Update to the model for waste-specific life cycle inventories of bottom ash and residual waste landfills

Commissioner:

Federal Office for the Environment (FOEN)

represented by Pierryves Padey

Author:

Gabor Doka

Doka Life Cycle Assessments, Zurich

Technical report, Zurich, April 2023

Author Gabor Doka, Doka LCA, Zurich, do@doka.ch

Title image Operational phase of a landfill with residual material waste (left) and incinerator bottom ash (right). Clay barrier in the middle. Source: (AWEL 2017). Editorial changes: Cropped, colour saturation

Declaration This report and all previous reports by Doka LCA were created without any use of generative artificial intelligences like ChatGPT or similar. Used were however traditional English-German dictionaries and thesauri.

Suggested citation: Doka G. (2023) Update to the model for waste-specific life cycle inventories of bottom ash and residual waste landfills. Doka Life Cycle Assessments, Zurich, Switzerland. Commissioned by Swiss Federal Office for the Environment (FOEN), Berne, Switzerland. April 2023. Available at http://www.doka.ch/publications.htm

Significant digits Figures in this report often feature several digits. This is not to imply that all the shown digits are really significant or that the data displayed is very precise. Showing several digits helps to minimise the avoidable accumulation of rounding mistakes along the chain of calculations performed here, and in possible future studies referring to this data.

Percent is not a unit A value like 100% is mathematically identical to 1, and "33%" is just a way to write the value 0.33 (which one could also write in yet another different format as "$3.3\cdot10^{-1}$"). Mere *formatting* does not and should not influence the *magnitude of a value*. There is therefore no need to introduce factors or divisors of 100 in formulas for percentages. "Per cent" literally means "per one hundred" and implies the instruction "divide by 100", therefore the mathematical value of the expression "33%" is 33/100 = 0.33 (not 33). In contrast, a formula to calculate a gram value from kilograms must include a factor of 1000, because gram is a *physical unit* (not just a different way to "format" a kilogram value).

Contents

1 Introduction

Life Cycle Assessment has the goal of establishing data for the comprehensive environmental burdens of various processes over their life cycle comprising manufacture, use and disposal. For the last phase it is advantageous to establish the specific burdens associated with a particular waste material, i.e. waste-specifically, instead of a generic, average waste stream.

One goal of this study is to update and regionalise the calculation model for inventories of the disposal of different types of wastes in two particular types of landfills: residual material landfills and the so-called slag compartments. These models were elaborated in (Doka 2003-III) and are used in the suite of Excel waste calculation tools to calculate waste-specific inventories for the disposal of particular materials. Slag compartments receive exclusively bottom ash from waste incineration and therefore assist the inventory calculation of waste incineration by heeding the burdens that result for landfilling a waste-specific bottom ash originating from the incineration of a particular waste material. Residual material landfills receive inorganic, highly polluted waste, for example fly ash and scrubber sludge from waste incineration and can therefore also be part of the waste incineration inventory. But residual material landfills can also receive other waste, which the user can specify.

An update is indicated, since it is always desirable that the models are based on a broader basis of field measurements, and also since other landfill models have been regionalised (Doka 2017-2, Doka 2018, Doka 2020-5) meaning they heed the specific climate and weathering conditions of a user-defined site, while in the initial models of (Doka 2003-III) only an average Swiss climate was used.

The purpose of these models are first and foremost to be able to supplement other activity inventories with the quantified burdens from the disposal of a specific waste composition, which can be defined by the user.

In Switzerland the terms "slag compartment" and "residual material landfill" have been superseded in the new 2016 waste ordinance (VVEA 2016) with the new terms "landfill type D" and "landfill type C". Since these terms are not self-explanatory and at worst prone to misunderstandings, especially in an international context[1], the original names are maintained in this report.

In the update performed now also more complexity and accuracy is added to the Excel waste disposal tools by introducing "Full Integration". Full integration means that disposal of secondary and higher order wastes generated from waste treatments are calculated by heeding the various treatment parameters a user has set for a site (while in previous models the treatment of higher order wastes was also considered dynamically and waste-specifically, but using a fixed-factor approach which represented only a particular, fixed treatment model setting). The new full integration approach means that a user of the tools needs not only to provide accurate parameters for the initial foreground treatment, but also for the required treatment processes of higher order wastes.

---

[1] For instance in United States' Resource Conservation and Recovery Act there are "RCRA Subtitle-D Landfills" which encompass Municipal Solid Waste Landfills (which corresponds to VVEA Type E), and Construction and Demolition (C&D) Debris Landfill (VVEA Type B). Neither of these are congruent with bottom ash landfills (VVEA Type D). So using merely a letter "D" to distinguish a landfill type is bound to be an error-prone terminology, and is thus avoided here. The VVEA type of landfills is mentioned in inventories in the synonym field.

2 Goal and scope

The goal of this study is to update the calculation models that allow the creation of waste-specific inventories of waste disposal. The update addresses two particular types of landfills: residual material landfills and the so-called slag compartments. The residual landfill model can either receive waste directly as defined by the user, or as an assistant model for disposing of example fly ash and scrubber sludge from waste incineration and be therefore part of waste-specific inventories of waste incineration. The slag compartment exclusively receives bottom ash from waste incineration and therefore is solely used as an assistant model for waste incineration inventories.

The models elaborated are based on the model methodologies outlined in (Doka 2003-III). They are transfer coefficient models for chemical elements. The transfer coefficients are derived from a working point model, using the typical average pollutant concentrations in leachate, the typical average pollutant content in deposited waste, and the leachate flow, which depends on precipitation and evaporation. In this project extended typical average literature data on leachate and waste concentrations in the two types of landfills are researched. The two landfill models are also regionalised, meaning the user's climate data entered for the landfill site will influence leachate volumes and landfill weathering.

The updated landfill models are integrated into the existing suite of Excl tools crated by Doka LCA, which allow the creation of inventories of various waste disposal processes, like waste incineration, sanitary landfill, wastewater treatment and many others. Depending on the user's choices these process might also be interlinked for a particular site/location. This is outlined in the following.

2.1 System boundaries

2.1.1 Aggregated datasets

The waste treatment activity inventories (or "datasets") can include treatment of downstream higher order waste materials. For instance, incineration of a particular waste fraction can result in solid incineration residues, which need to be landfilled. These higher order waste treatments are modelled also waste-specifically, i.e. heed the elemental composition of the initial waste material treated.[2] The process chain of higher order waste treatments can be rather long, or even include recursions. For instance wastewater treatment can result in sewage sludge, which can be incinerated, resulting in incineration residues, which can be landfilled in a sanitary landfill, leading to leachate, which might be treated (again) in wastewater treatment.

Since all waste disposal inventories are dependent on the treated composition, these higher order waste treatments are exclusively and only applicable to the originally disposed waste material. They serve only as a part of the inventory of the initial waste disposal process. In order to save database space the whole chain of downstream higher order waste treatments are *aggregated* into the inventory of the initial waste treatment resulting in one single inventory dataset. This practice was established in the ecoinvent database 1.0 in 2003. There is little reason to disaggregate the higher order treatments into their separate inventories, since their sole purpose would be to be linked up with the initial upstream

[2] This follows principles of mass flow accounting and mass conservation: For instance if a waste without cadmium is incinerated, then the landfilling of the resulting incineration residues will also have no cadmium input into the landfill and therefore no cadmium emissions from landfilling. In the treatment model, the composition of the residues is a *waste-specific excerpt* relating to a particular initial waste and can therefore be very different from the *average* composition of the real-world residues.

treatment. Therefore a waste treatment dataset can encompass several real-world disposal processes, but are always specific to the initial waste-composition.

Various higher order waste flows can occur. The further destination of these waste flows can be set be the user of the tool.

From municipal incineration: Municipal incineration generates solid residues which are landfilled. Commonly bottom ash goes to a slag compartment and filter ashes and scrubber sludges go to a residual material landfill. The user can also select that all solid residues are landfilled in a sanitary landfill.

From sanitary landfills: Sanitary landfills collect their short-term leachate which is treated in a wastewater treatment plant. Other types of landfills like unsanitary landfills or open dumps are modelled without collection of leachate (Cf. Doka 2017-2:16).

From wastewater treatment plants: Biological stages of wastewater treatment plants build up biomass and in settling tanks a sewage sludge is separated from the wastewater. Sewage sludges are either first digested on site to generate biogas, or directly disposed as secondary waste. Sewage sludges can be disposed in waste incinerators, in sanitary landfills, in unsanitary landfills or in agricultural spreading (landfarming).

Combinations of these choices can lead to quite complex process aggregations and also recursions. Due to modelling limitations it is not possible to combine sanitary landfills and unsanitary landfills.[3]

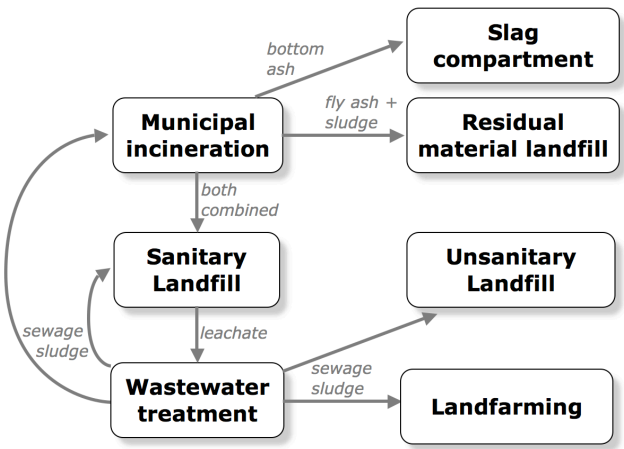

Fig. 2.1 Possible relations between disposal processes from higher order wastes.

Some disposals also generate *no* secondary wastes. Several types of landfills have no secondary wastes (unsanitary landfills, open dumps, slag compartment, residual material landfill, construction waste landfills, excavation material landfills, tailings impoundments). Also open burning and landfarming (surface spreading) create no secondary wastes.

[3] For instance it is not possible to have in one aggregated datasets an initial sanitary landfill disposal, generating a leachate, being treated in wastewater treatment plant and then have the generated sewage sludge going into a unsanitary landfill. It is however possible to have a first order wastewater treatment dataset, having aggregated-in disposal of its sludge in an unsanitary landfill.

2.1.2 Full integration

Full integration means that the calculation of higher-order treatment inventories will be done by linking up the several Excel tools containing those treatment models. This allows for inventory model calculation reflecting the user-set parameters in those tools. In previous instalments of the waste tools, the calculation of higher-order treatment inventories was also done waste-specifically, but was obtained by multiplying specific waste contents with a set of *constant* 'inventory factors' representing a particular treatment model setting, e.g. MSWI with a fixed share of DeNOx facilities (and was not reflecting any parameter changes a user might have made). With the new full integration, the user can change the available parameters in the various treatment models, and those settings will also be reflected in the higher-order waste treatments, not only in the treatment of the primary, initial waste. For full integration calculations all required workbooks containing the treatment various models need to be open.

2.1.3 Example system boundaries

The following chart shows four examples of inventories that can be created with the Excel waste tools, from a simple example (waste directly into residual material landfill) to more complex ones with several higher order waste flows.

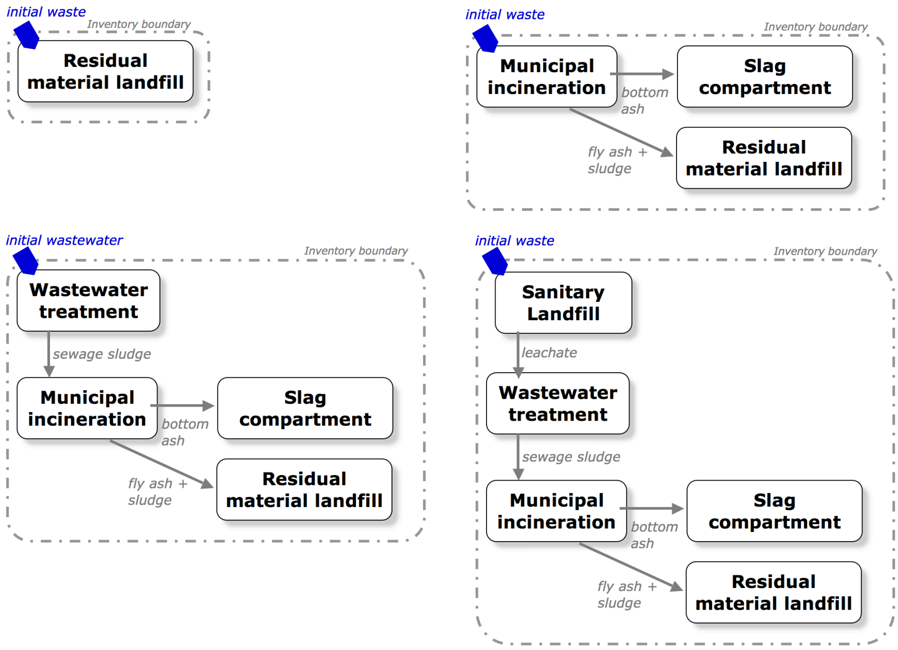

Fig. 2.2 Four examples of disposal inventories of increasing complexity.

2.2 Functional unit

The functional unit of waste disposal datasets is always 1 kg initial waste input (or 1 m^3 for wastewaters).

3 Working point model

3.1 Literature survey

Data on waste composition and landfill leachate composition from slag compartments (MSWI bottom ash) and residual material landfills (APC residues) was compiled for several chemical elements from 114 different literature sources listed in Appendix A. Care has been taken to gather field data from actual leachate in landfills, not from lab leaching experiments or lysimeters. The latter are usually not able to depict the chemical complexities establishing themselves in a landfill, especially over longer time periods. The data gathered—respectively the ratio of concentration in leachate vs. landfilled waste composition—helps to establish the elemental mobility behaviour encountered in these types of landfills. This initial mobility behaviour represents the starting point of the landfill model. The leachate concentrations—and therefore the landfill emissions—are further adapted, anticipating future events of carbonate buffer depletion (Doka 2003-III:26).

The term "initial behaviour" might be prone to misunderstandings. Initial does not mean "first weeks" or "first months". Water percolation in landfills is a *slow* process compared to everyday human activities. Knowing that in a normal landfill it can take *decades* for infiltration water having trickled through a landfill body[4], even leachate of a decades-old landfill can still be considered at its initial stages. Therefore it is appropriate in the literature survey not to dismiss older literature data, neither for leachate nor for waste composition. The goal is to typify the chemical environment in these kinds of landfills which determine the emission of each element. In the modelling employed here, it is surmised that the chemical environment and the solubility-limiting mineral phases, for instance in bottom ash landfills are effectively equivalent.

Compiling a large body of measured data points gives trustworthy information on how such a landfill behaves in the real word *on average*. Although behaviour in a particular individual landfill site could be different from average, the goal here is to generate a transfer coefficient model to establish the waste-specific landfill emissions that can typically be expected in these kinds of landfills.

A large collection of landfill leachate concentration data in the Canton Zurich is available from the DEMIS database. This data was published as PDF reports (AWEL 2006 – 2020) and as online charts (AWEL 2018b). The online charts feature more individual landfill sites, longer time series and more parameters measured. The PDF reports present aggregated data as the measured distribution over the past 10-year periods, while the online charts show values for each available year between 1988 and 2014. The PDF reports contain data of four to five individual slag compartments, while the online charts distinguishes and has data for *eight* individual slag compartments. The PDF reports contain data of four to six individual residual material landfills, while the online charts distinguishes *eight* individual residual material landfills. Not all the data found in the more complete online charts can be matched to the data found in the PDF reports; so although the data stems from the same database DEMIS, there is only a partial overlap in the coverage in the PDF reports and the online charts. In order not to miss any measured data points both sources are included in the compilation to establish typical leachate concentrations.

The PDF reports prior to (AWEL 2014) would present leachate data clustering together data from residual material landfills and inert material landfills (excavation and building waste). It is therefore

4 In a landfill of 20 m height in a temperate climate and 500 mm/yr infiltration and 20% water content of the landfilled waste. the mean residence time of water is 18 years.

not possible to use those earlier sources for data on leachate from residual material landfills. Data from slag compartment leachate is however presented separately.

Any literature data points published as being below a specified threshold were included as 50% of the threshold.[5]

3.2 Average waste and leachate compositions

From the literature survey typical average data on waste composition (waste material in landfill) and leachate concentrations (pollutants in percolating water) are established for either of the two landfill types. This data is shown in Tab. 3.1 below.

In total over 4200 datapoints were compiled for slag compartments and over 2000 datapoints residual material landfills. Each datapoint can represent several years and multiple measurements and therefore the data basis upon which the landfill models are founded is even much broader. The number of datapoints represents a large improvement over the previous model in (Doka 2003-III) where 790 and 200 datapoints were compiled for the slag compartment and the residual material landfill, respectively.

[5] For instance if a source would say the bromide concentration in leachate is "<0.05 mg/l" it would be included as "0.025 mg/l". This procedure does not affect the relevant results of the model, but helps establishing estimates for transfer coefficients for minor elements like silver, tungsten, iodine tin, and thallium.

Tab. 3.1 Mean values for waste composition and leachate concentrations in slag compartments and residual material landfills from literature survey (geometric means). Number of data points (n) are for all recorded points be they mean, upper, lower, minimal or maximal values.

Type	**Slag compartment Waste composition**		**Slag compartment Leachate concentration**		**Residual material landfill Waste composition**		**Residual material landfill Leachate concentration**	
unit	mg/kg	n	mg/L	n	mg/kg	n	mg/L	n
O	372'170	25	0.1	2	299'130	15	#NV	0
H	3'605	13	#NV	0	33'438	1	#NV	0
TOC	8390.9	79	23.905	12	8087.6	12	17.641	5
S	4574.9	109	434.16	38	25'411	54	559.87	16
N	368.65	33	50.113	27	240.02	9	8.038	11
P	3783.8	91	0.071556	7	4836.6	39	0.43734	5
B	179.09	26	2.9245	11	93.49	12	#NV	0
Cl	3565.2	77	4303.9	40	80'718	47	1195.6	17
Br	10.008	33	17.211	1	1544.7	24	20	2
F	485.7	37	1.0911	6	3543.9	16	1.5927	4
I	4.3813	3	#NV	0	0.31853	5	#NV	0
Ag	8.5506	44	#NV	0	27.799	34	#NV	0
As	13.149	108	0.015056	12	44.376	52	0.12294	5
Ba	1067.4	80	0.11951	5	636.8	48	#NV	0
Cd	6.9861	176	0.0029768	34	218.92	83	0.002557	21
Co	32.596	82	0.0039917	9	22.81	48	0.0040354	4
Cr	480.82	156	0.024632	34	277.23	65	0.41668	16
Cu	3240.1	187	0.147	45	1099.3	85	0.04225	17
Hg	0.20709	94	0.0023117	12	7.2788	50	0.0004648	9
Mn	880.5	91	0.039595	18	467.49	57	0.0070608	4
Mo	24.777	58	0.8663	15	16.039	28	1.5051	5
Ni	213.76	149	0.042694	22	95.279	68	0.020358	9
Pb	1416.8	185	0.010314	39	5007.2	85	0.017745	20
Sb	74.172	88	0.026788	6	699.41	50	#NV	0
Se	3.8298	22	0.10784	1	3.0307	25	#NV	0
Sn	120.15	83	0.0021008	2	849.51	55	0.0070711	2
V	38.232	56	0.014534	6	47.074	39	0.10188	1
Zn	3090.8	182	0.03496	34	19036	85	0.14296	15
Be	1.661	17	0.00086545	1	1.5436	22	#NV	0
Sc	4.424	10	#NV	0	2.1874	11	#NV	0
Sr	348.1	25	0.54727	4	281.56	23	#NV	0
Ti	5682.5	45	0.0011282	2	4236.6	37	#NV	0
Tl	0.25399	18	#NV	0	0.27613	11	#NV	0
W	19.945	12	0.096396	3	7.1229	11	1.839	1
Si	159'510	111	3.9877	7	47'394	55	61	2
Fe	53'197	141	0.19795	20	13'714	69	0.068588	3
Ca	99'999	126	349.29	23	172'170	63	21.551	4
Al	48'028	129	0.9508	13	36'816	67	9.601	3
K	7758.8	104	984.1	21	31'692	55	2469.2	4
Mg	11'456	109	18.463	20	10'782	55	1.2532	4
Na	21'429	111	1849.4	22	33'616	57	3978.3	5

3.2.1 Distribution plots

The following distribution plots illustrate the range of literature mean values recorded. Considered are only values given as mean or average values, and values given as extrema (minimum, maximum, upper or lower boundaries) were excluded in these plots.

The data is shown as a novel "distribution box plot", which is a standard box plot additionally showing the cumulative distribution of the data as a curve.

