Two models for waste-specific and climate-specific life cycle inventories of excavation material landfills and construction & demolition waste landfills

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Corrections Compared to the first version of 2020 of this report several typos were corrected.

Title image Waste concrete rubble deposit in open air. CC0 Public Domain from https://www.publicdomainpictures.net/en/view-image.php?image=85674

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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.

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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 "$3.3 \cdot 10^{-1}$"). 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 (see e.g. Eq. 3.3). "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.

For the life cycle database of ETH Zurich/ecoinvent modelling tools have been created that allowed to inventory the waste-specific burdens occurring during waste disposal. The first such tools were created in 1996 (Zimmermann et al. 1996), which were subsequently used in the influential Swiss EPA packaging studies (BUWAL 1996) and the seminal ecoinvent database (Doka 2003). For landfills, that involved to define the composition of the landfilled waste materials and establish the behaviour in the landfills leading to emissions into the environment. To reduce work effort it was approximated in ecoinvent that landfilled excavation material and inorganic building material waste would not be leading to any relevant direct emissions based on the low pollutant content of the landfilled material (see e.g. Frischknecht et al. 1996:F.27). Only land use and process energy was inventoried. I.e. unlike with burdens from other landfills like sanitary landfills, residual material landfills etc. for inert material landfills no direct emissions from the landfilled waste materials were inventoried.

Already in 2003 (Doka 2003-III:74) it was remarked that the simplification mentioned above was inconsistent with the rest of the database: LCA is used to establish burden data of processes and any data gap in this endeavour is potentially problematic. Instead of merely *assuming* the emissions in excavation and construction waste landfills were insignificant, it would be much more defendable to *show* that they indeed were relatively minor – with proper, quantified inventory results.

The work presented here establishes landfill models for excavation and construction waste materials which are conceptually compatible with the models already established for a wide range of different landfilling activities in ecoinvent (Doka 2003-III, Doka 2008a, Doka 2008b, Doka 2009, Doka 2017).

Thanks to Daniel Kellenberger of Intep (Zurich, Switzerland) for information regarding the inclusion of carbonation in building material assessment (chapter 3.7). Thanks to Andreas Ciroth of GreenDelta (Berlin, Germany) for his testing of tool-generated EcoSpold1 files in OpenLCA. Thanks to Mischa Zschokke of carbotech (Zurich, Switzerland) for his testing of tool-generated EcoSpold1 files in SimaPro software.

1.1 Rationale and motivation

Assuming a supposed irrelevance of emissions from inert material waste landfills in an LCA framework might be wrong for two reasons:

1. **Infrequent deviations (non-averageness):** It might be true that inert material waste landfills as a whole are relatively small pollutant sources. But LCA can also take a view along *product lifecycles* with their specific material uses. A specific material might cause above average burdens, which could be well relevant in a product view. The same material might not be noticeable in a view looking only at the average landfill as a whole with its average mixture of waste materials.
2. **Landfilled masses:** Inert wastes, especially excavation material, make up a large amount of the annually generated wastes in Switzerland. It might be true that for example *one kilogram* of concrete waste is much less burdening than one kilogram of incinerator bottom ash. But the annually generated mass of concrete waste is *much larger* than the mass bottom ash, and these relations will tend to be reproduced when inventorying processes and products. It is therefore possible that a specific small burden of inert waste is compensated by their comparative much larger magnitude in sheer mass.

One might argue that waste like excavated material is taken from nature and that its disposal is merely a restoration of what is taken from nature, and that no emissions should be attributed to that. Within the realm of LCA several arguments against such a neglect can be made:

A. The material has been transferred – however briefly – into the **technosphere** and as a disposed waste, its disposal should be included in LCA.

B. Until now, all landfilled inert wastes were considered emissions-free (in ecoinvent). It is the expressed goal of this project to **quantify landfill emissions** also of inert wastes to *show* their actual relevance in an LCA framework, instead of *a priori assuming* their relative harmlessness.

