automotive shredder residue. Mixture of fine remains from the shredding of used cars (German RESH, Reststoffe aus Shredderwerken).
BE	Canton of Berne in Switzerland
BUWAL	Swiss Agency for the Environment, Forests and Landscape (Bundesamt für Umwelt, Wald und Landschaft)
CHPP	Combined heat/power plant
DeNOx	Nitrogen oxide reduction facility
EMPA	Swiss Federal Laboratories for Materials Testing and Research (Eidgenössische Materialprüfungs- und Forschungsanstalt)
ETH	Swiss Federal Institute of Technology, ETHZ in Zürich (Eidgenössische Technische Hochschule), EPFL in Lausanne (Ecole polytechnique federal )
FGD	flue-gas desulfurisation (in coal power plants) (German REA Rauchgas-Entschwefelungs-Anlage)
GE	Canton of Geneva in Switzerland
GL	Canton of Glarus in Switzerland
GSD	(also SDg) Geometric Standard Deviation. Uncertainty measure for lognormally distributed values. The GSD cannot be smaller than 1 (i.e. 100%). The squared GSD value ( $GSD^2$ ) can be used to calculate 95% confidence intervals around the mean. More information on the uncertainty calculations in ecoinvent 2000 can be found in the methodology report (Frischknecht et al. 2003a).
H <sub>o</sub>	upper heating value (German 'oberer Heizwert')
H <sub>u</sub>	lower heating value (German 'unterer Heizwert')
HW	Hazardous Waste (German: Sonderabfall / Sondermüll)
HWI	Hazardous Waste Incinerator (German: SAVA Sonderabfall-Verbrennungsanlage)
kWh	KiloWatt-hours
l	lower (2.5%) confidence value
LA	Canton of Lausanne in Switzerland
m	mean value
MSW	Municipal Solid Waste (German: Haushaltabfall / Haushaltsmüll, Swiss expression: Kehricht)
MSWI	Municipal Solid Waste Incinerator (Swiss expression: KVA Kehrichtverbrennungsanlage, German expression: MVA Müllverbrennungsanlage)
n.a.	Not applicable (for Pedigree scores) or not available (for waste composition data).
PCB	Polychlorinated biphenyls. A group of chlorine-containing aromatic hydrocarbons.
PM	particulate matter (air emissions), usually given as PM <sub>2.5</sub> and PM <sub>10</sub> , aerodynamic cross-section <2.5 micrometers or <10 micrometers respectively, Total suspended particles TSP is also particulate matter. In ecoinvent 2000 the complementary fractions from 0 – 2.5 µm (fine), from 2.5 – 10 µm (coarse), and >10 µm (large) aerodynamic diameter are discerned in the inventory.

PSI	Paul Scherrer Institute, Villigen
SCR	Selective catalytic reduction (of nitrogen oxides). A DeNOx technology.
SNCR	Selective non-catalytic reduction (of nitrogen oxides). A DeNOx technology.
SO	Canton of Solothurn in Switzerland
TEQ	Toxicity equivalents. Sum parameter for dioxins and furans. In this report the international toxicity equivalents (I-TEQ) are applied.
TI	Canton of Tessin in Switzerland
TJ	Terajoule; 1'000'000 megajoule
TSP	Total suspended particles (air emissions). See also particulate matter PM.
u	upper (97.5%) confidence value
WWT	wastewater treatment