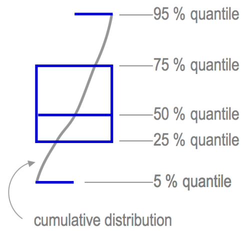

Fig. 3.1 Legend for novel "distribution box plot"

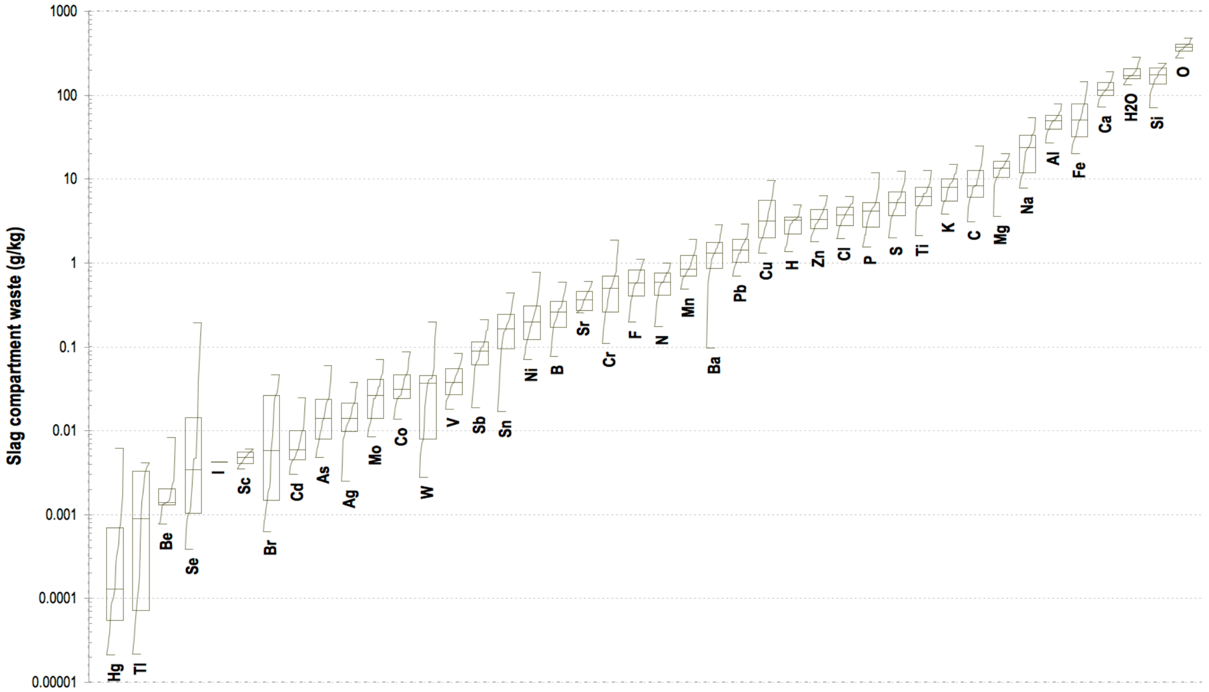

Fig. 3.2 **Distribution box plots of recorded literature mean values for waste composition in slag compartments/bottom ash landfills. In order of ascending median.**

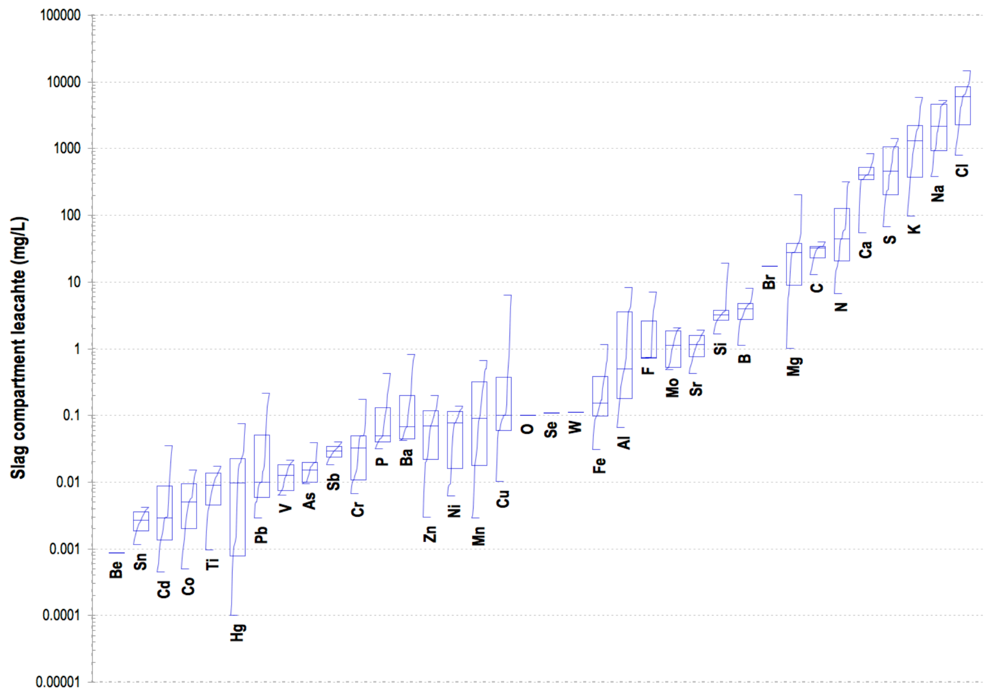

Fig. 3.3 **Distribution box plots of recorded literature mean values for leachate concentrations from slag compartments/bottom ash landfills. In order of ascending median.**

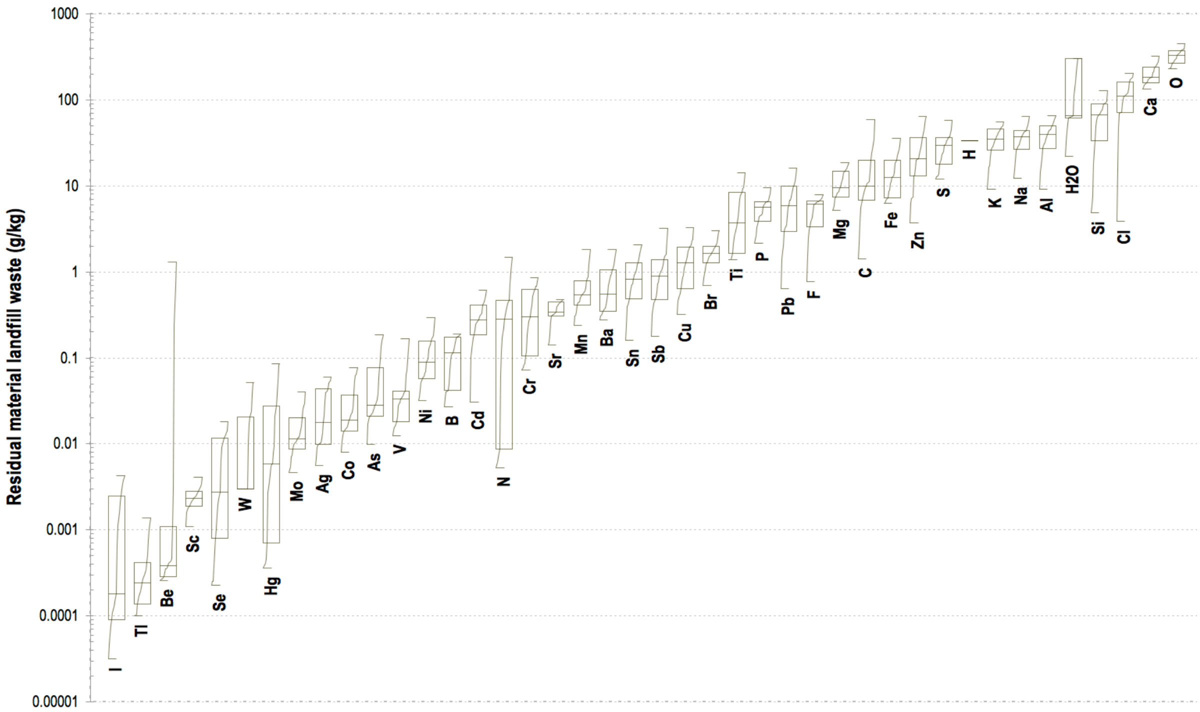

Fig. 3.4 **Distribution box plots of recorded literature mean values for waste composition in residual material landfills. In order of ascending median.**

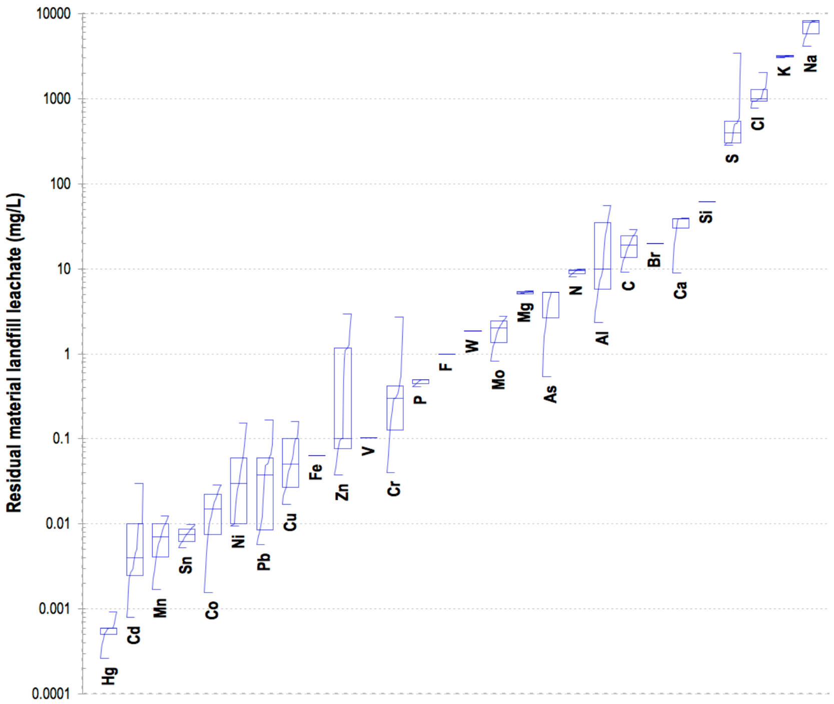

Fig. 3.5 **Distribution box plots of recorded literature mean values for leachate concentrations from residual material landfills. In order of ascending median.**

Some data points have rather narrow distributions, for example iron (Fe) in leachate from residual material landfills. This is usually because only one data point was recorded—and not because the variability of that parameter is very small in reality.

For residual material landfill, the an estimate for the years the data was measured is concentrated in the range 2000 to 2007. This time period can be used to specify the temporal validity of residual material landfill disposal activities.

3.3 Calculation of transfer coefficients

The average waste composition and the average leachate concentration indicate how well soluble an element typically is initially in the chemical environment of that type of landfill. This forms the basis to derive average transfer coefficients (TK) for chemical elements, based on the modelling concepts outlined in (Doka 2003-III). Climate plays a role as well in establishing average transfer coefficients, since the infiltrating rainwater co-determines the leachate volume. With regionalisation of the model the infiltrating water becomes variable depending on the site's given climate data. Therefore no transfer coefficients are listed here, as not to be mistaken for the sole and constant TK in the landfill models. A comparison of short-term transfer coefficients in a Swiss climate is however shown below.

High leachate volumes lead to faster weathering of the landfill's contents and washout. Speed of washout of calcium determines the length of the carbonate phase in the landfill, after which a drop in pH is assumed, which affects some leachate concentrations. With lower pH many cationic metals are washed out more, while oxianions[6] are washed out less. This can lead to counter-intuitive—but entirely accurate—inventory results of more emissions of oxianions with less infiltration water.

3.3.1 Approximations

Some of the modelled chemical elements are missing from the literature compilation. For those elements, the transfer coefficients are adopted from elements with similar behaviour. This is done dynamically from the calculated average transfer coefficients, which change with climate data.

Transfer coefficient approximations for slag compartment:

- Scandium and Thallium are based on the arithmetic mean of other available cations (Ag, Ba, Cd, Co, Cu, Hg, Mn, Ni, Pb, Sn, Zn, Be, Sr, Ti, Fe, Ca, Mg)
- Silver is copied from copper
- Iodine is copied from chlorine
- Hydrogen is copied from calcium

Transfer coefficient approximations for Residual material landfill

- Barium, beryllium, scandium, strontium, titanium, thallium are based on the arithmetic mean of other available cations (Ag, Cd, Co, Cu, Hg, Mn, Ni, Pb, Sn, Zn, Fe, Ca, Mg)
- Antimony and selenium are based on the arithmetic mean of other available oxianions (As, Cr, Mo, V)

---

6 Oxianions considered in the model are boron in HBO_3^{2-}, arsenic in AsO43-, chromium in CrO_4^{2-}, molybdenum in $HMoO_4^{-}$, antimony in SbO_4^{3-}, selenium in SeO_4^{2-}, vanadium in HVO_4^{2-}, and tungsten in WO_4^{2-}.

- Boron is copied from selenium, based on their similarity in the slag compartment data
- Silver is copied from copper
- Iodine is copied from chlorine
- Oxygen and hydrogen is copied from calcium

3.3.2 Comparison residual material vs. bottom ash landfill

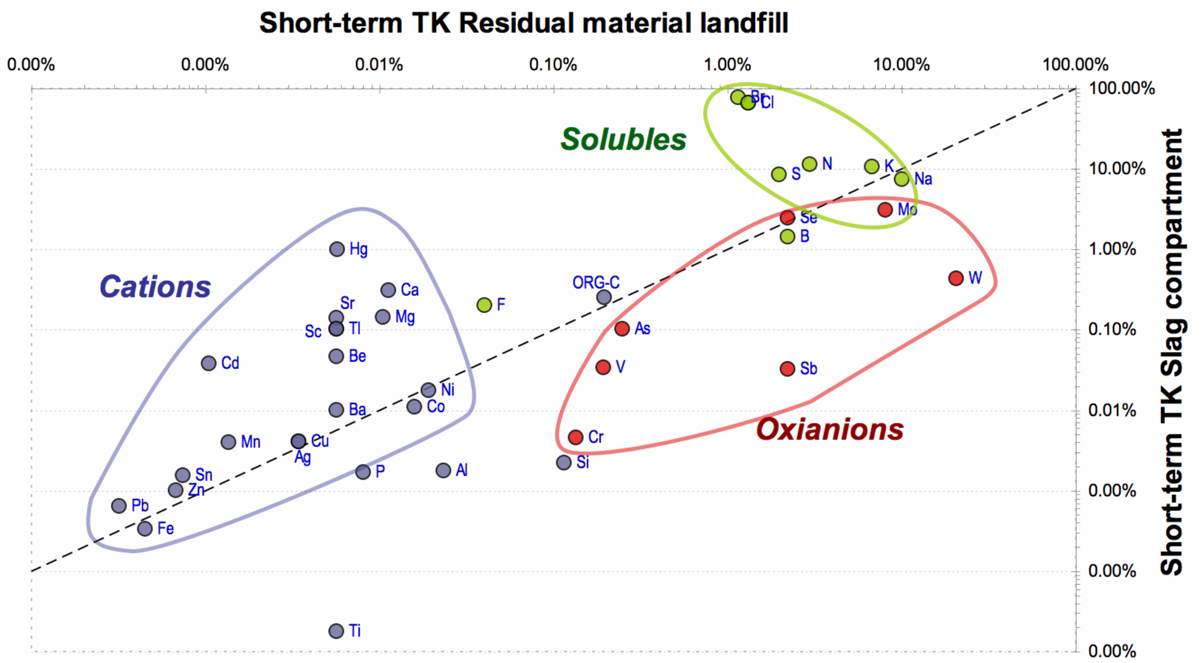

Fig. 3.6 **Comparison of short-term transfer coefficients of residual material landfills vs. slag compartment.**

The short-term transfer coefficients (STTK) in slag compartments vs. residual landfills in a Swiss climate with 500 mm/year infiltration is shown in Fig. 3.6. Both landfills are calculated for a total height of 15 m. This comparison of STTK allows to emphasize the different mobility of pollutants in these two landfill types. Some tendencies can be observed: The elements Cr, Sb, W, V, As, and Mo have significantly larger mobility in residual material landfills. These elements commonly form oxianions – together with oxygen these elements form anionic molecules (cf. footnote 6 on page 16).

A possible explanation of this behaviour is the effect of pH. Either landfill type has distinctly alkaline pH, with a propensity of residual material landfills being somewhat more alkaline. For instance (Hermanns & Moser 2012) list pH values for slag compartments typically ranging from 7.9 – 9, with a median of 8.28 (based on 543 measurements) and for residual material landfills typically ranging from 9 – 10.9, with a median of 9.9 (based on 247 measurements). Oxianions are typically more mobile at high pH values. The larger pH could explain the larger mobility of oxianions in residual material landfills compared to slag compartments.

Another potential explanation of higher oxianion mobility might be the anion exchange capacity AEC. Higher AEC reduces the mobility of anions. The anion exchange capacity is often associated with presence of oxide surfaces, notably iron, manganese, and aluminium oxides, carbonate surfaces, and insoluble organic matter (McLean & Bledsoe 1992:6). The concentrations of these elements are higher in slag compartments than in residual material landfills (e.g. 4 times higher for iron, 2 times for Mn and 1.3 times for Al). The chief reason for this difference is that these elements are predominantly transferred to bottom ash in waste incinerators (transfer coefficients to bottom ash in average municipal waste is 99.1% for Fe, 61% for Mn, and 89% for Al). The relative scarcity of these AEC-forming elements in residual material landfills leads to a low AEC and thus increases the oxianion mobility as observed in the model data. This explanation however seems not really pertinent, since *other anions* like Cl, Br, F, I and to a lesser degree also S as SO_4^{2-} and N as NO_3^- have lower mobility in residual material landfills (green cluster labelled "Solubles" in Fig. 3.6). If lower AEC would be the reason that oxianions are less mobile in residual material landfills, also lower mobility of other anions would be expected – but this is not observed.

In the comparison many cations exhibit lower mobility in residual material landfills (blue cluster in Fig. 3.6). The effect is especially pronounced for Cd, Hg, Sc, Sr, Tl, Ce, Ca, and Mg. A possible explanation here is that this is also an effect of the higher pH typically found in residual material landfills. Cations are commonly less soluble and therefore less mobile at higher pH values.

Generally it is an encouraging fact that these central tendencies can be observed. The tendencies are chemically sensible and the fact that they can be observed means that the base data which has gone into the calculation of transfer coefficients is sufficiently large. If the base data would be inadequate, the comparison would be more chaotic and the tendencies would be drowned out in data noise.

3.4 Waste density

The formulas in (Doka 2003-III) calculate the leachate water annually produced per kilogram of deposited waste. This requires the density of the waste to be known. This is the packed density as it is encountered in the *landfill body after compaction*, not density of surface piles.

Measured density in landfilled bottom ash is 1740 kg/m^3 (Weibel 2020:14). Previously estimated values of 1500 kg/m^3 had been used in the model, which are replaced with the new value.

For solidified residual material, a density of 1700–1800 kg/m^3 is found in literature (Thome-Kozmiensky 2013b:164). A value of 1750 kg/m^3 will be used in the model. Previously estimated values of 1600 kg/m^3 had been used in the model, which are replaced with the new value.

3.5 Organic carbon water emissions

In all ecoinvent versions, the summary parameters for emissions of organic carbon compounds are given in quadruplicate, as TOC, DOC, BOD, and COD. The landfill model calculates emissions of organic carbon (TOC). The emissions of BOD, COD, and DOC are then derived from the TOC figure using conversion factors. The conversion factors are not based many values and are therefore uncertain, but this is of very limited relevance, as hardly any LCIA method heeds emissions of TOC, BOD, COD, or DOC – and if they do their relevance in LCIA results of landfills is low compared to the emissions of toxic heavy metals and semi-metals.

Slag compartment

Based on the literature survey of slag compartments, the typical mean concentration of TOC in leachate is 29.52 mg/L, for COD it's 101.48 mg/L, and for BOD 27 mg/L. From this a COD/TOC ratio of 3.437 is derived and a BOD/TOC ratio of 0.9146. For DOC/TOC a ratio of 1 is used, i.e. all carbon is assumed to be dissolved. These ratios are employed for short- and long-term emissions alike.

Residual material landfill

Based on the literature survey of slag compartments, the typical mean concentration of TOC in leachate is 15.45, for COD it's 79.41 mg/L, and for BOD 7.8 mg/L. The COD/TOC ratio is therefore 5.14, and the BOD/TOC ratio 0.505. For DOC/TOC a ratio of 1 is used. Also here the ratios are employed for short- and long-term emissions alike.

3.6 Infrastructure of slag compartment

The slag compartment is not an individual landfill, but a part of a larger landfill, usually a sanitary landfill. The infrastructure inventory of the slag compartment heeds the smaller size parameters of the slag compartment. The compartment shape is approximated with a rectangular box. The depth is assumed to be 15 m and the area 25'000 m^2. The compartment volume is thus 375'000 m^3. The average density of the waste is 1740 kg/m^3 (cf. chapter 3.4 on page 18). The whole compartment capacity is thus 652'500 tons of waste.[7] Per kilogram of landfilled bottom ash therefore a fraction of $1.53 \cdot 10^{-9}$ of the whole compartment is inventoried.

Humus, loam, tree trunks and earth needs to be removed from the surface. A volume of 1 m^3 per m^2 landfill surface is estimated. The landfill is assumed to be partly submerged below the existing surface level and will rise above it after closure. So, additionally an estimated 50% of the landfill volume needs to be excavated. A total of 212'500 m^3 of excavated material with a density 1500 kg/m^3 is transported 20 km with trucks for recycling (cut-off boundary, i.e. no disposal of this material). Similar material is used for the recultivation after landfill closure.

The base and the flanks of the landfill need to be sealed watertight for the leachate collection system. In reality the flanks are inclined at an angle. For this estimate the flanks are assumed to be perpendicular to the base and have a height identical to the landfill height. The total surface to be sealed is 34'487 m^2. A layer of 50 cm of clay/gravel of 1600 kg/m^3 density is applied. The mineral barrier is covered with a polyethylene PE sheet (0.25 mm thickness, density 960 kg/m^3). The material thickness applied are the legal minimum defined in (VVEA 2016:26). The diesel consumption to apply the PE seals is 0.5 L/m^2 .