C. Even natural materials can lead to burdens accountable in LCA. For instance biomass removed from river water before it is turbinated for hydropower need legally to be disposed as waste an cannot be dumped back into the river after the turbine – even though they can be mostly considered natural materials. The disposal of this biomass waste is therefore also part of the process of hydropower production.

D. By breaking up the **grain and structure** of an excavated material that has been not disturbed for possibly millennia the material is made more reactive and prone to larger pollutant emissions than before. So the disposed material is not really identical to the original situation.

E. The goal of the Swiss waste ordinance and many waste management priorities is to maximise reuse and recycling. The **advantage of recycling** can only be judged correctly if the process of non-recycling, like landfilling, is complete and includes all its emissions and burdens. Otherwise the recycling option receives an unfair disadvantage. This should also extend to landfilling of inert materials.

2 Modelling concept

Like with other landfill models in ecoinvent (Doka 2003, 2017) the ultimate goal is to obtain process inventories for the disposal of a *specific waste material*, not merely the landfill's average content. Emissions from the landfill heed the composition of the specific waste under investigation. If a waste contains for instance no cadmium, then no cadmium emission will be inventoried for this waste. Other expenditures like for operation, materials and land use are added in the process inventory, which lead to indirect emissions.

In order to estimate the burdens specific to a certain waste material from landfills, a two-staged approach has been followed for the landfill inventories in the ecoinvent realm (cf. Doka 2003-III).

2.1 Stage one: Working point model

In a first stage a so-called **working point model** of the landfill is created. The aim of this is to describe the typically observed mobility of pollutants in a landfill type. As in LCA it is usually not known at which landfill site exactly a waste is disposed, the model is not intended for modelling *individual* landfills, but rather for depicting the generic, typical pollutant behaviour in a whole class of landfills. The term 'site' or 'location' in connection with the disposal models can therefore refer to an average process in a country, continent or even globally.

For the working point model, information on the average landfill waste composition is collected along with information on typically encountered leachate concentrations in such landfills. Collected are figures on heavy metals and other chemical elements actually measured in the field and published in obtainable literature sources.[1] Working with only chemical elements, which are not degradable, simplifies the model complexity, and usually covers the major causes of emission burdens in an LCA framework. The ratio of observed leachate concentration and average landfilled waste composition gives a realistic indication of the actually observed typical mobility of pollutants in that type of landfill. In this way, various complex and heterogeneous solubility-determining effects are implicitly heeded in an integral manner, without having to identify *how* those effects come about in detail.[2] The working point model can therefore be considered a top-down model, being constructed from observed behaviour in the field, and not bottom-up from a synthetic simulation of complex chemical reactions. With the typical mobility of pollutants established, data on infiltrating rainwater is used to estimate the magnitude of landfill pollutant outputs. Infiltrating water is the result of a combination of the location climate and landfill engineering design. Over time, changes in the landfill body can occur, especially a lowering of pH from loss of pH buffering materials. This changes some of the leachate concentrations and in turn the resulting emissions. The end result of the working point model creation is a set of **transfer coefficients** which define for a given climate how much of a landfilled pollutant will be emitted over time.[3] The transfer coefficients describe the pollutant behaviour in that landfill, based on the boundary conditions of climate and landfill engineering design.

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1 Laboratory column leaching data are excluded as they are usually very short-term and small-volume.

2 Conceptually this is similar to fate calculations in substance-specific LCIA methods, where generic solid-liquid distribution coefficients are used to discern a pollutants distributions and fate in various environmental media. The conceptual granularity is also similar to process inventories, where for instance the typical production energy for a good is derived from the mean of energy consumptions of several factories for that good, without knowing in detail all energy consuming stages and equipments in those productions.

3 For sanitary landfills and similar (open dumps, unsanitary landfills) also the *degradability* of different materials in the landfill is heeded leading to *waste-specific* transfer coefficients. For inorganic landfills (residual material landfills, bottom ash

Eq. 2.1 $$Emission_e = composition_e \cdot TK_e$$

where

Emission $_e$ = waste-specific emission for pollutant e, [kg e]

composition $_e$ = total content of pollutant e in specific waste, [kg e / kg waste]

TK $_e$ = transfer coefficient for pollutant e, between 0 and 1, [-]

2.2 Stage two: Application of waste-specific composition

The second stage of the creation of a waste-specific landfill inventory is the application of the working point transfer coefficients of the previous stage on a *particular and specific* waste material composition. The pollutants initially present in a waste material represent 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.[4] 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 input of waste to a landfill corresponds to its disposal service function of taking up unwanted waste.