## Glossary of terms

Boiler ash	Fly ash contribution collected in the steam boiler of a MSWI. See in Fig. 2.1 on page 3.
Bottom ash	See 'slag'.
Electrostatic precipitator ash	Also ESP ash. Fly ash contribution collected after the steam boiler of a MSWI with a electrostatic precipitator. See in Fig. 2.1 on page 3.
Fly ash	Combination of boiler ash and electrostatic precipitator ash from a MSWI.
Municipal Solid Waste	Municipal waste is household refuse and similar waste from small businesses or industry, excluding hazardous or special wastes (German: Haushaltabfall / Haushaltsmüll, Swiss expression: Kehricht). Main constituents are kitchen waste (biomass), paper, plastics, metals, glass and fine material.
process-specific	Most of the inventoried burdens of waste disposal are dependent on waste composition in this study ('waste-specific' burdens). Some burdens are constant and the same for every waste. These constant burdens are separated into distinct data modules (infrastructure and other process-specific burdens)
Residual material	Mostly inorganic waste landfilled in a residual material landfill. Residual material from MSWIs comprises a mixture of boiler ash, electrostatic precipitator ash and scrubber sludge from flue gas treatment solidified with cement. See in Fig. 2.1 on page 3.
Scrubber sludge	Solid remains of the wastewater treatment of the fluid from the flue gas scrubber in a MSWI. See in Fig. 2.1 on page 3.
Slag	Solid remains from incineration that remain on the incinerator grate of the MSWI. See in Fig. 2.1 on page 3. Since 'slag' suggests a completely molten material (as from metal ore smelters) some researchers suggest quite correctly to use the more accurate term 'bottom ash'. Only about 40% of the slag mass is actually molten material (Lichtensteiger 1995). In the present study the term 'slag' (German 'Schlacke') is used, based on the terminology in (TVA 2000).
waste-specific	In this study, burdens from the disposal of different waste materials are calculated. Wherever possible burdens are allocated to waste components

(chemical elements). Hence, the waste composition is a chief input parameter. All data items or data modules that change for different waste materials are called 'waste-specific', i.e. 'dependent on the waste composition'. Burdens that are constant for every waste are called 'process-specific'.

## **9 Appendix**

**Tab. A 1 Compositions of waste fractions (in w%) used for the calculation of transfer coefficients of burnable waste  $TK_{burn}$  according to chapter 3.2.1 'Differentiation of burnability of waste materials' on page 26**