At the landfill bottom perforated PE tubes collect the leachate (radius 10 cm, thickness 2.4 cm, density 960 kg/m^3). A grid length of 1000 m is assumed. An identical collection grid is located just below the base seal to detect seal breaks, so the tube material demand is doubled. Further inside drain tubes are added each 5 m of vertical distance, so an additional 1000 m of PE tubes every 5 m. The internal tubes are embedded in a 30 by 30 cm gravel bed (density 1600 kg/m^3).

The compartment must be sealed off against the rest of the landfill. The flank seals included above are assumed to cover that expenditure. No inside walls are inventoried, as the compartment is small enough. After the use phase the top of the landfill is covered with a 50 cm gravel layer and a 4 m recultivation layer of excavation material/soil. The latter is assumed to be recycled and obtained without upstream burden, but has also to be distributed by diggers and loaders.

Since the slag compartment is part of a larger sanitary landfill, certain expenditures are shared with the rest of the landfill. Only a part of those expenditures must be allocated to the slag compartment. These expenditures are the access road, the concrete storage tanks and the sewer pipe (and administrative energy use, cf. below). As an allocation key the relative volumes of MSW landfill and slag

7 Swiss legislation prescribes a minimal size for new slag compartments of 300'000 m^3 (VVEA 2016:Art.37).

compartment are taken[8]. A generic MSW landfill has a total volume of 1'800'000 m^3, of that volume the slag compartment occupies 375'000 m^3 or 21%. So instead of 3000 m road, 8 storage tanks and 3000 m of sewer pipes as in the MSW landfill, only 21% of that (625 m road, 1.67 storage tanks and 625 m sewer pipes) are allocated to the slag landfill. The sewer pipe is assumed to be a concrete tube of 50 cm diameter and 5 cm tube thickness, sewer additional materials are taken from a small scale sewer construction from (Doka 2021-6:43).

An access road of 3000 m length is inventoried. It is provided during all landfill phases of construction, operation and aftercare (see next chapter). The allocated road surface is 3750 m^2. After use, transformation of the road area to pasture is assumed.

3.7 Operation of slag compartment

3.7.1 Energy demand for landfill operations

During the landfill operation, loaders are used to distribute the bottom ash. The diesel consumption is adopted from the figure for residual material landfill waste distribution. An average consumption figure of 0.75 litre diesel per ton of waste (0.027 MJ/kg waste) is inventoried. No active gas collection with pumps is performed in slag compartments and no electricity is needed for that. An administrative building is shared with the whole landfill. The administrative demand is assumed to be mainly connected with waste volume (truckloads). The original energy demand figures of the sanitary landfill of 0.000015 kWh electricity and 0.0016 MJ fuel oil per kilogram municipal waste are decreased, since bottom ash is more dense than municipal waste (1740 kg/m^3 instead of 1000 kg/m^3). Per kilogram of bottom ash 0.0000086 kWh low voltage power grid electricity and 0.00092 MJ heating oil per kilogram bottom ash are inventoried for the administrative energy demand of the slag compartment. All these energies are assigned pro rata to each kilogram of landfilled bottom ash.

3.7.2 Land use exchanges

Land use exchanges are based on the occupied compartment surface of 25'000 m^2. The original land type is assumed to be pasture and meadow (Corinair type 231). For five years the location is a construction site. Landfill operations last approximately 30 years, where the land is inventoried as 'dump site' (Corinair type 132). After operations close a 75 year aftercare period begins. Renaturation is promoted by planting of shrubs. For five years the site is assumed to be of type 'sclerophyllous shrub land' (Corinair type 323). After that a transformation to forest land is assumed. The land occupation as forest land is attributed to forestry products (wood) and not included in the landfill inventory.

The inventoried expenditures for slag compartment depend on the amount of bottom ash generated in waste incineration. Therefore the amounts inventoried per kilogram of incinerated waste are not constant, but are in proportion of the amount of bottom ash generated.

---

8 The reasoning to take volume and not mass is that landfills provide space for waste. Slag is a denser waste than MSW and uses up less space. With the allocation key 'volume' this advantage of slag is respected. However the overall effect of this choice is small as the infrastructure expenditures of landfills are usually not environmentally very relevant.

Tab. 3.2 Infrastructure for one complete Swiss slag compartment (capacity 375'000 m^3 or 652'500 tons)

Process name (EcoSpold1)	unit	amount
excavation, hydraulic digger	m3	162'590
excavation, skid-steer loader	m3	162'590
diesel, burned in building machine	MJ	18'275'000
gravel, round, at mine	kg	51'674'000
bitumen, at refinery	kg	140'630
concrete, exacting, at plant	m3	53.587
polyethylene, HDPE, granulate, at plant	kg	142'210
extrusion of plastics	kg	59'616
reinforcing steel, at plant	kg	3272.8
chromium steel 18/8, at plant	kg	1135.8
cast iron, at plant	kg	552.94
polyvinylchloride, at regional storage	kg	89.666
polypropylene, granulate, at plant	kg	89.666
synthetic rubber, at plant	kg	29.889
sand, at plant	kg	25'107
tap water, at user	kg	697'410
electricity, low voltage, at grid	kWh	5625
transport, lorry 28t	tkm	7'425'800
transport, freight, rail	tkm	59'585
heat, light fuel oil, at boiler 10kW, non-modulating	MJ	603'860
Heat, waste	MJ	20'250
Transformation, from pasture and meadow	m2	28'750
Occupation, construction site	m2a	125'000
Transformation, to dump site	m2	25'000
Occupation, dump site	m2a	750'000
Transformation, from dump site	m2	25'000
Transformation, to shrub land, sclerophyllous	m2	25'000
Occupation, shrub land, sclerophyllous	m2a	125'000
Transformation, from shrub land, sclerophyllous	m2	25'000
Transformation, to forest	m2	25'000
Transformation, to traffic area, road network	m2	3750
Occupation, traffic area, road network	m2a	412'500
Transformation, from traffic area, road network	m2	3750
Transformation, to pasture and meadow	m2	3750

3.8 Infrastructure of residual material landfill

Residual material landfills must have essentially the same constructional details as sanitary landfills or as slag compartments (VVEA 2016:26). The infrastructure of the residual material landfill is therefore inventoried based on the information given for slag compartments in the previous chapter 3.6 'Infrastructure' on page 19. The infrastructure inventory of the residual material landfill heeds the different size parameters of the residual material landfill. The following text mentions only the constructional differences to the slag compartment infrastructure.

The landfill shape is approximated with a rectangular box. The depth is assumed to be 10 m and the area 30'000 m^2. The landfill volume is thus 300'000 m^3. The average density of the waste is 1750 kg/m^3 (cf. chapter 3.4 on page 18). The landfill capacity is thus 525'000 tons of residual material[9]. Per kilogram of landfilled residual material waste therefore a fraction of $1.90476 \cdot 10^{-9}$ of the whole landfill is inventoried.

The residual material landfill is excavated and sealed at the base and the flanks in the same manner as the slag landfill. Due to the larger landfill area compared to the slag compartment two base drain tubes are 1200 m long each. No additional drain tubes inside the landfill body nor compartment walls are inventoried. Four leachate collection tanks are built and connected to the municipal sewer with a 3000 m long small-size sewer.

An access road of 3000 m length is inventoried. It is provided during all landfill phases of construction, operation and aftercare. The road is renovated pro-rata every 50 years. The overall road surface is 18'000 m^2. All road material surface or thickness figures and specific diesel consumption figures are identical to the sanitary landfill or slag compartment.

3.9 Operation of residual material landfill

3.9.1 Energy demand for landfill operations

During the landfill operation, loaders are used to place the solidified residual material. Unlike for MSW in sanitary landfills, compaction of the waste is less an issue here. An average consumption figure of 0.75 litre diesel per ton of waste (0.027 MJ/kg waste) is inventoried. No gas collection with pumps occurs and no electricity is needed for that. A similar administrative building as for the sanitary landfill is inventoried. But its energy demand is lowered since residual waste material is more dense than municipal waste (1750 kg/m3 instead of 1000 kg/m^3). Per kilogram of residual waste material 0.0000086 kWh low voltage power grid electricity and 0.00092 MJ heating oil per kilogram residual waste material are inventoried for the administrative energy demand of the residual waste material landfill.

3.9.2 Land use exchanges

Land use exchanges are based on the occupied landfill surface of 30'000 m^2. The original land type is assumed to be pasture and meadow (Corinair type 231). For five years the location is a construction site. Landfill operations last approximately 30 years, where the land is inventoried as 'dump site' (Corinair type 132). After operations close a 40 year aftercare period begins. Renaturation is promoted by planting of shrubs. For five years the site is assumed to be of type 'sclerophyllous shrub land' (Corinair type 323). After that transformation to forest land is assumed. The land occupation as forest land is attributed to forestry products (wood) and not to the landfill.

The inventoried expenditures for residual material landfill infrastructure depend on the amount of residual material generated in waste incineration. Therefore the amounts inventoried per kilogram of incinerated waste are not constant, but are in proportion of the amount of bottom ash generated. For waste being directly landfilled in a residual material landfill, a constant pro rata part of the infrastructure is assigned.

---

9 Swiss legislation prescribes a minimal size for new residual material landfills of 100'000 m^3 (VVEA 2016:Art.37).

Tab. 3.3 Infrastructure for one complete residual material landfill (capacity 300'000 m^3 or 525'000 tons)

Process name (EcoSpold1)	unit	amount
excavation, hydraulic digger	m3	157'950
excavation, skid-steer loader	m3	157'950
diesel, burned in building machine	MJ	15'023'000
gravel, round, at mine	kg	71'768'000
bitumen, at refinery	kg	675'000
concrete, exacting, at plant	m3	246.42
polyethylene, HDPE, granulate, at plant	kg	130'720
extrusion of plastics	kg	42'955
reinforcing steel, at plant	kg	15710
chromium steel 18/8, at plant	kg	5451.7
cast iron, at plant	kg	2654.1
polyvinylchloride, at regional storage	kg	430.4
polypropylene, granulate, at plant	kg	430.4
synthetic rubber, at plant	kg	143.47
sand, at plant	kg	120510
tap water, at user	kg	3'347'500
electricity, low voltage, at grid	kWh	4500
transport, lorry 28t	tkm	6'890'100
transport, freight, rail	tkm	175'630
heat, light fuel oil, at boiler 10kW, non-modulating	MJ	410'410
Heat, waste	MJ	16'200
Transformation, from pasture and meadow	m2	48'000
Occupation, construction site	m2a	150'000
Transformation, to dump site	m2	30'000
Occupation, dump site	m2a	900'000
Transformation, from dump site	m2	30'000
Transformation, to shrub land, sclerophyllous	m2	30'000
Occupation, shrub land, sclerophyllous	m2a	150'000
Transformation, from shrub land, sclerophyllous	m2	30'000
Transformation, to forest	m2	30'000
Transformation, to traffic area, road network	m2	18'000
Occupation, traffic area, road network	m2a	1'350'000
Transformation, from traffic area, road network	m2	18'000
Transformation, to pasture and meadow	m2	18'000

4 Calculation model

The transfer coefficients derived from the working point model describe the typical, average pollutant mobility of a landfill type in a particular climate.

The second stage of the creation of a waste-specific landfill inventory model is then the application of those region-specific transfer coefficients on a *particular and specific* waste material composition. The pollutants initially present in a waste material correspond to the maximum of the pollutant emissions that are *theoretically possible* for this waste, and the transfer coefficients determine how much of those emissions are *likely to actually occur* in the landfill situation, such as it was designed.[10] The resulting flows are inventoried as the direct emissions from the landfill body. This approach guarantees waste specificity of the inventory, for instance disposal of a waste without any cadmium content will not include any direct cadmium emissions—and for a waste with cadmium, due to mass conservation not more cadmium can be emitted that is initially present in the waste. As in other ecoinvent landfill inventories, the waste-specific inventory is created for a functional unit of one kilogram of waste input into the landfill, referring to a wet weight composition. The material input of waste to a landfill corresponds to its provided disposal service function of taking up unwanted waste.

The landfill disposal inventory is complemented with the common process expenditures like processing energy (waste distribution and compaction, earthworks), materials for any installed beddings, liners, and drainage tubes, and land use exchanges. These expenditures are not waste-specific, but are attributed pro rata to each kilogram of landfilled waste in the same way.

Naturally, the working point model, which describes the average generic behaviour of pollutants at the represented landfill locale, can be applied for several different specific material being deposited in such a landfill, resulting in several waste-specific disposal inventories.

4.1 Solidification with cement

Waste to residual material landfills can sometimes be solidified with cement in order to dampen the acute burden intensity and to adhere to local landfill regulations.

For first order wastes to residual material landfill, the user can choose, if a waste is being solidified with cement or not. This is considered to be a waste–specific characteristic. For instance sludges or highly polluted materials tend to be solidified. With this approach it is also possible to make inventories of *mixtures* of wastes (so called complex wastes) where one material is solidified and the other not, and the inventory will heed the correct amount of cement used.

If the residual material landfill disposes of residues of waste incineration as a higher order waste (fly ashes and scrubber sludge) a solidification is assumed and always included in the model.

For solidified waste, cement and water is added in the weight proportions waste–cement–water of 50%–20%–30%. In the landfill modelling, cement used for solidification is inventoried as an input of cement and water and as an output of solidified cement to landfill as a separate dataset ("disposal, cement, hydrated, 0% water, to residual material landfill"). This allows centralised changes of the

[10] In slag compartments and residual material landfills no leachate treatment is assumed. For sanitary landfills however, part of the outputs can go first to wastewater treatment or to landfill gas capture systems prior to being finally emitted. Also no future conversion, like landfill mining or remediation/conversion, is included, since those are procedures which the currently employed landfill technology does not anticipate or design for.

cement composition, since the added cement is not mixed in the model with the composition of the landfilled waste.

The residual material landfill model has no waste-specific transfer coefficients (unlike the model for municipal mixed waste landfill). Therefore the waste-specific emissions from a waste in a residual material landfill will not be altered in the model by addition of cement. This is a simplification, as at least in the short term the degradation and disintegration of the waste can be slowed down with solidification. Solidification is then a means to postpone emissions from the short term into the long term. As the technical barrier functions of landfills have limited lifetime, the abating effect of solidification will cease to be effective in the long run.

4.2 Included processes

Already in previous models since 2003 a waste treatment could include treatment of generated secondary and higher order wastes. With full integration (cf. chapter 2.1.2 on page 9) the treatment of higher order wastes is calculated dynamically using the workbook models in a dynamic fashion.

In order to detail the involved higher order treatments in aggregated datasets, the inventory will contain an automatically generated text in the metatext field "IncludedProcesses" (ID 402), detailing the waste-specifically aggregated-in processes and quantities, e.g. here for a sanitary landfill activity:

> *... Aggregated are also contributions from higher order waste treatments: purification in wastewater treatment of 6.367kg higher order (secondary) leachate from upstream landfilling including recursive amounts from process chain ; plus controlled combustion in municipal incineration of 0.002793kg higher order (tertiary) sewage sludge from upstream wastewater purification ; plus landfilling in residual material landfill of 0.00002382kg higher order (quaternary) incineration residues from upstream controlled combustion ; plus landfilling in slag compartment of 0.0001257kg higher order (quaternary) bottom ash from upstream controlled combustion.*

In the above example text it can be noted that the landfilling of 1 kg waste generates 6.367 kg leachate as a secondary waste. At first sight, this might seem like a violation of mass conservation, but is explained by the fact the leachate consists mainly of infiltrated rainwater percolating through a landfill and much less of the water present in waste.

4.3 Comment on regional embedding

The models heed the technology and other parameters the user enters. These can be for a particular region, country, and climate and for instance the landfill emissions will be influenced significantly by the climate parameters entered. For the technosphere inputs the regionalisation depends on the database a disposal process is used in.

In EcoSpold2/ecoinvent v3+ technosphere inputs are declared with product names only (E.g. "cement, unspecified") and the database's internal linking mechanisms will link that request to processes producing that good. In this linking, processes with matching geography will be preferred. But also a GLO or RoW process can be linked, if it is the only available one.

In EcoSpold1 technosphere inputs are declared by selecting a particular dataset (e.g. "cement, unspecified, at plant//CH//0//kg"). The geography of that dataset is included, here CH for Switzerland. A proper regional modelling also of the background processes—not only the foreground disposal process—would require to give all linked up processes the same geography as the foreground process. So if a landfilling process in Finland is modelled, then the input of cement and all other auxiliaries should reflect the consumption mix in Finland. Since in EcoSpold1 the identity of a dataset is also defined by its location a dataset like "cement, unspecified, at plant//FI//0//kg" is different from "cement, unspecified, at plant//CH//0//kg". Proper regionalisation also for background processes would require that they are all available for the relevant geography of the foreground process, and this would invoke a major effort to inventory all missing datasets. If the waste tools were to put the same geography as the foreground process for *all* technosphere inputs it would result in invalid ES1 files, until all the missing dataset were created. Of course this problem is not unique to waste disposal processes, but essentially any inventoried activity.[11] Any activity in for instance Finland is only properly regionalised if also the background inputs are reflecting the supply situation in Finland.

> Although the waste models are region specific as far as possible—and as far the user provides region-specific data—the background inputs of material and services are based on the available datasets. The waste tools will not enforce using the geography of the foreground process for all background inputs.

4.4 Emissions categories

As with previous landfill model inventories (Doka 2003-III), the direct emissions from both landfills are divided into emissions short-term after waste placement and a following time period. For the leached pollutants during the first 100 years, no further treatment is assumed and are emitted into river water (surface freshwater). For subsequent pollutants the collapse of sewer pipes is assumed and emission into the groundwater is inventoried.

These choices are appropriate for landfills in *wet climates* where leachate flows *downward* and were based on the situation in Switzerland. In 2017 a regionalisation and adaptation to different climates of the municipal waste landfill model was introduced in (Doka 2017-2:chapter 2.3), which later was also expanded to other landfills like tailings impoundments and inert material landfills (Doka 2018, Doka 2020-5). This extension of the model included also consideration of hyperarid climates, where on average evaporation is larger than precipitation. In such climates, leachate flows *upward* to the surface, where it is evaporated, leaving a brittle, salty crust called evaporite. These emissions are inventoried to industrial soils, and a part which is windblown into air (Doka 2017-2:chapter 2.3).

[11] Geographic approximations are frequent in any LCI database, but are more or less accepted. For instance electricity production and supply is very frequently regionally resolved appropriately, while for internationally traded products a generic GLO dataset is less frowned upon. For waste disposal it would for instance be appropriate to consider the burden for cement in residual material landfill (used for solidification of waste) with the same geography as the foreground landfill process. In EcoSpold1/UVEK this process is however currently only available as a process in Switzerland (CH).

5 Exchange alterations and additions

The new workbooks exchange their LCI information with each other (=full integration), while before higher order treatments were heeded with fixed multiplying factors. During the harmonisation of the full integration some alterations have been introduced and some have been added.

5.1 Landfill land transformations

In previous waste models the land transformations to and from landfill were differentiated into various types of landfills, for instance "Transformation, to dump site, *sanitary landfill*". This was done to allow indirect consideration and valuation of waste mass into landfills done in the Swiss ecoscarcity LCIA method (see Doka 2003-III:41). Since this done now with new dedicated exchanges which are directly accessible for that purpose (see chapter 5.5 on page 28). The distinction of landfill types is not required anymore for these land transformations.[12] All land transformations to and from landfills are therefore assessed with the generic "Transformation, to dump site" and "Transformation, from dump site" respectively.

5.2 Oxygen uptake

Though methodologically not strictly necessary, the oxygen uptake of processes is inventoried in waste incineration, wastewater treatment and sewage sludge digester gas combustion. For incinerations and combustions oxygen uptake is chemically required; in wastewater treatment during organic carbon degradation and nitrification of ammonia oxygen is part of the metabolism of microbes. Oxygen uptake will not have any characterisation factor in LCIA and is inventoried merely as a housekeeping figure for mass balances, i.e. to point out that the outputs from waste disposal can sum up to be larger than the 1 kg waste input. Oxygen uptake is inventoried as natural resource in the subcategory "in air".

5.3 Triazine compound input

In waste incineration the triazine compound trimercapto-s-triazine (TMT15) is used specifically to remove mercury (Hg) and cadmium (Cd) from the incinerator's flue gas scrubber liquid.

In older incineration models this exchange was lacking in the database and had to be approximated with "chemicals, organic". This was now updated to the available "triazine-compounds, at regional storehouse" (ES1) resp. "triazine-compound, unspecified" (ES2). The "chemicals, organic" exchange is still used, but this is for infrastructure of wastewater treatment plant.

5.4 Water resource uptake in sanitary landfill

In the sanitary landfill the short-term leachate from the landfill is collected and sent to wastewater treatment. The water output from the treatment is then inventoried as a return water output exchange.

---

[12] As explained in (Doka 2020:25f) the previous conversion can now be faulty, since the proportionality from m^2 surface area to kilogram waste is not constant anymore since the height of the landfills became a user-adjustable parameter. Also inventories of additional landfill types were introduced since 2003 (e.g. unsanitary landfills, open dumps), for which the former—now obsolete— land transformation types are misleading.

The water initially entering the landfill and creating leachate was added as a water resource uptake to the inventory. This largely balances out the water release from the wastewater model. Without this addition it would mistakenly seem that the process chain creates a water output out of nothing, which can distort results from LCIA method that heed water resources.

For landfills without any sewering of wastewater (unsanitary landfills, open dumps, or sanitary landfills in arid climates with reversed dry sites with evaporation > precipitation) neither water uptake nor release is recorded, as these flows are considered identical to the common natural hydrology of the site.

5.5 Biosphere exchanges for waste mass and organic carbon

The method of ecological scarcity (a.k.a. eco-scarcity, or MOeK, or UBP) is a Swiss LCIA method (BAFU 2021).

Apart from the hundreds of characterisation factors for emissions and resources in this method, there is also a special characterisation factor for **organic carbon placed in landfill**. A similar, parallel characterisation factor is for **total waste mass placed in landfill**. The special quality of these LCIA characterisation factors is that they are intended to be applied to a *technosphere flow*, i.e. to a flow between a waste-producing activity and a landfill, while usually LCIA factors are only applied to *biosphere* exchanges, like emissions or resource uptakes.

Commonly, LCA software and databases cannot apply LCIA factors to technosphere flows and therefore these factors (which existed since 2008) could previously not be included in ecoscarcity LCIA calculations of entire life cycle chains—or attempts to do so introduced serious distortions. In order to remedy that, new, additional biosphere exchanges were introduced in the disposal models of Gabor Doka since 2020. These new exchanges are to be used to include the intended technosphere flows in LCIA to properly calculate results for environmental scarcity LCIA. The new exchanges were placed in the category "resource" subcategory "in ground" (in EcoSpold1) and the names are "Waste mass, total, placed in landfill" and "Organic carbon, placed in landfill". The Ecoinvent association also included these new exchanges in April 2022 for the database version 3.9, but used different (sub)categories.

These new exchanges can also become pertinent in disposals, where landfilling is not the *initial* technology, for instance waste incinerations can have secondary wastes being landfilled, or wastewater disposal can generate sewage sludge that is landfilled. To augment existing disposal datasets from the past with the required inventory amounts a list of the required information has been complied for a large range of datasets from the ecoinvent world (v2.2 and 3.6) and KBOB world (2016) and is available for free at http://www.doka.ch/publications.htm under the heading "New inventory exchanges for characterisation factors of the Swiss method of environmental scarcity".

These exchanges are only intended for use in the environmental scarcity LCIA. More details on purpose, LCIA implementation see chapter 3.10 in (Doka 2020). The significance of these new exchanges in connection with wastewater disposal is presented in chapter 21 of (Doka 2021).

Also in the landfill models for slag compartment and residual material landfill, these two new exchanges were already included in inventory calculations since 2020 and will be provided in future inventories.

5.6 Elementary Exchanges for EN 15804

For the EcoSpold2 inventories in the ecoinvent v3+ database, its managers—the ecoinvent Association—has sought to extend the range of elementary exchanges in 2022. The motivation is, that the European norm listing the core rules for environmental product declarations in construction (EN 15804) has special informational requirements to provide its final LCA results. Apart from the conventional LC(I)A results it also wants to display "other data" with the results, which is data derived from LCA but not assigned to the impact categories of LCIA. For example the cumulated mass of non-hazardous waste from the life cycle process chain. The standards correctly point out that these additional result figures are <u>not</u> part of the <u>impact</u> assessment (LCIA), but merely "additional environmental information" or life cycle data of interest to *accompany* the LCIA results[13] (see also ISO 21930, Chapter 8.2.2).

5.6.1 New elementary exchanges for waste mass

To allow the calculation of such "additional environmental information" (AEI), an extension of the inventoried elementary exchanges is helpful.[14] While the resulting AEI is not part of the LCIA results, the calculation framework is similar to LCIA results. But exchanges like waste mass flows are within the technosphere, not the biosphere: Waste-producing activities produce waste delivered to waste treatment activities, and both those activities are in the technosphere. To calculate the intended cumulated "waste mass sum" it is easiest to introduce an additional exchange that allows tracing waste generation within the life cycle process chain. Within the framework of the LCA calculation this exchange *behaves* like a biosphere exchange, for instance a resource extraction, but the meaning of it is a technosphere exchange. Adding this new exchange allows to calculate cumulated waste masses within the existing calculation framework, just like cumulated resource extractions can be calculated. The ecoinvent Association created two new Elementary Exchanges to count waste masses:

13 The distinction of these categories is important, as "non-hazardous waste" is a much too coarse class of materials to allow a meaningful summation without heeding the different toxicological and pollutant characteristics of the summed up wastes. It is by comparison equally coarse as summing up all air emissions by mass without heeding fate, exposure and effect, or summing up all heavy metal emissions by mass. The EN 15804 text does not make clear what purpose these "additional environmental information" (AEI) serve, and there is a considerable risk that they will nevertheless become involved in result discussions and comparative statements, like "Product A has much larger generated waste mass than product B" although not representing a real impact. Practitioners should do their utmost to avoid this misuse.

14 The results could in theory also be obtained from the cumulated *technosphere* matrix, i.e. the summation of all relevant treatment activities required for a particular dataset. But deriving LCA results from the technosphere amounts is not done usually, and LCA software pre-supposes to derive LCIA impact indicator results from making weighted sums of *elementary* exchange amounts only.

Tab. 5.1 Ecoinvent's elementary exchanges to count waste masses for EN 15804 in EcoSpold2 inventories.

Elementary Exchange name	*Compartment*	*Subcompartment*	*unit*	*comment*	*Elementary Exchange id*
Hazardous waste disposed	inventory indicator	waste	kg	Inventory indicator EN15804. Classification of waste to hazardous based on existing legislation	36a21e09-9ed5-4e04-a76e-140096f89069
Non-hazardous waste disposed	inventory indicator	waste	kg	Inventory indicator EN15804. Classification of waste to non-hazardous based on existing legislation	739ad964-45ab-4623-bad7-05f718433e40

These two new exchanges were added to all the waste tools in the exported EcoSpold2 inventories. On behest of the ecoinvent Association the exchanges will have no properties attached to them (water, carbon , or pollutant content etc.).

5.6.2 Position of exchanges

The new Elementary Exchanges need to be put in inventories to indicate waste masses. There is a choice to do this either in the *waste-generating* activity or the *waste-receiving* treatment activity. For ecoinvent v3+ it was decided to put the elementary exchanges in the waste treatment processes (Symenoidis 2021). This way a consistent accounting of waste masses can be achieved and the authors of the waste-generating activity inventories do not need to familiarize themselves with the categorisation of the waste. While the waste exchanges represent *technosphere* flows, for the LCIA calculation they are in the group of emissions "to environment" (i.e. OutputGroup = 4).

5.6.3 Distinction of waste exchanges regarding hazardousness

For the purposes of EN15804, the waste masses need to be distinguished into hazardous and non-hazardous wastes. Also here a choice is possible of making this distinction either based on the inherent waste material properties and its legal status or based on the waste treatment processes. Shall for instance an CFT light bulb which is put illegally into a municipal waste incinerator be counted as a hazardous waste, since such light bulbs are hazardous waste and would actually need special treatment for a legal disposal? Or shall this counted as a non-hazardous waste, since the input to municipal waste incineration is generally regarded as non-hazardous? The ecoinvent Association decided to use the latter approach (Symeonidis 2022), i.e. waste masses are counted based on the type of treatment process, not on the nature of the waste material[15]. Following classification of waste treatments is applied:

[15] This choice by the ecoinvent Association seems however to be in contrast to the comment given in the corresponding exchanges which state: "Classification of waste to hazardous based on existing legislation".

Tab. 5.2 Ecoinvent's categorisation of waste masses into hazardous and non-hazardous for EN 15804.

Treatment	Comment	Category
Municipal incineration	For MSW	non-hazardous
Slag compartment	For MSWI bottom ash, goes in a landfill similar to a sanitary landfill.	non-hazardous
Residual material landfill	For polluted + mostly inorganic residual wastes, with or without solidification	hazardous
Sanitary landfill	For MSW	non-hazardous
Unsanitary landfill	For MSW	non-hazardous
Open dumping	For MSW	non-hazardous
Open burning	For mixed municipal waste	non-hazardous
Inert material landfill	For excavation or inorganic building waste	non-hazardous
Wastewater treatment	For wastewater and sewage	non-hazardous †
Landfarming	Various sewage and high-organic sludges, based on EU's list of waste "sludges from treatment of urban waste water"	non-hazardous
Underground deposit	In exploited deep salt mines	hazardous
Hazardous waste incineration	Various solid and liquid hazardous wastes	hazardous
Impoundment	For metal ore tailings and coal tailings/slurry	hazardous
Opencast refill	For lignite ash, hazardous, as in the ISIC category	hazardous
Final repository	Nuclear fuel chain/radioactive waste	hazardous
Plasma torch incineration	Nuclear fuel chain/radioactive waste	hazardous
Surface or trench deposit	Nuclear fuel chain/radioactive waste	hazardous

† According to the EU's list of waste, also liquid wastes, leachate (and even waste gasses) are considered wastes, not only solid wastes.

5.6.4 Scope of waste exchanges

A further issue in waste mass accounting stems from the question of whether to include secondary and higher order wastes in the AEI sum or not. If for instance 1 kg of waste polyethylene is incinerated, which generates 34 grams of solid remains which are landfilled, shall this sequence lead to a total of 1 kg waste (because the waste-specific remains come from the already counted waste polyethylene) or shall this be counted as a total of 1.034 kg waste, because the solid remains are a *different* waste than the original polyethylene and require different and additional treatment? The ecoinvent Association decided to use the latter approach (Ioannidou 2022). Therefore higher order wastes are counted in the AEI waste mass sums, not only the waste mass treated initially.

5.6.5 Summary

New elementary exchanges are introduced on behest of the ecoinvent Association in EcoSpold2 inventories that allow the counting of waste mass flows *within the technosphere*. These exchanges only coarsely distinguish hazardous waste mass and non-hazardous waste mass. Categorisation of hazardousness is based on the processing waste treatment activity, not on waste material characteristics. The Elementary Exchanges are placed in waste-receiving treatment activities, not the waste-generating activities. Higher order waste masses are included, e.g. residues from waste incineration are counted again.

These new elementary exchanges for EN 15804 are not to be confounded with the elementary exchanges introduced for the Swiss ecoscarcity LCIA method (see previous chapter 5.5 on page 28). Those latter exchanges are indeed part of the LCIA result, unlike the EN 15804 exchanges, but they only refer to *landfilled* waste, do not distinguish any hazardousness categorisation, but include a separate assessment of landfilled organic carbon.

6 Alternative allocation comment

The EcoSpold1 files exported from the calculation tools are mono-functional in nature and carry 100% of the process burden which is attributed solely to the initial disposal service as the reference product. Any co-product goods are allocated 0% of the burden, which in essence means they are cut-off from the inventory. Co-product goods that can be produced are net energy (heat and electricity), recyclable metals, fertilizer functions on agricultural land and others. For purely informational purposes, if a process does produce any co-products, their waste-specific amounts are mentioned as a text in the GeneralComment field.

Additionally, the GeneralComment mentions also the results of an *alternative* allocation scheme with economical keys, to complement the standard 100%–0% allocation used. This alternative allocation will have no effect on inventory results and is also provided only as additional text information[16]. The comment is added, if applicable, in municipal incineration, sanitary landfills with landfill gas utilization, and wastewater treatments with utilisation of digester gas and/or sewage sludge. A process not producing any co-products, for instance glass disposal, will not have an additional comment.

The employed price data for the alternative allocation with economical keys is shown below. Net energy can be produced from incinerator heat, from landfill gas combustion in sanitary landfills, or from digester gas combustion from sewage sludge from wastewater treatment. Metallic scraps can be extracted from waste incineration residues, but also from dismantling of the wastewater treatment plant. Zinc concentrate is only produced in municipal incineration from washing of fly ashes with the FLUWA process (virtually only in Switzerland). Fertilizer functions are provided, when sewage sludge is spread on agricultural areas (landfarming).

In the case of the zinc concentrate output generated from any FLUWA facilities in the waste incinerator process its price is adapted according to the *waste-specific zinc content* in the zinc hydroxide. The zinc concentrate used for the generic price data of 0.0042 EUR/kg has a generic zinc content of 59%. The zinc hydroxide from MSWI usually has lower zinc concentrations.

The result of the alternative allocation scheme depends on waste characteristics and can not be given generally. For average Swiss municipal solid waste in incineration, an allocation of 91% of the burdens on the disposal function results, a sum of 8% on the two energy products, while all solid recyclates amount to less than 1%. This alternative allocation confirms the prime purpose of waste incinerators as disposal facilities (not power plants) and is close to results from alternative economical allocation calculated in (Doka 2003:22) where burden on disposal would have been 93%.

---

[16] A text might read for instance *"100% allocation on disposal function in EcoSpold1. Alternative allocation with economical key would put 99.22% of the activity burden on the disposal function, 0.6698% on generated net electricity and 0.1144% on generated net useful heat"*.

Tab. 6.1 Base data for creation of the comment text on alternative economical allocation. Not actually used in the inventory figures. The price unit of Euro of 2005 is adopted from price properties in the ecoinvent database v3.6.

Product or co-product	Unit	Price per unit EUR2005	Source
Waste incineration service	kg	0.3225	1
Wastewater treatment	m^3	0.9675	2
Municipal landfilling	kg	0.08	3
Produced net heat	MJ	0.00412	4
Produced net electricity	kWh	0.04352	4
Steel scrap	kg	0.175	5
Aluminium scrap	kg	0.806	5
Copper scrap	kg	2.84	5
Zinc concentrate (hydroxide sludge)	kg	0.0042	5, 7
Nitrogen fertiliser (as N)	kg	0.4515	6
Phosphorus fertiliser (as P_2O_5)	kg	0.7095	6
Potassium fertiliser (as K_2O)	kg	0.3225	6

1 Derived from an average disposal fee of 0.5 CHF per kilogram in Switzerland ca. 2005 for the average 2.2 person household (Rohrer 2006:16) and a conversion factor of 0.645 EUR/CHF for 2005 (https://freecurrencyrates.com/en/exchange-rate-history/CHF-EUR/2005/cbr).

2 Derived from an average treatment fee of 1.5 CHF per m^3 wastewater in Switzerland ca. 2005 for the average 2.2 person household (Rohrer 2006:13) and a conversion factor of 0.645 EUR/CHF for 2005.

3 Median value for landfilling of non-hazardous waste in EU-28 (EEA 2013). Ranging from 5 EUR/t (Bulgaria) to 155 EUR/t (Sewden).

4 Derived from average revenue from all 27 Swiss MSWI in 2002 for sold heat and electricity (0.023 and 0.068 Fr/kWh respectively) from Dettli et al. (2004:19+22) and a conversion factor of 0.645 EUR/CHF for 2005. Inflation of Swiss Francs from 2002 to 2005 is negligible (https://de.inflation.eu/inflationsraten/schweiz/historische-inflation/vpi-inflation-schweiz.aspx). Please note that in the table heat has the unit MJ = 3.6 kWh.

5 Price properties of pertinent exchanges in ecoinvent database v3.6 (Sept 2019), undocumented source.

6 Fertiliser prices in Switzerland 2004 from (Raaflaub & Genoni 2005) converted with 0.645 EUR/CHF. Lower end products were used since this is for sewage sludge.

7 This price is for the generic Zn content in the used exchange "zinc concentrate" of 59% dead from master data "Product Information" from ecoinvent database v3.9.1 (Dec 2022), undocumented source

7 Legacy waste compositions

The residual material landfill is for mostly inorganic polluted wastes, and its model is used to inventory the disposal of various specific waste materials from particular industrial processes. Most of those waste compositions of materials going in to residual material landfill were described in (Doka 2009-I:chapter 4) for ecoinvent v2.1 using the previous residual landfill model created in 2003 (Doka 2003-III). A list of those wastes and a short description is shown below. The description is the text used for the provided "recommended use" text for wastes in residual material landfill (see point 26 in Calculation Manual, Doka 2023).

Some new and updated compositions are shown in the next chapter.

Waste name	Description
decarbonising waste, 30% water	The waste is a carbonate-rich waste obtained during decarbonisation of water (cf. Doka 2003-I on Waste material compositions).
drilling waste, 71.5% water	The waste is the obtained from drilling operations in oil and gas production (cf. Doka 2003-I on Waste material compositions).
waste, Si waferprod., inorg, 9.4% water	The waste is a mixture of various inorganic wastes generated during production of silicon wafers for photovoltaic cells (cf. Doka 2003-I on Waste material compositions).
H3PO4 purification residue, 0% water	The waste is the inorganic, solid residue created during the purification step of phosphoric acid production (cf. Doka 2003-I on Waste material compositions).
green liquor dregs, 25% water	The waste is the insoluble residues from green liquor in paper pulp production, where green liquor is an aqueous solution of the ashes from black liquor incineration (cf. Doka 2003-I on Waste material compositions).
ash from deinking sludge, 0% water	The waste is the residual ash from the incineration of sludge generated in the deinking process step in the recycling of printed paper (cf. Doka 2003-I on Waste material compositions).
sludge, NaCl electrolysis, 0% water	The waste is the inorganic brine filtration sludge produced in chlorine-alkali electrolysis of sodium chloride (cf. Doka 2003-I on Waste material compositions).
sludge, NaCl electrolysis Hg, 0% water	The waste is the inorganic brine filtration sludge produced in chlorine-alkali electrolysis of sodium chloride with mercury cells (cf. Doka 2003-I on Waste material compositions).
pollutants from rail ballast, 0% water	The waste is the removed fines in the recycling of gravel and sand used in the supporting base (ballast) of a railway track (cf. Doka 2003-I on Waste material compositions).
cement, hydrated, 0% water	The waste is hydrated, used cement (composition updated from Doka 2020 on demolition waste landfills). Used for additional solidification of various wastes in residual material landfill.
ash from paper production WWTP sludge, 0% water	The waste is the residual ash from the incineration of treatment sludge generated in the internal wastewater treatment of paper production (cf. Doka 2003-I on Waste material compositions).

carbon SPL, Al elec.lysis, 0% water	The waste is the carbon part (first cut) of a used pot lining (spent pot lining SPL) which is the carbon-ceramic composite used as cathode container in aluminium electrolysis from bauxite (cf. Doka 2003-I on Waste material compositions).
refractory SPL, Al elec.lysis, 0% water	The waste is the ceramic part (second cut) of a used pot lining (spent pot lining SPL) which is the carbon-ceramic composite used as cathode container in aluminium electrolysis from bauxite (cf. Doka 2003-I on Waste material compositions).
filter dust Al electrolysis, 0% water	The waste is the filter dust collected from the air in primary alumina electrolysis (cf. Doka 2003-I on Waste material compositions).
dross from Al electrolysis, 0% water	The waste is a solid waste generated in alumina electrolysis (cf. Doka 2003-I on Waste material compositions).
redmud from bauxite digestion, 0% water	The waste is an inorganic tailings slurry called redmud produced during the purification (digestion) of raw bauxite ore (cf. Doka 2003-I on Waste material compositions).
salt tailings potash mining, 0% water	The waste are non-target salts and minerals separated during the mining and refining of potash (KCO3, KOH) (cf. Doka 2003-I on Waste material compositions).
sludge, pig iron production, 8.6% water	The waste is a gas scrubber sludge generated in pig iron production (cf. Doka 2003-I on Waste material compositions).
slag, unalloyed electr. steel, 0% water	The waste is slag from production of steel from scrap metal in electric arc furnaces, EAF (cf. Doka 2003-I on Waste material compositions).
dust, unalloyed EAF steel, 15.4% water	The waste is the filter dust collected from the air in production of *unalloyed* (carbon) steel from scrap metal in electric arc furnaces, EAF (cf. Doka 2003-I on Waste material compositions).
dust, alloyed EAF steel, 15.4% water	The waste is the filter dust collected from the air in production of *alloyed* (chrome) steel from scrap metal in electric arc furnaces, EAF (cf. Doka 2003-I on Waste material compositions).
basic oxygen furnace wastes, 0% water	The waste is a weighted average of desulphurisation slag and furnace slags created during steel making in basic oxygen furnaces, BOF (cf. Doka 2003-I on Waste material compositions).
sludge from steel rolling, 20% water	The waste is a treatment sludge generated from the internal wastewater treatment of cooling and process water used in steel rolling and drawing. (cf. Doka 2003-I on Waste material compositions).
nickel smelter slag, 0% water	The waste is a slag produced from primary nickel smelters (cf. Doka 2003-I on Waste material compositions).
catalyst base CH2O production, 0% water	The waste consists of the unrecycled catalyst matrix material used in the production of formaldehyde (CH2O) (cf. Doka 2003-I on Waste material compositions).
catalyst base Eth.oxide prod., 0% water	The waste consists of the unrecycled catalyst base material used in the production of ethylene oxide (cf. Doka 2003-I on Waste material compositions).
residues Na-dichromate prod., 0% water	The waste is a residue generated during the processing of chromium ore during the production of sodium dichromate (Na2Cr2O7) (cf. Doka 2003-

	I on Waste material compositions).
residue from TiO2 prod. SO4, 30% water	The waste is an inorganic solid residue (called digester residue) generated in the production of titanium dioxide (TiO2) with the *sulfate* process (cf. Doka 2003-I on Waste material compositions).
residue from TiO2 prod. Cl, 56% water	The waste is an inorganic solid residue (called neutralised spray vessel solid) generated in the production of titanium dioxide (TiO2) with the *chloride*process (cf. Doka 2003-I on Waste material compositions).

8 Updated waste compositions

Waste into residual material landfill with unchanged composition data was described in the preceding chapter. Since 2009, some updated or additional waste compositions were created, which are described below.

In ecoinvent v2.2 and UVEK 21 almost all residual wastes are modelled to be landfilled in a Swiss geography (CH).[17] This is in most cases not appropriate, as the waste-producing industrial activities are not in Switzerland in the real world, but a landfill model in a Swiss climate was the only one available in 2003. For future datasets the location/climate could be adjusted to the countries and climates where the respective industries are located in the real world. Specific climate data from pertinent counties could be used. To reduce the number of datasets also the approach of *infiltrations classes* introduced in (Doka 2018:Tab 4.1) can be used, where instead of datasets for potentially some 200+ regions in the world, only 5 infiltrations classes are used, which represent a range of typical climates. When only a few producing countries are inventoried, using country-specific climate data is more accurate.