Name	food stuff	paper	plastics	cardboard and cartons	minerals	plastic coated paper	metals	glass	diapers and hygienic pads	tetrapack packaging	textiles	wood	other biomass products	hazardious waste	bones
Sources and Comments of Stefanie Helweg	26.0%	21.0%	15.0%	8.0%	8.0%	4.0%	3.5%	3.0%	3.0%	2.0%	2.0%	2.0%	2.0%	1.5%	0.5%
share in average waste	4	14.2	30	16.6	0	17.07	0	0	0	17.1	17.1	14.5	13.3	15	12
H <sub>2</sub> O	60	5.75	6.11	4.71	49.2473	44.15187365	4.71	8.857	30.18	27.11	34.4	18.385	16	41.1	28.8451
O	12.74	46.1227	2	41.92	25	44.46189324	5.987188142	30.18	6.674	5.02	5.03	4.3	34.4	4.3	19.632
H	2	4.9	14.5	6.1	30	6.02922816	43.95779948	48.19	39.21	44.13	34.6	4.3	5.03	4.3	
C	16.24	38.6	79.822	45.31	30	44.26646634	0.24	0.174666753	0.2185	0.28	0.28	0.07	0.07	0.07	
S	0.149526611	0.066	0.16	0.16	0.078174757	0.174666753	0.174666753	0.174666753	0.174666753	0.174666753	0.174666753	0.174666753	0.174666753	0.174666753	0.174666753
N	0.4	0.0943	0.3	0.15	0.175893204	0.175893204	0.175893204	0.175893204	0.175893204	0.175893204	0.175893204	0.175893204	0.175893204	0.175893204	0.175893204
P	0.113	0.0123	0.0189	0.0189	0.0189	0.0189	0.0189	0.0189	0.0189	0.0189	0.0189	0.0189	0.0189	0.0189	0.0189
B	0.001024	0.00189	2.7	0.23	1.1	0.2633	0.02	0.9358	0.9358	0.9358	0.9358	0.9358	0.9358	0.9358	0.9358
Cl	0.4	0.0189	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075
Br	0.0006	0.0006	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014
F	0.02	0.02	0.0045	0.0045	0.0045	0.0045	0.0045	0.0045	0.0045	0.0045	0.0045	0.0045	0.0045	0.0045	0.0045
J															
Ag	0.0002	0.0005	0.00035	0.00035	0.00035	0.00035	0.00035	0.00035	0.00035	0.00035	0.00035	0.00035	0.00035	0.00035	0.00035
As	0.00283	0.00283	0.00487	0.00487	0.00487	0.00487	0.00487	0.00487	0.00487	0.00487	0.00487	0.00487	0.00487	0.00487	0.00487
Ba	0.000943	0.000943	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005
Ca	0.0005	0.0005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005
Co	0.0008	0.0008	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075	0.0075
Cr	0.0018	0.0018	0.000283	0.000283	0.000283	0.000283	0.000283	0.000283	0.000283	0.000283	0.000283	0.000283	0.000283	0.000283	0.000283
Cu	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043	0.00043
Hg	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043
Mn	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043
Mo	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043
Ni	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542	0.000542
Pb	0.001856	0.001856	0.00471	0.009	0.0043	0.00316	0.3	0.001284	0.001284	0.001284	0.001284	0.001284	0.001284	0.001284	0.001284
Sb	4.98755E-05	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513	0.000513
Se	0.000795609	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792	0.000792
Sn	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853	0.000299853
V	0.005824	0.005824	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1
Zn	3.998042958	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066
Si	2.1	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066
Fe	0.06	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066
Ca	2.18	0.377	0.377	0.377	0.377	0.377	0.377	0.377	0.377	0.377	0.377	0.377	0.377	0.377	0.377
Al	0.999510739	1.23	1.23	1.23	1.23	1.23	1.23	1.23	1.23	1.23	1.23	1.23	1.23	1.23	1.23
K	0.35	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943	0.0943
Mg	0.282	0.471	0.471	0.471	0.471	0.471	0.471	0.471	0.471	0.471	0.471	0.471	0.471	0.471	0.471
Nb	0.15	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066	0.066

Tab. A 2 Weight increase from oxidation of elements. See chapter 'Oxidation of solid outputs' on page 21

Element	Considered oxide	Weight increase factor
S	SO <sub>3</sub>	250%
N	NO <sub>3</sub>	443%
P	P <sub>2</sub> O <sub>5</sub>	229%
B	B <sub>2</sub> O <sub>3</sub>	322%
Cl	Cl	100%
Br	Br	100%
F	F	100%
I	I	100%
Ag	Ag <sub>2</sub> O	107%
As	As <sub>2</sub> O <sub>3</sub>	132%
Ba	BaO	112%
Cd	CdO	114%
Co	CoO	127%
Cr	Cr <sub>2</sub> O <sub>3</sub>	146%
Cu	CuO	125%
Hg	HgO	108%
Mn	MnO	129%
Mo	MoO <sub>3</sub>	150%
Ni	NiO	127%
Pb	PbO	108%
Sb	Sb <sub>2</sub> O <sub>3</sub>	120%
Se	SeO	120%
Sn	SnO <sub>2</sub>	127%
V	V <sub>2</sub> O <sub>5</sub>	179%
Zn	ZnO	124%
Be	BeO	278%
Sc	Sc <sub>2</sub> O <sub>3</sub>	153%
Sr	SrO	118%
Ti	TiO <sub>2</sub>	167%
Tl	Tl <sub>2</sub> O	104%
W	WO <sub>3</sub>	126%
Si	SiO <sub>2</sub>	214%
Fe	Fe <sub>2</sub> O <sub>3</sub>	143%
Ca	CaO	140%
Al	Al <sub>2</sub> O <sub>3</sub>	189%
K	K <sub>2</sub> O	120%
Mg	MgO	166%
Na	Na <sub>2</sub> O	135%

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