8.1 Average incineration residue, 0% water

This waste is an average solid residue from the incineration of municipal solid waste. It consists of the collected fly ashes and the scrubber sludge. The updated composition is based on the updated incinerator model of (Doka 2013) which featured updated incinerator transfer coefficients and updated municipal solid waste composition. The composition is (wet, in ppmw): H2O na; O 383510; H na; C 23003; S 23821; N 8.7626; P 4930.5; B 40.551; Cl 87663; Br 2869.7; F 8880.4; I 0.18135; Ag 6.5144; As 31.671; Ba 831.52; Cd 333.2; Co 7.0076; Cr 517.27; Cu 2548.9; Hg 28.032; Mn 819.3; Mo 10.987; Ni 71.922; Pb 5870.3; Sb 1474.3; Se 13.502; Sn 2164.1; V 41.653; Zn 26937; Be 1.1681; Sc na; Sr na; Ti 15534; Tl na; W na; Si 67641; Fe 9108.9; Ca 135070; Al 56995; K 55666; Mg 10568; Na 72988.

8.2 Frit for CRT tube production

The frit is a kind of a "ceramic glue" to combine the glass funnel and the glass screen of a cathode ray tube (CRT). During CRT production around 4.8 w% of the used frit is wasted and is being landfilled. The waste composition is given in (Lehmann & Hischier 2007-III:175) as 69.624% lead, 9.641% zinc, 2.7949% boron, 1.7913% barium, 0.93493% silicon, 15.214% oxygen.

8.3 Slags from copper, tin, or zinc smelters

David Fitzgerald compiled three missing non-ferrous metal slags from literature in 2019 to include in ecoinvent. They are for copper slags, tin slags, or zinc slags, respectively.[18] Copper slag composition

[17] Exceptions are the hard coal ashes which were inventoried for 12 different countries (AT, BE, CZ, DE, ES, FR, HR, IT, NL, PL, PT, SK) dependent on the country-specific coal composition and were given the respective geography (although the 2003 model could only model a landfill in a Swiss climate). The other exception is lead smelter slag, intended to be a typical global average, and therefore a GLO geography was given to the disposal dataset (although also here the landfill model is for a Swiss climate). And in datasets created in ecoinvent v3+ an compulsory GLO forerunner datasets must be created before any country-specific datasets can be created.

[18] A nickel smelter slag was already contained in (Doka 2003-I).

was compiled from (Lim & Chu 2006). Tin slag composition was compiled from (Rustandi et al. 2018) and (Jamil Hashim et al. 2018). Zinc slag composition was compiled from (MK 1992:Tab 4-3). In all compositions the oxygen content was adjusted to attain 100% mass without gaps.

Activities for the disposal of these three slags in a residual material landfill were created for ecoinvent v3.6ff, but in a global geography (GLO). The datasets were however based on the old residual landfill model in a Swiss climate.

Tab. 8.1 Composition of smelter slags for copper, tin and zinc smelters, respectively

	Copper slag kg/kg wet	**Tin slag** kg/kg wet	**Zinc slag** kg/kg wet
H2O			
O	0.36679	0.518470005	0.49857055
H			
Org.-C			
S			
N			
P		0.029637	
B			
Cl			
Br		0.000002745	
F			
I			
Ag	0.00004		
As	0.00063	0.0001029	0.000263
Ba	0.0024	0.00042	0.001101
Cd	0.00004		0.000016
Co		0.0000261	0.00001485
Cr	0.00028	0.00018	0.00036795
Cu	0.00807		0.0017175
Hg	0.00002		
Mn	0.0004		0.06405
Mo			
Ni	0.00211		0.00021695
Pb	0.00071		0.0005585
Sb		0.00000325	0.00002875
Se	0.00007		
Sn		0.032295	
V			0.00005125
Zn	0.00606	0.00024	0.0666
Be			0.0000017
Sc			
Sr			
Ti		0.040995	
Tl			
W			
Si	0.155	0.18557	0.144
Fe	0.379	0.06085	0.153
Ca	0.00713	0.040309	0.02735
Al	0.0267	0.069861	0.022
K	0.0317		0.0066
Mg	0.00485		0.012685
Na	0.008	0.021038	0.000807

8.4 Slags from Antimony and Ferro-molybdenium smelters

Avraam Symeonidis compiled three missing non-ferrous metal slags from literature in 2021 to include in ecoinvent.

Two different antimony slags (desulfurised and quenched) were compiled from (Guo et al. 2014:Tab.1+2). Ferro-molybdenum slags are the leftovers of alumino-thermic smelting reaction of molybdenum ores and silicon, iron, aluminium and other metals. A composition was compiled from three sources (Boehme & Van Den Hende 2011, Chen et al. 2018, Kornievskiy et al. 2015).

The composition of ferro-molybdenum slag was now corrected, as in the original compilation of 2021 errors were made in the elemental conversion of oxides (and review comments pointing out those errors were ultimately disregarded by the author). As a result the moybdenium content is now 30% lower. Oxygen is adjusted to fulfil a 100% mass balance.

Tab. 8.2 Composition of smelter slags for antimony and ferro-molybdenum smelters.

	Desulfurised antimony slag	Water-quenched antimony slag	ferro-molybdenum slag
	kg/kg wet	kg/kg wet	kg/kg wet
H2O			
O	0.40094	0.5125	0.4807
H			
Org.-C			
S	0.0054	0.1622	
N			
P	0.002662		
B			
Cl			
Br			
F		0.0708	
I			
Ag			
As	0.0057	0.0007	0.000007746
Ba			
Cd			1.7321E-07
Co	0.0000326	0.00000229	
Cr	0.000213	0.0000283	0.00018099
Cu	0.000139	0.00000956	0.0002535
Hg	0.00000263	0.000466	2.2361E-07
Mn	0.0010842		
Mo			0.0032457
Ni	0.0000247	0.0000114	0.000061164
Pb	0.0000517	0.0000202	0.00007604
Sb	0.0111	0.00693	
Se			
Sn	0.00000274	0.00000043	
V			
Zn	0.000665	0.000019	0.00023004
Be			
Sc			
Sr			
Ti	0.0032365		
Tl			
W			
Si	0.18885	0.013837	0.27243
Fe	0.15247	0.0017485	0.094998
Ca	0.15295	0.21584	0.040096
Al	0.045145	0.0080446	0.075789
K	0.0046488		0.0060601
Mg	0.012302	0.0037992	0.015264
Na	0.012389	0.0030416	0.010608

8.5 Residue from rutile production, synthetic, 56% water

This waste is for an inorganic solid residue (called neutralised spray vessel solid) generated in the production of titanium dioxide (TiO2) with the chloride process.

A material with this name was created for ecoinvent v3+, but it is merely a copy of "residue from TiO2 prod. Cl, 56% water" from (Doka 2003-I) with the same pollutant contents. This disposal was created as an approximation (proxy), to be updated, when a more pertinent waste composition became available, but conveniently without the waste-producing inventory figures having to be changed. This proxy still remains in the ecoinvent database unchanged (situation for ecoinvent v3.9.1 of 2022). This proxy waste is produced in the processes "rutile production, synthetic, 95% titanium dioxide, Becher process//AU" and "rutile production, synthetic, 95% titanium dioxide, Benelite process//IN"

The name of the new proxy exchange for EcoSpold2 is "residue from rutile production, synthetic, 56% water" (UUID 93ecf39b-b753-44a6-b58a-c44d87ca7186), with the water content given in the name (which is atypical for ecoinvent v3+).

A disposal activity for "residue from rutile production, synthetic, 56% water" in residual material landfill was created in ecoinvent v3+, but in a global geography (GLO). This dataset was however based on the old residual landfill model in a Swiss climate.

The ecoinvent v3+ database contains also the original waste from ecoinvent v2.2 under the EcoSpold2 name "residue from TiO2 production, chloride process" (UUID cdbd9da0-1040-4894-b349-e346828a8b35).

9 Calculation Manual

The elaborated disposal models for slag compartments and residual material landfills are implemented into an Excel calculation workbook and integrated into the suite of disposal Excel workbooks, which can export EcoSpold1 and EcoSpold2 process inventory files. The slag compartment model is only an assistant model to municipal waste incineration for the landfilling of higher order waste from bottom ash. The residual material landfill model is both an assistant model to municipal waste incineration (for landfilling of fly ashes and scrubber sludge) as well as a stand-alone model for direct disposal of specific first order wastes into residual material landfills.

The tools include a centralised repository for waste composition definitions, site parameters like climate, EcoSpold2 Master Data etc. The usage of the tools is described in updated report (Doka 2023).

10 Results for residual material landfill

The LCIA results of 36 different materials landfilled in a residual material landfill are shown in Fig. 10.1.[19] A Swiss climate is assumed here and data is taken from the compilation of (Doka 2018) which is population-weighted precipitation data (instead of simply area-averaged) which is considered more pertinent for landfill activities. The results are coarsely grouped into generating sectors: chemicals, energy, iron, other metals, paper and some others.

The modelling of waste-specific emissions results in clear distinctions of different waste materials, encompassing over two orders of magnitude. Compared to merely generic residues, burdens for specific waste materials can be over a factor 50 lower or a factor 10 higher, emphasizing the gravely distorting effects in either direction, if only one generic, unspecific waste were to be inventoried for a waste into residual material landfill.

In wastes with little pollutant content the infrastructure and processing burdens are dominant—marked with a small horizontal line in Fig. 10.1 (log-scale) and the grey column contribution in Fig. 10.2 (linear scale). This part can include burdens from solidifying cement and if this is the case it is marked here with a suffix # in the waste name. The addition of cement rises the processing burden significantly.

The contributions to the LCIA burdens is shown in Fig. 10.2. Contributions are from infrastructure & processing, from all short-term emissions, from four particular long-term emissions (**arsenic, barium, phosphorus, zinc**), and from all remaining long-term emissions. Arsenic, barium, phosphorus and zinc show to be relevant for a range of different waste materials.

Chromium and **vanadium** becomes very relevant for "slag, unalloyed electr. steel". Chromium emissions—not unexpectedly—also have a large contribution to "residues Na-dichromate prod.". Vanadium emissions also have some relevance for "redmud from bauxite digestion" and "residue from TiO2 prod. Cl".

---

[19] ReCiPe'13 endpoint LCIA is used since it has characterisation of groundwater emissions – the main emissions from a landfill – while for instance the Swiss ecoscarcity LCIA method has none and is therefore not suited to assess landfill processes.

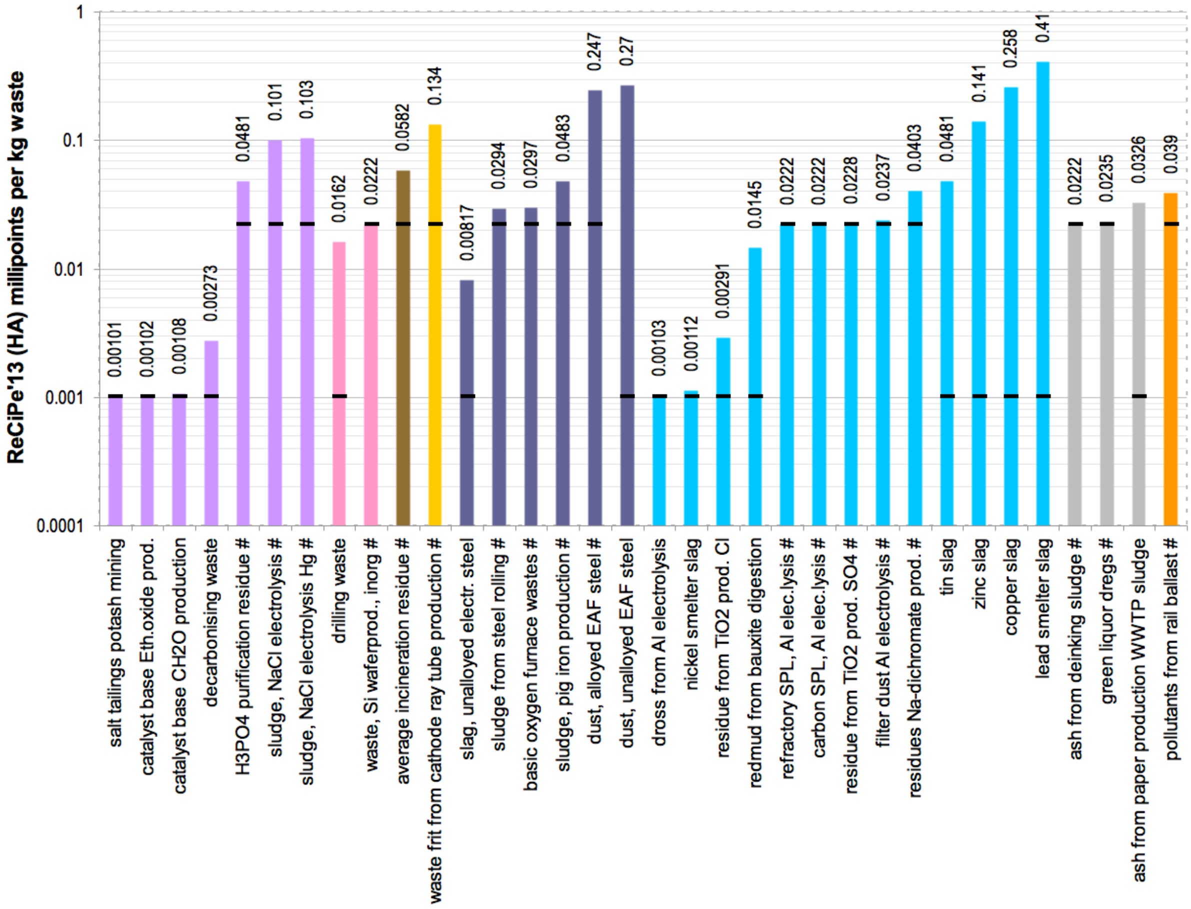

Fig. 10.1 LCIA results on a logarithmic scale for 36 waste materials in residual material landfill grouped into waste-generation branches: chemicals (purple), energy (pink), waste management (brown), electronics (yellow), iron (dark purple), non-iron metals (cyan), paper (grey), and transport (orange). The small horizontal line indicates the burden from landfill infrastructure and processing and includes the waste-specific choice of cement solidification. The column above that mark therefore represents on the log-scale the burdens from the waste-specific direct landfill emissions to water. A # suffix in the waste materials name denotes that it is landfilled with cement solidification.

The short-term emissions (0–100 years) end up to be negligible contributions in all datasets (blue bar in Fig. 10.2). In contrast, the long-term emissions (100–60'000 years) are frequently the dominant burdens, except for waste with low pollutant potential like salt tailings or dross from aluminium electrolysis, where infrastructure and processing is dominant.

The relevance of long-term emissions, which is confirmed here, underlines the general importance of looking at long timeframes to capture the burdens in systems with **slow temporal dynamics**. In systems like soil or landfills, where water movement is comparatively slow, is crucial to look at **appropriately long timescales** in order to determine what the burdens caused by an activity actually are. Within the precepts of LCA, the temporal scope of assessment must be appropriate for the analysed system. If burdens in a slow system are to be determined, suitable long timescales to recognize those burdens are therefore compulsory. It would conversely be wrong to look at a process with *air emissions*, but limit the damage assessment model to only the *first millisecond* after pollutant release. This period is much too short to appropriately capture the dynamics of an atmospheric system in a meaningful manner. That a millisecond is short and 60'000 years is long is a subjective and anthropocentric view of humans as land-dwelling mammals, who are very familiar with air and surface water in their everyday lives, but unfamiliar with the environment of underground soil. The familiarity or unfamiliarity of laypeople has no bearing on properly assessing an investigated system's effects. If the system under investigation has consequences for a very long time in the future—be it by design or by serendipity—it is the function of LCA to be able to point out the extent of those consequences accordingly. Long-term burdens must therefore be included in systems with landfill processes. Process inventories and assessment methods that fail to do so, are unsuitable in LCA.[20]

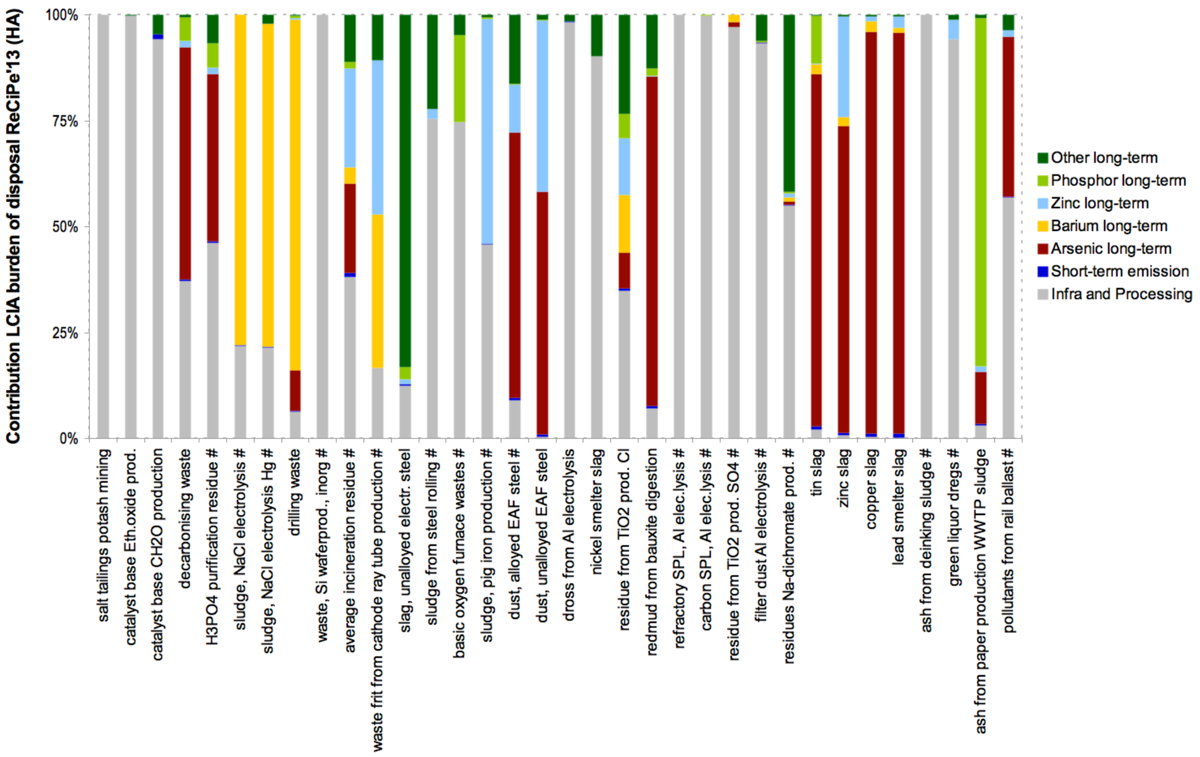

Fig. 10.2 **Contribution analysis of the LCIA results for 36 disposal datasets in residual material landfill. In the same ordering sequence as in the previous chart.**

It might feel like including long timeframes in LCA are extreme choices which are maximising burdens. This concern is without foundation and not confirmed by the model data. For the modelled waste materials, typically only 40±10 w% of the total landfilled waste mass is emitted in the long-term. The remainder 60% mass stays in the landfill even after 60'000 years of weathering. The emitted mass typically represents only 20%±15% of the toxicity potential of a waste material.[21] The established burdens therefore do not represent extremes or worst cases – which would be 100% of the toxicity potential emitted. The various landfill models in (Doka 2017, 2020) give similar midway and non-extreme results. The models are able to differentiate emission behaviour of different landfill types. They do neither maximise nor minimise the projected emissions and are therefore suitable to present an appropriate picture of the burdens set to be brought about by putting a waste in a landfill.

---

20 Introduction of long timescales in LCA began with inclusion of long-term air emissions of radioactive radon (Rn-222) from the waste of uranium ore processing (tailings). In those waste materials the radon emissions are fed by ongoing decay of radioactive isotopes with lifetimes of 770'00 years. The emissions of Radon-222 were integrated over a time frame of 110'000 years in order to capture 63% of the expected long-term air emission burdens caused by uranium ore processing (Dones & Zollinger 1994:45).

21 These are the typical values found in the datasets for a Swiss climate, but they are not generally valid for just any waste material or any climate. The waste-specific and climate-specific models can result in different values.

11 Troubleshooting

The workbook are interlinked and in rare cases of recursion or unopened required workbooks the inventory amounts can exponentially grow towards infinity, instead of converging on a constant amount—so-called "recursion catastrophes". Or the reverse: amounts that should be zero, but were non-zero in previous calculations can take a long time in the recursion to fall below 10^{-307} (which is the smallest positive non-zero number in MSExcel). To clean the calculation from any remainders of such exponential catastrophes, click the button "clean calc" located in the top left corner of sheet 'DS info' (below the ES2 and ES1 export buttons).

The "clean calc" macro temporarily decouples all the workbooks, effectively unmaking the full integration, and makes the selected initial waste the only input for all workbooks. It then calculates one step and then re-establishes full integration links again.

12 Summary

The present project updated the treatment models of slag compartments and residual material landfills using a larger number of real world measurements. For slag compartments the datapoints the models is based on was increased by over a factor 5 (from 790 datapoints to over 4200 datapoints). For the residual material landfill model the datapoint count was increased by a over factor 10 (from 200 datapoints to over 2000 datapoints). This is a considerable improvement for capturing the typical chemical environment these types of landfills represent. At the same time the models have been regionalised, meaning made dependent on local climate data. This provides more accurate data for a range of locations with different climatic conditions a user can now heed.

The updated slag compartments and residual material landfills now also feature a regionalisation, meaning that the user can choose to model landfills in various climates by specify precipitation rates and evaporation rates. This will influence leachate generation, landfill weathering, and calculated emissions in the model.

Lastly the suite of Excel waste tools underwent what is called a "Full Integration". With full integration, the various calculation models are linked up and exchange process inventory results, for instance when leachate from a sanitary landfill is treated in wastewater treatment plant, or when sludge from a wastewater treatment plant is disposed in a sanitary landfill treated in a municipal incineration. (while in previous models the treatment of higher order wastes was also considered dynamically and waste-specifically, but using a fixed-factor approach which represented only a particular, fixed treatment model setting). The new full integration approach means that the models have grown more complex and a user of the tools needs not only to provide accurate parameters for the initial foreground treatment, but also for the required treatment processes of higher order wastes.

13 Outlook

In the waste incineration model, the fate of the bottom ash is landfilling in either a slag compartment or (per user choice) in sanitary landfill. Although some metal recycling can be included, these two destinations are the only ones in the model. In the real world incinerator bottom ash (IBA) might also be recycled in the building industry for instance as road materials or in cement production or in concrete.

In a cut-off system methodology, the recycling of bottom ash would result in a cutting-off of the burdens for landfilling, which would be easily implemented in the model due to the modular mass flow accounting principles used in the model. But in system methodologies with allocation (e.g. APOS), the waste-specifity should be maintained. Meaning that the likely increased pollutants loads in building materials and resulting larger downstream emissions during production, use phase and dismantling should be heeded.

If building materials with secondary raw materials in them are inventoried identically as building materials from primary raw materials, it can be seen as favouritism of recycling, since the possibly increased emissions from that recycling are ignored. A similar issue presents itself already now with the zinc concentrate from Swiss waste incineration produced from fly ash extraction (FLUWA). Within the model framework, the zinc concentrate output is a waste-specific fractional division of the real world zinc concentrate output of very variable composition.[22] The zinc concentrate goes into Waelz kilns to utilize the zinc content, creating various emissions and slag outputs. At that stage the waste-specifity is lost in current life cycle assessment calculation since the input to the Waelz kiln is only modelled as generic average. The fate of many pollutants originally in the incinerated waste producing the zinc concentrate is not modelled accurately anymore, leading to over- or underestimations.

This appears to be a formidable problem and the current LCI database architectures seem not to be the right approach for this: If waste-specifity is heeded, it means for instance every one building product with real-world IBA content is replaced by *dozens* or *hundreds* of products (one for each incinerated waste material). In the recycling/production processes the changed pollutant content can then be heeded using mass flow accounting principles, e.g. from changed feed into cement kilns. The processes *using* these secondary products would also need to be able to trace the pollutant fate from input to output channels (e.g. leaching from roads, or facades. Or into waste treatment). The current dataset model architecture of LCA software is not well equipped to trace such a "pollutant relay race" between different processes.

Before approaching solutions of this problem of modelling waste specific pollutant signals in upstream recyclate processes, checking the general relevance of such emissions can be enlightening. But even if such an analysis reveals little relevance in an average situation, burden signals for specific, non-average waste materials might become relevant.

[22] Since it is waste-specific the zinc concentrate does not even to contain zinc, if the original waste contains no zinc. The composition then represents the elements going into this output channel.

14 Glossary

1O	"One-Oh". Abbreviation of "first order" relating to the initial waste material or its treatment. Depending on the treatment technology, a 1O treatment might cause a subsequent chain of higher order downstream wastes (→HO). The initial, first order waste might be also seen as the 'primary waste' causing 'secondary' and other higher order wastes. The wording in analogous to "primary resources" and "secondary resources". I.e. the secondary materials are formed from the initial primary materials.
AEI	"Additional Environmental Information": a result category in the environmental product declarations (EN 15804) separate and different from Impact Assessment Results (LCIA). E.g. total mass of generated hazardous waste. Cf. chapter 5.6 'Elementary Exchanges for EN 15804' on page 29.
IBA	Incinerator bottom ash. Bottom ash from waste incinerators (as opposed to bottom ash from coal furnaces FBA).
aUPR	Aggregated UPR. This relates to an UPR of a waste treatment activity where any necessitated treatments of higher order wastes are added into the same UPR, and not in separate UPRs. The waste treatment datasets generated from the Excel waste disposal tools contain also any treatment of higher order wastes in aggregated form. This is mainly done to conserve database space and not create a multitude of dataset which have precisely one single purpose within the whole database.
Full integration	Within the realm of the Excel waste tools by Gabor Doka, full integration means that the treatment of generated secondary and higher order waste (→ HO. For instance sewage sludge from wastewater treatment) is actively calculated according to the user-defined settings chosen in the tool for that kind of higher order disposal. Previously without full integration the treatment of higher order wastes was already included waste-specifically in the generated treatment inventories, but based on a fixed-factor model representing a fixed set of operational parameters for that treatment, but still maintaining the core concept of delivering waste-composition-specific results for that treatment.
HO	"higher order". Abbreviation for materials and processes relating to secondary, tertiary and quaternary waste, i.e. downstream of an initial waste treatment (→1O). Examples of higher order wastes would be bottom ash generated from municipal waste incineration, or sewage sludge generated from wastewater treatment. In the Excel waste disposal tools, the HO wastes are considered with their non-average, waste-specific composition, i.e. depend on the composition of the initial waste.
Landfill type C	(VVEA 2016), a.k.a. residual material landfill. A landfill for largely inorganic with high pollutant load. Filter ashes and scrubber sludge from Swiss waste incinerators are frequently landfilled in residual material landfills. But they can also receive waste from other sources.
Landfill type D	(VVEA 2016), a.k.a. Slag compartment.. A landfill for waste incinerator bottom ash.

Recursions	Within the realm of the Excel waste tools by Gabor Doka, a recursion occurs if a higher order waste (→ HO) is being fed back as an input to a treatment that has already been involved upstream in the process chain. For instance a sanitary landfill might produce some leachate, going into a wastewater treatment, which might produce some sewage sludge, which in turn could go back to the sanitary landfill. The secondary sewage sludge in this example is a recursive waste. Generally in mathematics a recursion occurs when the result of a calculation depends on its own result, i.e. the result of a calculation is being fed back as an input into the same calculation.
RMLF	Residual material landfill. See → landfill type C
SC	Slag compartment. See → landfill type D
UPR	Unit Process inventory. An inventory containing the technosphere and biosphere exchanges of a particular activity. In waste treatment UPRs the UPRs are often not relating to an average operation of a real world activity (e.g. incineration of average mixed municipal waste), but to a calculated part of that operation relating to a particular waste material (e.g. incineration of polyvinylchloride polymer). I.e. UPRs are usually non-average and waste-specific to reflect the consequences of treatment of a particular waste material.
GLO	Geographic designation of a dataset nominally referring to a global average.
RoW	"Rest of the World". Geographic designation of a dataset derived as the complement of available specific national inventories and the global average.

15 Appendix A

For the working point model of the updated slag compartment and residual material landfill model measured data on typical leachate composition and average landfill contents was compiled from following literature sources.

Åberg et al. 2006:Fig 3-6; AIB 1993; Alwast & Riemann 2010:Tab 4; Anthonissen et al. 1993; Astrup et al 2016:Tab 24.1; Astrup et al. 2006:Tab S1; AWEL 2007, 2012, 2014, 2015, 2016, 2021, 2018b; Baur et al. 1999; Baur et al. 2001; Belevi & Langmeier 2000:Tab 2; Belevi & Moench 2000:Tab 1; Belevi et al. 1992:Tab 1; Birgisdottir 2005:Tab 1; Bisinella et al. 2016:Tab S17; BLU 1983; BMG 2010; BMG 2011; BMG 2012; Bogush et al. 2015:Tab 2; Bogush et al. 2019:Tab 1; Böhmer et al 2007:Tab 13; Böhmer et al 2007:Tab 37; Bösch et al. 2011:Tab 2; Bösch et al. 2011:Tab A7.1; Bouvier et al. 2005:p.212; Bouvier et al. 2005:Tab 14,15,17; Bouvier et al. 2005:Tab 18; Bouvier et al. 2005:Tab 21, 22, 24; Bouvier et al. 2005:Tab 25; Bouvier et al. 2005:Tab 32; Bouvier et al. 2005:Tab 39; Bouvier et al. 2005:Tab 46; Bouvier et al. 2005:Tab 53; Bouvier et al. 2005:Tab 57; Bouvier et al. 2005:Tab 58; Bouvier et al. 2005:Tab 59; Bouvier et al. 2005:Tab 6,10; Bouvier et al. 2005:Tab 60; Bühler & Schlumberger 2010 (BAFU):Tab 2; Bühler & Schlumberger 2011 (ISWA); BUWAL 1995; Chandler et al. 1997; Chandler et al. 1997:Tab 3.3; Chandler et al. 1997:Tab 9.19; Chandler et al. 1997:Tab 9.21; Chandler et al. 1997:Table 11.11; Chang et al. 2009:Tab 4; Chen et al. 2008:Fig 2-5; Crannell et al. 2000:Tab 2; Dijkstra 2007:Tab 1; Dijkstra et al 2019:Fig 4; Doka 2003-II; Doka 2013; Eggenberger & Mäder 2002:13f; Eggenberger & Mäder 2010:Fig 11-13; Eggenberger & Mäder 2010:Tab 1; Eggenberger & Mäder 2010:Tab 3; Eggenberger & Mäder 2010:text 125ff.; EKESA 1992; ETH 1992; Faulstich 1993; Frühwirth et al. 1993; Ganguin 2012:Tab 2; Goetz 1989; Gutmann & Vonmont 1994:Tab 1+2; Hermanns & Moser 2012:Anhang A1, p.28ff.; Hjelmar et al 2013:Tab 1; Huber et al 2019:Tab 1; Huber et al. 1996; Huber et al. 1996:p.24; Hyks 2008:Tab 1.; Johnson & Huter 2012:Fig 1; Johnson & Huter 2012:Tab 1; Johnson et al. 1996:Fig. 3+4+5; Johnson et al. 1999:Tab 1; Jutz & Schlumberger 2011:Tab 1; Kahle et al 2016:Tab 1; Karlfeldt Fedje 2010:Tab 3.3; Karpov et al. 2004:Tab 27; Kersten et al. 1998; Klein 2002:Tab 16; Kraxner et al. 2001; Lam et al. 2010:Tab 1; Lam et al. 2010:Tab 2; Lam et al. 2010:Tab 3; Lam et al. 2010:Tab 4; Lechner 2001; Lechner et al. 2010:Tab 13; Leuchs 1990; Lin et al. 2015:Tab 1+2; Lindsay 1979:Tab 9.19; Linsmeyer et al 2009:Tab 13; Löschau 2006:Tab. B-3:; Löschau 2006:Tab. B-4; Ludwig & Johnson 1999; Mehr et al. 2021:Tab 3; Mehr et al. 2021:Tab 3+2; Mocker et al. 2013:Tab 1; Mocker et al. 2013:Tab 2; Morf & Kuhn 2009:Tab 5-2; Morf & Kuhn 2009:Tab 5-3; Morf & Kuhn 2009:Tab 9-15; Morf & Kuhn 2009:Tab 9-3; Morf & Kuhn 2009:Tab 9-4; Morf & Kuhn 2009:Tab 9-5; Morf 2006:Tab 19; Morf 2006:Tab 21; Morf 2006:Tab 22; Morf 2006:Tab. 35; Morf 2010:Fig 15; Morf 2010:Tab 2; Morf et al. 2010:Tab 5.1; Morf et al. 2013; Piantone et al. 2008; Reichelt & Pfrang 1998; Reimann 1989; Reuter & Schirmer 1988; Rey 1992; Rylander & Wiqvist 2011; Sabbas et al. 1998:26; Sales Bandarra et al. 2021:Tab 2; Schachermayer et al. 1994; Schlumberger & Bühler 2013:Fig 14; Schlumberger & Bühler 2013:Fig 5+6; Schlumberger & Bühler 2013:Tab 1; Schlumberger 2011:sl 15; Schlumberger 2011:sl 15; Schweizer 1999:p14; Schweizer 1999:Tab 2; Simon & Holm 2013:Tab 1; Sivula 2012:Tab 2; Skutan et al. 2014:Tab 3+2; Speiser et al. 2002; Stark 1993; Sun et al. 2008:Tab 2; Svensson 2006:p. 122; Syc et al. 2010:Tab 2; Syc et al. 2010:Tab 4; Taverna 2011:Tab 8-10 + Chap. 8.6 Aufschluss 1; Taverna 2011:Tab 8-11 + Chap. 8.6 Aufschluss 1; Todorovic 2010; Weibel 2020:Fig 16; Weibel 2020:Fig 7 + 2; Weibel 2020:Tab 10; Weibel 2020:Tab 11; Weibel 2020:Tab 3; Winter et al 2009:Tab 13; Zeltner & Lichtensteiger 2002; Zimmermann et al. 1996:B.17; Zimmermann et al. 1996:p.B.158; Zimmermann et al. 1996:p.B.159

16 References

Web addresses indicate the source of electronic documents. The subsequent (date in brackets) refers to the date of retrieval.

Åberg et al. 2006 — Åberg A, Kumpiene J, Ecke H (2006) Evaluation and prediction of emissions from a road built with bottom ash from municipal solid waste incineration (MSWI). Science of the Total Environment 355 (2006) 1–12 http://www.bvsde.paho.org/bvsacd/cd43/holker.pdf (5 Sept 2012)

AIB 1993 — Amt für industrielle Betriebe (1993) Mittelwerte Sickerwasser verfestigte Reststoffe. Kanton Baselland, Behördenbericht, Liestal, Switzerland.

Alwast & Riemann 2010 — Alwast H, Riemann A (2010)Verbesserung der umweltrelevanten Qualitäten von Schlacken aus Abfallverbrennungsanlagen. Im Auftrag des Umweltbundesamtes, Dessau-Roßlau, Oktober 2010. http://www.umweltdaten.de/publikationen/fpdf-l/4025.pdf (12 Feb 2012)

Anthonissen et al. 1993 — Anthonissen I., Meijer P. (1993): Informatiedocument AVI-Reststoffen. Rijksinstituut voor volksgezondheid en milieuhygiene, Bilthoven (NL) 1993

Astrup et al 2016 — Astrup TF, Muntoni A, Polettini A, Pomi R, Van Gerven T, Van Zomeren A (2016) Treatment and Reuse of Incineration Bottom Ash. In: Prasad MNV, Shih K (Eds.), Environmental Materials and Waste. Elsevier, pp. 607–637. https://publications.tno.nl/publication/34627970/5nP6r7/b16004.pdf (24 Aug 2022)

Astrup et al. 2006 — Astrup T, Dijkstra JJ, Comans RNJ, van der Sloot HA, Christensen TH (2006) Geochemical Modeling of Leaching from MSWI Air-Pollution-Control Residues. Environ. Sci. Technol., 2006, 40 (11), pp 3551–3557 http://pubs.acs.org/doi/suppl/10.1021/es052250r/suppl_file/es052250rsi20060425_120717.pdf (5 Jan 2013)

AWEL 2007 — AWEL (2007) "Deponiestatistik 2006". Amt für Abfall, Wasser, Energie und Luft (AWEL), Baudirektion Kanton Zürich, April 2007. https://www.abfall.ch/Informationen_Merkblaetter/pdf/Deponiestatistik_2006.pdf (20 Sept 2018)

AWEL 2012 — AWEL (2012) "Deponiestatistik 2011". Amt für Abfall, Wasser, Energie und Luft (AWEL), Baudirektion Kanton Zürich, Mai 2012. http://www.awel.zh.ch:80/internet/baudirektion/awel/de/abfall_rohstoffe_altlasten/zahlen_und_fakten/deponiestatistik/_jcr_content/contentPar/downloadlist/downloaditems/deponiestatistik_des_0.spooler.download.1336652586659.pdf/deponiestatistik_2011.pdf (23 Oct 2012)

AWEL 2014 — AWEL (2014) "Deponiestatistik 2013". Amt für Abfall, Wasser, Energie und Luft (AWEL), Baudirektion Kanton Zürich, Mai 2014. https://awel.zh.ch/content/dam/baudirektion/awel/abfall_rohstoffe_altlasten/abfall/dokumente/statistiken/deponie_statistiken/deponiestatistik_2013.pdf (17 Nov 2018)

AWEL 2015 — AWEL (2015) "Deponiestatistik 2014". Amt für Abfall, Wasser, Energie und Luft (AWEL), Baudirektion Kanton Zürich, Mai 2015. https://awel.zh.ch/dam/baudirektion/awel/abfall_rohstoffe_altlasten/abfall/dokumente/statistiken/deponie_statistiken/deponiestatistik_2014.pdf (20 Sept 2018)

AWEL 2016 — AWEL (2016) "Deponiestatistik 2015". Amt für Abfall, Wasser, Energie und Luft (AWEL), Baudirektion Kanton Zürich, April 2016. https://awel.zh.ch/content/dam/baudirektion/awel/abfall_rohstoffe_altlasten/abfall/dokumente/statistiken/deponie_statistiken/deponiestatistik_2015.pdf (17 Nov 2018)

AWEL 2018b — AWEL (2018b) Daten aus Deponiestatistik Kanton Zürich - Auswertung nach Kompartimenten. Amt für Abfall, Wasser, Energie und Luft (AWEL Baudirektion Kanton Zürich), Abteilung Abfallwirtschaft & Betriebe, Sektion Abfallwirtschaft. https://awel.zh.ch/content/dam/baudirektion/awel/abfall_rohstoffe_altlasten/abfall/googlechartskompartimente/index.html via https://awel.zh.ch/internet/baudirektion/awel/de/abfall_rohstoffe_altlasten/zahlen_und_fakten/auswertung_3spaltig.html in tab "Auswertung nach Kompartimenten" (19 Nov 2019)

AWEL 2021 — AWEL (2021) "Deponiestatistik 2020". Amt für Abfall, Wasser, Energie und Luft (AWEL), Baudirektion Kanton Zürich, Dez 2021. https://www.zh.ch/content/dam/zhweb/bilder-dokumente/themen/umwelt-tiere/abfall-rohstoffe/abfallwirtschaft/jahresberichte/deponiestatistik_zh_2020.pdf (18 Jul 2022)

Baur et al. 1999 — Baur I., Ludwig C., Johnson C. A., Stierli M.A (1999) Comparison of Leaching Processes in Field and Laboratory: The Behaviour of cement-stabilised Incinerator Filter Ashes. Presented at INSA'99, First congress of 'Institut National des Sciences', March 13–14, 1999, Poitiers, France. http://people.web.psi.ch/ludwig/insateuftal.pdf (11 Aug 2002)

Baur et al. 2001 — Baur I, Ludwig C, Johnson AC (2001) The Leaching Behavior of Cement Stabilized Air Pollution Control Residues: A Comparison of Field and Laboratory Investigations. Environ. Sci. Technol. Vol.35, p.2817-2822. http://www.internal.eawag.ch/~baur/Baur%20et%20al.%202001.pdf (21 Aug 2001)

Belevi & Langmeier 2000 — Belevi H, Langmeier M (2000) "Factors Determining the Element Behavior in Municipal Solid Waste Incinerators. 2. Laboratory Experiments" Environmental Science & Technology Vol. 34 (2000) pp.2507-2512.

Belevi & Moench 2000 — Belevi H, Moench H (2000) Factors Determining the Element Behaviour in Municipal Solid Waste Incinerators – part 1 Field Studies. Environ. Sci. Technol., Vol. 34, p. 2501-2506.

Belevi et al. 1992 — Belevi H, Stämpfli BM, Baccini P (1992), Chemical Behavior of Municipal Solid Waste Incinerator Bottom Ash in Monofills, Wast Manage. Res. 10, 153-167

Birgisdottir 2005 — Birgisdottir H (2005) Life cycle assessment model for road construction and use of residues from waste incineration. Ph.D. Thesis, Institute of Environment & Resources, Technical University of Denmark DTU, Lyngby, Denmark, July 2005.

http://www.affaldsinfo.dk/files/Filer/F%20og%20U/Ph%20D%20afhandlinger/Life%20cycle%20assessment%20model%20for%20road%20construction%20and%20use%20of%20residues%20from%20waste%20incineration.pdf (29 Jan 2009)

Bisinella et al. 2016 Bisinella V, Conradsen K, Christensen TH, Astrup TF (2016) A global approach for sparse representation of uncertainty in Life Cycle Assessments of waste management systems. The International Journal of Life Cycle Assessment 21(3) p.378–394

BLU 1983 Bundesärztekammer (1995): Gesundheitsgefährdung der Bevölkerung durch Mülldeponien (Siedlungsabfall). Arbeitskreis Gesundheitsgefährdung des Wissenschaftlichen Beirates der Bundesärztekammer, Bundesärztekammer Germany. http://www.bundesaerztekammer.de/30/Richtlinien/Empfidx/Muelldepopdf.pdf (3 Jan 2002)

BMG 2010 anonymous (2011) Analysen-Bericht. BMG Engeneering AG, Schlieren. 7. Jan 2011. http://www.kva-linthgebiet.ch/assets/media/Archiv/Analyse%20Schlacke%2011-01%20A11-00005.pdf (28 Nov 2012)

BMG 2011 anonymous (2011) Analysen-Bericht. BMG Engeneering AG, Schlieren. 14 Sept 2011. http://www.kva-linthgebiet.ch/assets/media/Archiv/Analyse%20Schlacke%2011-09%20A11-01510.pdf (28 Nov 2012)

BMG 2012 anonymous (2012) Analysen-Bericht. BMG Engeneering AG, Schlieren. 10 Mai 2012. http://www.kva-linthgebiet.ch/assets/media/Archiv/Analyse%20Schlacke%2012-03%20A12-00868.pdf (28 Nov 2012)

Bogush et al. 2015 Bogush A, Stegemann JA, Wood I, Roy A (2015) Element composition and mineralogical characterisation of air pollution control residue from UK energy-from-waste facilities. Waste Management 36 (2015) 119–129. https://www.sciencedirect.com/sdfe/reader/pii/S0956053X14005674/pdf (25 Aug 2022)

Bogush et al. 2019 Bogush AA, Stegemann JA, Roy A (2019) Changes in composition and lead speciation due to water washing of air pollution control residue from municipal waste incineration. Journal of Hazardous Materials, vol. 361, pp. 187-199. https://pure.coventry.ac.uk/ws/portalfiles/portal/24040442/Binder2.pdf (25 Aug 2022)

Böhmer et al 2007 Böhmer S, Kügler I, Stoiber H, Walter B (2007) Abfallverbrennung in Österreich - Statusbericht 2006. Report REP-0113. Hrsg: Umweltbundesamt, Wien. https://www.bmk.gv.at/dam/jcr:36a1f2c0-318e-4a40-9ea4-26ead1748758/Verbrennung_Statusbericht2006.pdf (14 Aug 2022)

Bösch et al. 2011 Bösch ME, Haupt M, Hellweg S (201) Ökobilanzielle Untersuchung der sauren Wäsche von KVA Flugasche in der Schweiz. Endbericht (Version 1.7) 10.Okt 2011. ETH Zürich, Institut für Umweltingenieurswissenschaften, Gruppe für Ökologisches Systemdesign und Aveny GmbH, Zürich http://www.awel.zh.ch/internet/baudirektion/awel/de/betriebe_anlagen_baustellen/abfallanlagen/kehrichtverbrennung_kva.html

Bouvier et al. 2005 Bouvier R., Wahl A., Dubois D., Ammann P., Guinand H. (2005) Investigation sur la qualité du mâchefer de 7 usines suisses d'incinération de déchets 2004-2005. SIG service de l'environnement, les Cheneviers and ISDS-Oulens SA

Bühler & Schlumberger 2010 (BAFU) Bühler A., Schlumberger S. (2010) Schwermetalle aus der Flugasche zurückgewinnen: Saure Flugaschenwäsche - FLUWA Verfahren, ein zukunftsweisendes Verfahren in der Abfallverbrennung, in KVA-Rückstände in der Schweiz - Der Rohstoff mit Mehrwert. 2010, Bundesamt für Umwelt (BAFU), Bern. http://www.bafu.admin.ch/publikationen/publikation/01538/index.html?lang=de&lang=de (27 Feb 2012)

Bühler & Schlumberger 2011 (ISWA) Bühler A., Schlumberger S. (2011) Metal recovery from air pollution control residues - filter ash. 7th ISWA Beacon Conference on Waste to Energy in Malmö 3-4 November 2011 http://beacon-wte.net/index.php?id=38 and http://beacon-wte.net/fileadmin/avfallsverige/Documentation_2011/2-5_BSH.pdf (28 Feb 2014)

BUWAL 1995 BUWAL (1995): Reststoffqualität von stabilisierten Rückständen aus der Rauchgasreinigung von Kehrichtverbrennungsanlagen. Umwelt-Materialien Abfall Nr. 44, BUWAL, Berne, Switzerland.

Chandler et al. 1997 Chandler AJ, Eighmy TT, Hartlén J, Hjelmar O, Kosson DS, Sawell SE, van der Sloot HA, Vehlow J (1997) Municipal solid waste incinerator residues, Studies in Environmental Sciences vol. 67, Elsevier, Amsterdam 1997. Cited in Rylander & Wiqvist 2011:Tab.12 and Karlfeldt Fedje 2010:Tab3.3

Chang et al. 2009 Chang CY, Wang CF, Mui DT, Cheng MT, Chiang HL (2009) Characteristics of elements in waste ashes from a solid waste incinerator in Taiwan. Journal of Hazardous Materials 165 (2009) 766–773. http://144.206.159.178/ft/596/606746/12595451.pdf (Bibliographic resource on Nanotechnology, RU) (22 Feb 2012)

Chen et al. 2008 Chen CK, Lin C, Wang LC, Lin YC, Chang-Chien GP (2008) Size Distribution of Metals in Bottom Ash of Municipal Solid Waste Incinerators. J. Environ. Eng. Manage., 18(2), 105-113 http://ser.cienve.org.tw/download/18-2/jeeam18-2_105-113.pdf (22 Feb 2012)

Crannell et al. 2000 Crannell BS, Eighmy TT, Krzanowski JE, Dykstra Eusden Jr J, Shaw EL, Francis CA (2000) Heavy metal stabilization in municipal solid waste combustion bottom ash using soluble phosphate. Waste Management 20 (2000) 135-148. http://www.bvsde.paho.org/bvsacd/cd43/taylor.pdf (5 Sept 2012)

Dijkstra 2007 Dijkstra JJ (2007) Development of a Consistent Geochemical Modelling Approach for Leaching and Reactive Transport Processes in Contaminated Materials. Dissertation Wageningen University and Research Centre, Department of Soil Quality. ISBN 90-8504-607-6. https://library.wur.nl/WebQuery/wurpubs/fulltext/121881 (24 Aug 2022)

Dijkstra et al 2019 Dijkstra JJ, Comans RNJ, Schokker J, van der Meulen MJ (2019) The geological significance of novel anthropogenic materials: Deposits of industrial waste and by-products. Anthropocene 28 (2019) 100229 https://pdf.sciencedirectassets.com/283168/1-s2.0-S2213305419X00077/1-s2.0-S2213305419300402/main.pdf (2 Feb 2020)

Doka 2003-II Doka G (2003) Life Cycle Inventories of Waste Treatment Services, part II: Waste incineration. ecoinvent report No. 13, part III. Swiss Centre for Life Cycle Inventories, St. Gallen, 2003

Doka 2003-III Doka G (2003) Life Cycle Inventories of Waste Treatment Services, part III: Landfills - Underground Deposits - Landfarming. ecoinvent report No. 13, part III. Swiss Centre for Life Cycle Inventories, Dübendorf, 2003. http://www.doka.ch/13_III_Landfills.pdf

Doka 2013 Doka G. (2013) Updates to Life Cycle Inventories of Waste Treatment Services - part II: waste incineration. Doka Life Cycle Assessments, Zurich, 2013. Available at http://www.doka.ch/ecoinventMSWIupdateLCI2013.pdf

Doka 2017-2 Doka G (2017) A model for waste-specific and climate-specific life cycle inventories of open dumps and unsanitary landfilling of waste. Doka Life Cycle Assessments, Zurich, Switzerland. September 2017. Available at http://www.doka.ch/publications.htm

Doka 2018 Doka G. (2018) A model for waste-specific and climate-specific life cycle inventories of tailings impoundments. Report Version 2. Doka Life Cycle Assessments, Zurich, Switzerland. Available at http://www.doka.ch/publications.htm

Doka 2020-5 Doka G (2020) Two models for waste-specific and climate-specific life cycle inventories of excavation material landfills and construction & demolition waste landfills. Doka Life Cycle Assessments, Zurich, Switzerland. Commissioned by Swiss Federal Office for the Environment (FOEN). June 2020. Available at http://www.doka.ch/publications.htm

Doka 2021-6 Doka G. (2021) A model for composition-specific life cycle inventories of regionalised wastewater fates. Doka Life Cycle Assessments, Zurich, Switzerland. Commissioned by Swiss Federal Office for the Environment (FOEN), Berne, Switzerland. Available at http://www.doka.ch/publications.htm

EEA 2013 (2013) European Environment Agency Typical charge (gate fee and landfill tax) for legal landfilling of non-hazardous municipal waste in EU Member States and regions. European Environment Agency (EEA) Created 20 Feb 2013. https://www.eea.europa.eu/data-and-maps/figures/typical-charge-gate-fee-and [sic] (29 Oct 2022)

Eggenberger & Mäder 2002 Eggenberger U., Mäder U. (2002) Charakterisierung von thermisch nachbehandelten Schlackenproben bezüglich Inhaltsstoffen und Auslaugverhalten. Contribution within the SNF Project 5001-058295 Transfer PECK. Institute for Geology, University Berne, Switzerland

Eggenberger & Mäder 2010 Eggenberger U, Mäder U (2010) Charakterisierung und Alterationsreaktionen von KVA-Schlacken. Fachstelle Sekundärrohstoffe, Institut für Geologie, Bern, Switzerland. http://www.bafu.admin.ch/publikationen/publikation/01538/index.html (12 Feb 2012)

EKESA 1992 Belevi H., Johnson C.A., Lichtensteiger T., Zeltner Ch. (1992): Emissions- abschätzung für Kehrichtschlacke unter verschiedenen Ablagerungsbedingungen. Projekt EKESA, Amt für Gewässerschutz und Wasserbau des Kt. Zürich.

ETH 1992 ETH (1992): Reaktordeponie Riet bei Winterthur – Rauchgasreinigungsrückstände. Fallstudien-Bericht der Gruppe 6, Abteilung für Umweltnaturwissenschaften, ETH Zürich, Switzerland.

ETH 1992 ETH (1992): Reaktordeponie Riet bei Winterthur – Rauchgasreinigungsrückstände. Fallstudien-Bericht der Gruppe 6, Abteilung für Umweltnaturwissenschaften, ETH Zürich, Switzerland.

Faulstich 1993 Faulstich M. (1993): Schmelzen von Rückständen aus der Müllverbrennung - Integrieren oder Nachschalten? In: Thomé-Kozmiensky, K.J. (Hrsg.): Reaktoren zur thermischen Abfallbehandlung. EF-Verlag, 175 - 188. Cited in Frühwirth et al. 2000:147

Frühwirth et al. 1993 Frühwirth W., Wöginger H., Döberl G., Huber R., Eder M., Pierrard R., Hutterer H., Stark W., Baumeler A., Rechberger H., Fassbender S., Prinz C. (2000): Bewertung abfallwirtschaftlicher Maßnahmen mit dem Ziel der nachsorgefreien Deponie (BEWEND) Endbericht Version 3. Umweltbundesamt GmbH Vienna

Ganguin 2012 Ganguin J (2012) Kupfer im Sickerwasser einer Schlackedeponie. In: KVA-Rückstände in der Schweiz - Der Rohstoff mit Mehrwert. 2010, Bundesamt für Umwelt (BAFU), Bern. http://www.bafu.admin.ch/publikationen/publikation/01538/index.html?lang=de&lang=de (27 Feb 2012)

Goetz 1989 Goetz D. & W. Gerwinski (1989): Beurteilung der Umweltverträglichkeit von Müllverbrennungsschlacken im Strassenbau. - VGB Kraftwerkstechnik, 69: 504-508; Aachen, Germany.

Gutmann & Vonmont 1994 Gutmann E, Vonmont H (1994) Elektrofilterasche aus Müllverbrennungsanlagen - Quantitative Zusammensetzung. Umweltwissenschaften und Schadstoff-Forschung -Z. Umweltchem. Ökotox. 6 (5) 257-263 (1994) http://www.hs-augsburg.de/~bette/Master%201Semester/Energieverfahrenstechnik/Spiegel/Ebooks/MVA2.pdf (1 Mar 2012)

Hermanns & Moser 2012 Hermanns Stengele R, Moser R (2012) Anforderungen an die Einleitung von Deponiesickerwasser. Empfehlungen für die Beurteilung, Behandlung und Einleitung von Deponiesickerwasser. Bundesamt für Umwelt, Bern. Umwelt-Vollzug Nr. 1223. Mit dem Verband Schweizer Abwasser- und Gewässerschutzfachleute VSA sowie dem Verband der Betreiber Schweizerischer Abfallverwertungsanlagen VBSA. https://www.bafu.admin.ch/dam/bafu/de/dokumente/wasser/uv-umwelt-vollzug/anforderungen_andieeinleitungvondeponiesickerwasser.pdf.download.pdf/anforderungen_andieeinleitungvondeponiesickerwasser.pdf (17 Nov 2016), s auch https://www.bafu.admin.ch/bafu/de/home/themen/wasser/publikationen-studien/publikationen-wasser/anforderungen-einleitung-deponiesickerwasser.html

Hermanns & Moser 2012 Hermanns Stengele R, Moser R (2012) Anforderungen an die Einleitung von Deponiesickerwasser. Empfehlungen für die Beurteilung, Behandlung und Einleitung von Deponiesickerwasser. Bundesamt für Umwelt, Bern. Umwelt-Vollzug Nr. 1223. Mit dem Verband Schweizer Abwasser- und Gewässerschutzfachleute VSA sowie dem Verband der Betreiber Schweizerischer Abfallverwertungsanlagen VBSA. https://www.bafu.admin.ch/dam/bafu/de/dokumente/wasser/uv-umwelt-vollzug/anforderungen_andieeinleitungvondeponiesickerwasser.pdf.download.pdf/anforderungen_andieeinleitungvondeponiesickerwasser.pdf (17 Nov 2016), s auch https://www.bafu.admin.ch/bafu/de/home/themen/wasser/publikationen-studien/publikationen-wasser/anforderungen-einleitung-deponiesickerwasser.html

Hjelmar et al 2013	Hjelmar O, van der Sloot HA, Van Zomeren A (2013) Hazard property classification of high temperature waste materials. Proceedings Sardinia 2013, CISA Publishers, Italy; Fourteenth International Waste Management and Landfill Symposium S. Margherita di Pula, Cagliari, Italy, 30 September - 4 October 2013. https://www.dakofa.dk/fileadmin/user_upload/documents/Netvaerk/Farligt_Affald/131023/Hjelmar_et_al__2013_paper_on_classification_Sardinia.pdf (24 Aug 2022)
Huber et al 2019	Huber F, Blasenbauer D, Aschenbrenner P, Fellner J (2019) Chemical composition and leachability of differently sized material fractions of municipal solid waste incineration bottom ash. Waste Management, Volume 95, 15 July 2019, Pages 593-603. https://www.researchgate.net/profile/Florian-Huber-4/publication/334271961_Chemical_composition_and_leachability_of_differently_sized_material_fractions_of_municipal_solid_waste_incineration_bottom_ash/links/5d22cd0c299bf1547ca1b6bc/Chemical-composition-and-leachability-of-differently-sized-material-fractions-of-municipal-solid-waste-incineration-bottom-ash.pdf (23 Aug 2022)
Huber et al. 1996	Huber H., Jaros M., Lechner P . (1996): Emissionsverhalten von MVA-Schlacke nach künstlicher Alterung. Endbericht, Magistratsabteilung Umweltschutz der Stadt Wien, Österreich, Dezember 1996. http://www.magwien.gv.at/ma22/pool/doc/alterung_schlacke.pdf resp. http://www.wien.gv.at/ma22/pool/abfall.htm (24 Feb 2001)
Hyks 2008	Hyks J (2008) Leaching from Municipal Solid Waste Incineration Residues. Ph.D. Thesis, February 2008, Technical University of Denmark, Department of Environmental Engineering. http://www2.er.dtu.dk/publications/fulltext/2008/ENV2008-033.pdf (1 Mar 2012)
Ioannidou 2022	Personal Communication with Dimitra Ioannidou, construction sector project manager, Ecoinvent Association, Zürich on April 13th 2022
Johnson & Huter 2012	Johnson CA , Huter C (2012) Characterization and geochemical properties of selected incineration residues. In: KVA-Rückstände in der Schweiz - Der Rohstoff mit Mehrwert. 2010, Bundesamt für Umwelt (BAFU), Bern. http://www.bafu.admin.ch/publikationen/publikation/01538/index.html?lang=de&lang=de (27 Feb 2012)
Johnson et al. 1996	Johnson A. C., Kersten M., Ziegler F, Morr H.C. (1996) Leaching behaviour and solubility-controlling solid phases of heavy metals in municipal solid waste incinerator ash. Waste Management, Vol. 16, pp. 129–134. [hard copy obtained by A. Johnson in 2002]
Johnson et al. 1999	C.A Johnson, M. Kaeppeli, S. Brandenberger, A. Ulrich and W. Baumann (1999) Hydrological and geochemical factors affecting leachate composition in municipal solid waste incinerator bottom ash. Part II: The geochemistry of leachate from Landfill Lostorf, Switzerland. J. Contam. Hydrol. 40(3), 239-259.
Jutz & Schlumberger 2011	Jutz M., Schlumberger S. (2011) Wertstoffrückgewinnung aus den Rückständen der Müllverbrennung und Reduktion der Abwasserbelastung - Erfolgreiche Forschungskooperationen, ein Beispiel aus der Praxis. GreentechEvent 2011 Ressourceneffiziente Wassertechnologien 20.Okt 2011, i-net Basel. http://www.inet-innovation.ch/Fileadmin/documents/inet/events/cleantech/111020-06_Jutz-SAM-Medox.pdf (28 Feb 2014)
Kahle et al 2016	Kahle K, Kamuk B, Kallesøe J, Fleck E, Lamers F, Jacobsson L, Sahlén J (2016) Bottom ash from WtE plants - Metal recovery and utilization. ISWA Report. International Solid Waste Association, Austria.
Karlfeldt Fedje 2010	Karlfeldt Fedje K (2010) Metals in MSWI fly ash - problems or opportunities?. Dissertation of Karin Karlfeldt Fedje. Department of Chemical and Biological Engineering, Division of Environmental Inorganic Chemistry, Chalmers University of Technology, Göteborg, Sweden. 2010. Serial nr 3067. ISBN 978-91-7385-386-6 http://publications.lib.chalmers.se/records/fulltext/120763.pdf (1 Mar 2012)
Karpov et al. 2004	Karpov S, Boutoussov M, Hermann L (2004) Inertisierung von Flug- und Filteraschen der Abfallverbrennung mit dem Pellet-Verfahren. In: Thome-Kozmiensky KJ (Ed.): Optimierung der Abfallverbrennung 1. Neuruppin: TK Verlag. Karl Thomé-Kozmiensky, 2004, S. 650. Cited in Thome-Kozmiensky 2013b, p.193
Kersten et al. 1998	Kersten M., Schulz-Dobrick B., Lichtensteiger Th., Johnson A. C. (1998) Speciation of Cr in Leachates of a MSWI Bottom Ash Landfill. Environ. Sci. & Technol., Vol. 32, No. 10, p. 1398-1403.
Klein 2002	Klein R (2002) Wasser-, Stoff- und Energiebilanz von Deponien aus Müllverbrennungsschlacken. Dissertation Technische Universität München, Deutschland. http://bwplus.fzk.de/berichte/ZBer/98/ZBerPD96183.pdf (4 Mar 2002)
Kraxner et al. 2001	Kraxner M., Kochberger M., Lukas W., Lechner P., Mostbauer P., Rautner T., Sabbas T., Saxer A., Wruss W. (2001): Evaluierung von Testmethoden und Modellen für die Bewertung des mittel- bis langfristigen Emissionsverhaltens schadstoffhältiger anorganischer Abfälle und Immobilisierungsprodukte (Projekt IMMO). Final report of the joint project by Technical University Vienna, University Innsbruck and Universität für Bodenkultur (BOKU) Vienna. Published by BOKU; Abteilung Abfallwirtschaft, Austria. http://www.wien.gv.at/ma22/pool/abfall.htm (23 JUl 2002)
Lam et al. 2010	Lam CHK, Ip AWM, Barford JP , McKay G (2010) Use of Incineration MSW Ash: A Review. Sustainability 2010, 2, 1943-1968; https://www.researchgate.net/publication/45267239_Use_of_incineration_MSW_ash_A_review (21 Oct 2016)
Lechner 2001	Lechner P . (2001): Deponietechnik und Altlastensanierung, V orlesungsunterlagen, Universität für Bodenkultur Wien; Abteilung Abfallwirtschaft, Austria. http://www.boku.ac.at/abf/520002/inhalt01.htm and http://www.boku.ac.at/abf/h526a6/SkriptumETGL.htm (11 Jan 2002)
Lechner et al. 2010	Lechner P, Mostbauer P, Böhm K (2010) Grundlagen für die Verwertung von MV-Rostasche - Teil A: Entwicklung des Österreichischen Behandlungsgrundsatzes. Im Auftrag des Bundesministeriums für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, Sektion Stoffstromwirtschaft, Umwelttechnik und Abfallmanagement (Sektion VI). Juni 2010
Leuchs 1990	Leuchs W. (1990): Untersuchung von Abfällen und Rückständen zu Beurteilung bei der Ablagerung und Verwertung. In: Abfallwirtschaft in Forschung und Praxis. Volume 36, Cited in BAK 1995

Lin et al. 2015	Lin WY, Heng KS, Sun X, Wang JY (2015) Accelerated carbonation of different size fractions of MSW IBA and the effect on leaching Waste management 41 7584 https://dr.ntu.edu.sg/bitstream/handle/10220/34458/WM-15-26R1_Revised%20clean.pdf?sequence=1 (19 Sept 2017)
Lindsay 1979	Lindsay WL(1979) Chemical Equilibria in Soils, J. Wiley & Sons, New York, 1979. Cited in Chandler et al. 1997:Tab 9.19
Linsmeyer et al 2009	Linsmeyer T, Dini E, Gruber G, Joos M, Konig F, Lust E, Musil K, Scherndl, H, Schoffmann J, Wengler T (2009) Composition and chemical properties of slag from MSW combustion in a grate fired incinerator and coarse bottom ashes from RDF combustion in a fluidized bed incinerator. 3rd BOKU Waste Conference, April 15-17.2009. Editor: Lechner P., Facultas, Vienna. Cited in Lechner et al. 2010
Löschau 2006	Löschau M (2006) Input-Output-Analyse als Methode zur stofflichen Bilanzierung komplexer Entsorgungssysteme. Dissertation D83, Fakultät III - Prozesswissenschaften der Technischen Universität Berlin, Germany. http://opus.kobv.de/tuberlin/volltexte/2006/1321/pdf/loeschau_margit.pdf (9 Jan 2013)
Ludwig & Johnson 1999	Ludwig C., Johnson C.A. (1999) Weathering of Cemented MSWI Filter Ashes in a Landfill. Poster presented at the R'99 World Congress on Integrated Resources Management, February 2 – 5, 1999 Geneva, Switzerland. http://people.web.psi.ch/ludwig/r99teuftal.pdf (11 Aug 2002)
McLean & Bledsoe 1992	McLean JE, Bledsoe BE (1992) Ground Water Issue Behavior of Metals in Soils. US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Office of Research and Development, Nr. EPA/540/S-92/018. http://www.earthwardconsulting.com/library/EPA_arsenic_in_soil_issue14.pdf (10 Oct 2002)
Mehr et al. 2021	Mehr J, Haupt M, Skutan S, Morf L, Raka Adrianto L, Weibel G, Hellweg S (2021) The environmental performance of enhanced metal recovery from dry municipal solid waste incineration bottom ash. Waste Management 119 (2021) 330–341. https://papers.sim2.be/assets/uploads/files/49262-1-s2.0-s0956053x20305110-main.pdf (19 Aug 2022)
Mocker et al. 2013	Mocker M, Stenzel F, Franke M (2013) Potenzial der Metalle in Stäuben. In: (Thome-Kozmiensky 2013a)
Morf & Kuhn 2009	Morf L, Kuhn E (2009) Qualitätsentwicklung konventionell ausgetragener Schlacke. Baudirektion des Kantons Zürich AWEL Amt für Abfall, Wasser, Energie und Luft Abteilung Abfallwirtschaft und Betriebe (AW) http://www.awel.zh.ch/internet/baudirektion/awel/de/abfall_rohstoffe_altlasten/veroeffentlichungen/_jcr_content/contentPar/publication_2/publicationitems/schlacke_qualit_tsen/download.spooler.download.1320054462148.pdf/qualitaetsentwicklung_austrag_schlacke_2009.pdf (12 Feb 2012)
Morf 2006	Morf L. (2006) Chemische Zusammensetzung verbrannter Siedlungsabfälle Untersuchungen im Einzugsgebiet der KVA Thurgau. Report Nr. UW-620. Bundesamt für Umwelt, Bern. http://www.bafu.admin.ch/publikationen/publikation/00019/index.html (20. Sept 2008)
Morf 2010	Morf LS (2010) Stoffflussanalysen für die thermische Abfallbehandlung: Erkenntnisse und Ausblick nach mehr als 25 Jahren Erfahrung. Baudirektion des Kanton Zürich, Amt für Abfall, Wasser, Energie und Luft (AWEL), Sektion Abfallwirtschaft, Teilbereich Thermische Abfallbehandlung. http://www.bafu.admin.ch/publikationen/publikation/01538/index.html (12 Feb 2012)
Morf et al. 2010	Morf LS, Kuhn E, Sieber C, Balsiger C (2010) Qualität von Sickerwasser aus Zürcher Schlackekompartimenten. AWEL Amt für Abfall, Wasser, Energie und Luft, Abteilung Abfallwirtschaft und Betriebe, Baudirektion des Kantons Zürich, Zürich, 19. Mar 2010. http://www.awel.zh.ch/internet/baudirektion/awel/de/abfall_rohstoffe_altlasten/veroeffentlichungen/_jcr_content/contentPar/publication_2/publicationitems/sickerwasser_qualit_/download.spooler.download.1320054224761.pdf/sickerwasser_schlackenkompartimente_bericht_2010.pdf (23 Nov 2012)
Morf et al. 2013	Morf LS, Gloor R, Haag O, Haupt M, Skutan S, Di Lorenzo F, Böni D, (2013) Precious metals and rare earth elements in municipal solid waste - Sources and fate in a Swiss incineration plant. Waste Manag. 33, 634–44. doi:10.1016/j.wasman.2012.09.010 Obtained from Leo Morf on 29 Nov 2016
Piantone et al. 2008	Piantone P, Autret E, Bodenan F, Blanc P (Red.); Mâchefers d'incinération des ordures ménagères – État de l'art et perspectives, ADEME & BRGM Éditions, Dunod, Paris 2008. Cited in Rylander & Wiqvist 2011:Tab.12
Plüss 1993	Plüss A (1993): Charakterisierung von Rauchgasreinigungsrückständen aus Müllverbrennungsanlagen und deren Immobilisierung mit Tonmineralien. Verlag der Fachvereine vdf, Zürich, Switzerland.
Raaflaub & Genoni (2005)	Raaflaub M, Genoni M (2005) Preise für landwirtschaftliche Produktionsmittel in der Schweiz und in den EU-Nachbarländern Deutschland und Frankreich. Schlussbericht, August 2005. Schweizerische Hochschule für Landwirtschaft SHL, Zollikofen. Studie im Auftrag des Bundesamtes für Landwirtschaft BLW. https://www.blw.admin.ch/dam/blw/de/dokumente/Nachhaltige%20Produktion/Produktionsmittel/Preise%20f%C3%BCr%20landwirtschaftliche%20Produktionsmittel.pdf.download.pdf/Preise%20f%C3%BCr%20landwirtschaftliche%20Produktionsmittel.pdf (29 Oct 2022)
Reichelt & Pfrang 1998	Reichelt J, Pfrang G (1998) Qualitätsverbesserung von Müllverbrennungsschlacken. PWAB-Projects PD 96 183 and PD 96 184, University Karlsruhe, Germany. http://bwplus.fzk.de/berichte/ZBer/98/ZBerPD96183.pdf (4 Mar 2002)
Reimann 1989	Reimann DO (1989): Müllverbrennungsschlacke - Inhaltsstoffe, Mengen, Verwertbarkeit. UWSF Z, Umweltchemie 2, p. 18 – 25, ecomed-Verlag
Reuter & Schirmer 1988	unreferenced source in Zimmermann et al. 1996
Rey 1992	unreferenced source in Zimmermann et al. 1996:B.116 (MSWI bottom ash composition)
Rohrer 2006	Rohrer Z (2006) Gebührenvergleich für Wasser, Abwasser und Abfall für die 30 grössten Städte der Schweiz. Wirtschaftsdepartement, Preisüberwachung, Schweizerische Eidgenossenschaft, Oktober 2006. https://www.preisueberwacher.admin.ch/dam/pue/de/dokumente/studien/gebuehrenvergleichwasserabwasserabfall.pdf.download.pdf/gebuehrenvergleichwasserabwasserabfall.pdf (23 Feb 2020)

Rylander & Wiqvist 2011 — Rylander H, Wiqvist W (2011) Förbättring av bottenaskors kvalitet - Rapport U2011:17. Avfall Sverige Utveckling, Malmo, Sweden. ISSN 1103-4092. http://www.avfallsverige.se/fileadmin/uploads/Rapporter/Utveckling/Rapporter_2011/U2011-17.pdf (14 Nov 2012)

Sabbas et al. 1998 — Sabbas T., Mostbauer P., Lechner P. (1998): Deponien – Prozesse und Faktoren jenseits der Nachsorge, Magistratsabteilung Umweltschutz der Stadt Wien, Österreich. http://www.magwien.gv.at/ma22/pool/doc/deponien.pdf via http://www.wien.gv.at/ma22/pool/abfall.htm (28 Dec 2001)

Sales Bandarra et al. 2021 — Sales Bandarra B, Pereira JL, Martins R, Maldonado A, Chimenos JM, Quina M (2021) Incineration Bottom Ash from Municipal Solid Waste. Encyclopedia, MDPI (Multidisciplinary Digital Publishing Institute), Basel, Switzerland. Version of 24 Dec 2021. https://encyclopedia.pub/entry/17538 (24 Aug 2022)

Schachermayer et al. 1994 — Schachermayer E., Bauer G., Ritter E., Brunner P. (1994): Messung der Güter und Stoffbilanz einer Müllverbrennungsanlage, Institut für Wassergüte und Abfallwirtschaft, Abteilung für Abfallwirtschaft, Technical University Vienna. Cited in Huber et al. 1996:28.

Schlumberger & Bühler 2013 — Schlumberger & Bühler 2013 Metallrückgewinnung aus Filterstäuben der thermischen Abfallbehandlung nach dem FLUREC-Verfahren. In: (Thome-Kozmiensky 2013) http://www.vivis.de/phocadownload/2013_ass/2013_ass_377_398_schlumberger.pdf (13 Jan 2015)

Schlumberger 2011 — Schlumberger S. (2011) Wertstoffgewinnung durch saure Wäsche von KVA- Filterasche. Stand der Technik in der Abfall- und Ressourcenwirtschaft Tagung im Rahmen der Umwelt 11, 9. September 2011. Kanton Zürich, Baudirektion, Amt für Abfall, Wasser, Energie und Luft (AWEL) http://www.awel.zh.ch/internet/baudirektion/awel/de/betriebe_anlagen_baustellen/abfallanlagen/stand_der_technik/_jcr_content/contentPar/morethemes/morethemesitems/904_1315901058306.spooler.download.1320054848912.pdf/Vortraege_Umwelt11_Nachmittag.pdf (27 Feb 2014)

Schweizer 1999:14 — Schweizer C.R. (1999) Calciumsilikathydrat-Mineralien – Lösungskinetik und ihr Einfluss auf das Auswaschverhalten von Substanzen aus einer Ablagerung mit Rückständen aus Müllverbrennungsanlagen. Dissertation No. 13074, ETH Zürich, Switzerland. https://www.research-collection.ethz.ch/bitstream/handle/20.500.11850/144133/eth-22998-02.pdf (7 Feb 2020)

Simon & Holm 2013 — Simon FG, Holm O (2013) Aufschluss, Trennung und Rückgewinnung von Metallen aus Rückständen thermischer Prozesse – Verdoppelung der Metallausbeute aus MVA-Rostasche. In: (Thome-Kozmiensky 2013a)

Sivula 2012 — Sivula L (2012) Characterisation and Treatment of Waste Incineration Bottom Ash and Leachate. Dissertation Nr 235. Jyväskylä Studies In Biological And Environmental Science, Department of Biological and Environmental Science, University of Jyväskyla, Finnland. ISBN 978-951-39-4658-6 https://jyx.jyu.fi/dspace/bitstream/handle/123456789/37356/9789513946586.pdf?sequence=1 (1 Mar 2012)

Skutan et al. 2014 — Skutan S, Gloor R, Morf L (2014) Methodenband - Probenahme, Probenaufbereitung und Analyse fester Rückstände der thermischen Abfallbehandlung und deren Aufbereitungsprodukten. März 2014. Stiftung Zentrum für nachhaltige Abfall- und Ressourcennutzung ZAR, Hinwil, Switzerland. https://zar-ch.ch/fileadmin/user_upload/Contentdokumente/Oeffentliche_Dokumente/Methodenband_DE.pdf (14 Nov 2016)

Speiser et al. 2002 — Speiser C., Heuss-Assbichler S., Spiegel W. (2002): Significance of Ettringite for Long-Term Stability of Boiler Ashes and Gas Cleaning Residues of MSWI. Presentation at R'02 World Congress on Integrated Resources Management, February 12 – 15, 2002 Geneva, Switzerland

Stark 1993 — Stark R. (1993): Verglasung von Rauchgasreinigungsrückständen - Systeme, Ziele, Stand und Perspektiven. In: Anonymus (1993): Möglichkeiten der thermischen Abfallbehandlung und -verwertung. Stuttgarter Berichte zur Abfallwirtschaft, Band 55, 1993

Sun et al. 2008 — Sun J, Fernández Bertos M, Simons SJR (2008) Kinetic study of accelerated carbonation of municipal solid waste incinerator air pollution control residues for sequestration of flue gas CO2. Energy Environ. Sci., 2008, 1, 370-377. https://pubs.rsc.org/en/content/articlehtml/2008/ee/b804165m (25 Aug 2022)

Svensson 2006 — Svensson 2006 Svensson M. (2006) Controlling the Mobility of Organic Carbon (OC) and its Impact on Metal Transport from Incineration Residues. Lulea University of Technology Department of Civil and Environmental Engineering, Division of Waste Science and Technology. Licentiate thesis 06/32. http://epubl.ltu.se/1402-1757/2006/32/LTU-LIC-0632-SE.pdf (1 Mar 2012)

Syc et al. 2010 — Syc M, Keppert M, Pohorely M, Novák P, Puncochár M, Fiserová E, Pekárek V (2010) Fly Ash Treatment Technology in Modern Waste Incineration Plant. Proceedings of the Second International Conference on Sustainable Construction Materials and Technologies , June 28 - June 30, 2010, Università Politecnica delle Marche, Ancona, Italy. http://www.claisse.info/2010%20papers/l41.pdf (13 Jan 2015)

Symeonidis 2021 — Personal communication with Avraam Symeonidis, Junior Project Manager, Ecoinvent Association, Zürich on Oct 22nd 2021, "Re: SWM and WW sector updates for 3.9"

Symeonidis 2022 — Avraam Symeonidis (2022) Waste and wastewater product information – Design and discussion. File name "Waste product information DRAFT v3 20220311.docx", of March 11th 2022, Ecoinvent Association, Zurich, Switzerland.

Taverna 2011 — Taverna R (2011) Routinemässiges Stoffflussmonitoring auf der MVA Spittelau Messperiode 1.5.09–30.4.10 – Endbericht. Im Auftrag der Magistratsabteilungen 22 und 48 der Stadt Wien, Fernwärme Wien GmbH. GEO Partner AG http://www.wien.gv.at/umweltschutz/pool/pdf/mape.pdf (4 Oct 2012)

Thome-Kozmiensky 2013a — Thome-Kozmiensky KJ (2013a) Aschen - Schlacken - Stäube. VIVIS Verlag. ISBN 978-3-935317-99-3 http://www.vivis.de/index.php?option=com_content&view=article&id=260:2013-09-26-12-26-36&catid=54:recycling-und-rohstoffe&Itemid=76 (23 Jan 2015)

Thome-Kozmiensky 2013b — Thome-Kozmiensky KJ (2013b) Möglichkeiten und Grenzen der Verwertung von Sekundärabfällen aus der Abfallverbrennung. In: (Thome-Kozmiensky 2013a)

Todorovic 2010	Todorovic J; Siktning av askor från avfallsförbränning, Waste Refinery Excellence Center, Borås 2010, rapport WR-25, www.wasterefinery.se. Cited in Rylander & Wiqvist 2011:Tab.12
VVEA 2016	Schweizerische Bundeskanzlei (2016) Verordnung über die Vermeidung und die Entsorgung von Abfällen (Abfallverordnung, VVEA). Systematische Sammlung des Bundesrechts (SR) 814.600. Vom 4. Dezember 2015 (Stand am 1. Januar 2016). https://www.admin.ch/opc/de/classified-compilation/20141858/index.html (17 Feb 2016)
Weibel 2020	Weibel G (2020) Deponie Chrüzlen - Monitoring und Emissionsprognose des Trockenschlackenkompartiments. Schriftenreihe der Stiftung ZAR, Nr. 002, April 2020. Stiftung Zentrum für nachhaltige Abfall- und Ressourcennutzung ZAR, Hinwil, Switzerland. https://zar-ch.ch/fileadmin/user_upload/Contentdokumente/Oeffentliche_Dokumente/1.02_Schriftenreihe_002_Deponiemonitoring_Chruezlen_2020_01.pdf via https://zar-ch.ch/zar/publikationen/veroeffentlichungen/ (20 Aug 2022)
Winter et al 2009	Winter B, Szednyj I, Reisinger H, Böhmer S, Jahnsen T (2005) Abfallvermeidung und -verwertung: Aschen, Schlacken und Stäube in Österreich. UBA Report REP-0003, Umweltbundesamt Wien. Cited in Lechner et al. 2010
Zeltner & Lichtensteiger 2002	Zeltner C, Lichtensteiger T (2002) Thermal waste treatment and resource management – a petrologic approach to control the genesis of materials in smelting processes. Environmental Engineering Policy, No. 3, p.75-86.
Zimmermann et al. 1996	Zimmermann P., Doka G., Huber F., Labhardt A., Menard M. (1996): Ökoinventare von Entsorgungsprozessen, Grundlagen zur Integration der Entsorgung in Ökobilanzen. ESU-Reihe, 1/96, Zürich: Institut für Energietechnik, ETH Zurich

Lehmann & Hischier 2007-III	Lehmann M, Hischier R (2007) Life cycle inventories of Electric and Electronic Equipment: Production, Use and Disposal – Part III: Electronic Devices. ecoinvent report No. 18. Empa / Technology & Society Lab, Swiss Centre for Life Cycle Inventories, Dübendorf, 2007 http://db.ecoinvent.org/ecoquery/files/18_I_Electronics_components.pdf?area=463ee7e58cbf8 (7 Feb 2011)
Doka 2009-I	Doka G (2009) Life Cycle Inventories of Waste Treatment Services. ecoinvent report No. 13. Ecoinvent v2.1, Swiss Centre for Life Cycle Inventories, St. Gallen
Lim & Chu 2006	Lim T-T & Chu J (2006). Assessment of the use of spent copper slag for land reclamation. Waste Management & Research, 24(1), 67–73. https://doi.org/10.1177/0734242X06061769 (19 Apr 2018)
Rustandi et al. 2018	Rustandi A, Cahyadi A, Siallagan S, Nawawi F, Pratesa Y (2018). Corrosion Behavior of Carbon Steel in Concrete Material Composed of Tin Slag Waste in Aqueous Chloride Solution. IOP Conference Series: Materials Science and Engineering. 299. 012047. https://doi.org/10.1088/1757-899X/299/1/012047
Jamil Hashim et al. 2018	Jamil Hashim M, Mansor I, Pauzi Ismail M, Sani S, Azmi A, Sayuti S, Zaidi M, Adli A, Adha Abdul Rahim A, (2018). Preliminary study of tin slag concrete mixture. IOP Conference Series: Materials Science and Engineering. 298. 012014. https://doi.org/10.1088/1757-899X/298/1/012014
MK 1992	Morrison Knudsen Corporation (1992) Report for Zinc Slag Pile Remedial Investigation at the California Gulch Site Leadville, Colorado. Prepared by Morrison Knudsen Corporation (MK), Environmental Services Division. https://semspub.epa.gov/work/08/318849.pdf (20 Apr 2018)
BAFU 2021	BAFU (Ed.) (2021) Ökofaktoren Schweiz 2021 gemäss der Methode der ökologischen Knappheit. Methodische Grundlagen und Anwendung auf die Schweiz. Bundesamt für Umwelt, Berne. Umwelt-Wissen Nr. 2121. https://www.bafu.admin.ch/bafu/de/home/themen/wirtschaft-konsum/publikationen-studien/publikationen/oekofaktoren-schweiz.html (13 Apr 2022)
Dones & Zollinger 1994	Dones R, Zollinger E (1994) Teil VII – Kernenergie. In: Frischknecht R, Hofstetter P, Knoepfel I, Menard M, Dones R, Zollinger E (eds.) Ökoinventare für Energiesysteme – Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. 1st Edition, March 1994, ETH Zürich, and Paul Scherrer Institut, Villigen/Würenlingen, Switzerland.
Doka 2023-7	Doka G. (2023) Calculation manual for LCI calculation tools for regionalised waste treatment. Extended and updated version of May 2023. By Doka Life Cycle Assessments, Zurich, Switzerland. Available at http://www.doka.ch/publications.htm

Kornievskiy et al. 2015	Kornievskiy VN, Panchenko AI, Logozinskiy IN, Salnikov AS, Gasik MI, Shibeko PA, Budnik IL, Yakovitskiy AV (2015) Ferromolybdenum production according to silicoaluminthermal method: High quality, chemical and granulometric composition stability. Proceeding of the 14th International Ferroalloys Congress- May 31-June 4, 2015 Kiev, Ukraine. INFACON 2015, 2, pp. 139–144
Chen et al. 2018	Chen WA, Liang WC, Lee CH (2018) Study of Molybdenum Extraction from Alkali Roasted and Water Leaching of Ferro-Molybdenum Slag by Using TOA and TBP. 9th International Conference on Environmental Science and Technology 2018, IOP Conference Series: Earth and Environmental Science, 182(1), pp. 0–8. doi: 10.1088/1755-1315/182/1/012015.
Boehme & Van Den Hende 2011	Boehme L, Van Den Hende D (2011) Ferromolybdenum slag as valuable resource material for the production of concrete blocks. 2nd International Slag Valorisation Symposium, Leuven, Belgium 18-20 April 2011. https://www.researchgate.net/publication/228485096_Ferromolybdenum_slag_as_valuable_resource_material_for_the_production_of_concrete_blocks (20 Nov 2020)
Guo et al. 2014	Guo X, Wang K, He M, Liu Z, Yang H, Li S (2014) Antimony smelting process generating solid wastes and dust: Characterization and leaching behaviors- Journal of Environmental Sciences (China). Elsevier B.V., 26(7), pp. 1549–1556. doi: 10.1016/j.jes.2014.05.022.