The landfill disposal inventory is complemented with the more 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 commonly not waste-specific, but are attributed 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.

3 Working point models for inert material waste landfills

The goal of this project was to establish new landfill models and calculation tools for emissions and exchanges from disposal of inert material waste. In Switzerland two types of inert material landfills are distinguished since 2016:

Type A landfills: receiving mostly clean excavation material

Type B landfills: receiving mostly inorganic construction and demolition waste material and polluted excavation material

The current Swiss Waste Ordinance of 2016 emphasizes reuse and recycling over landfilling, so these landfills will receive materials that could not be reused or recycled, for whatever reason.

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landfills, tailings impoundments, inert material landfills) only one *average set* of transfer coefficients is created per landfill locale and applied to all waste materials alike.

[4] In inert material landfills no leachate treatment is assumed. For sanitary landfills, part of the flows can go first to wastewater treatment or to landfill gas capture systems prior to being finally emitted.

A first goal of this project was to establish whether the type A and type B landfills are so different that they warrant also two different landfill models, or whether the landfills are so similar in behaviour of pollutant emissions that a single model for "inert wastes" is sufficient.

3.1 Literature survey

Data on material composition and leachate concentrations was compiled from available literature sources (Tab. 3.1). Data for the main contents of the investigated landfills was collected, which are mineral construction waste and excavation material. Literature data points published as being below a specified threshold were included as 50% of the threshold.[5] In total 117 different material measurements for 90 different parameters were compiled, which represent roughly 20'000 single measurements.

Tab. 3.1 Literature sources used to derive generic average material composition and leachate concentrations in construction waste landfills and excavation material landfills.

Analysed samples	Used literature data
Mineral construction waste	Burkhardt et al. 2015, Butera et al. 2014, Dagan 2017, Doka 2003-I, EPA 1995, Graf 1998, Grauwiller 1992, Hermanns & Moser 2012, Holcim 2010, Laner 2011, Lechner & Dreier 1995, Pladerer et al. 2004, Pratt 2018, Rubli et al. 2017, Schachermayer et al. 1998, Scheibengraf & Reisinger 2005, Schmutz 2015
Clean excavation material	AWEL 2016a, AWEL 2016b, Gimmi & Waber 2004
Concrete [1]	Butera et al. 2014, Engelsen et al. 2010, Schachermayer et al. 1998, Scheibengraf & Reisinger 2005
Cement [1]	Butera et al. 2014, Doka 2003-I, van der Sloot et al. 2011
Clean soil [2]	Beesley & Dickinson 2011, Bradford et al. 1996, Kolbas et al. 2018, NABO 2017, NABO 2017, Salminen et al. 2005, Shacklette & Boerngen 1984
Groundwater & tap water [2]	AWEL 2018a, Ayotte et al. 2011, Gallert 2000, Kerndorff et al. 1993, Langmuir et al. 2004, NAQUA 2019, Salminen et al. 2005, WVZ 2001
Geogenic crust [2]	Haxel et al. 2002, CRC 1985:F-145, Zimmermann et al. 1996

1 Used for proxy gap filling in mineral construction waste landfill

2 Used for proxy gap filling in excavation material landfill

From the literature data a typical, average, generic composition of landfill contents and the leachate concentrations was derived as the geometric mean of the available data points. Data from actual construction waste landfills and excavation material landfills was used with priority. In order to avoid data gaps some approximations from similar materials are used. Data for concrete and cement is used to fill data gaps in mineral construction waste; data from clean soil, groundwater, tap water, and geogenic crust fills data gaps for excavation material.[6] The data suggests that the composition of clean excavation material and clean (top)soil are very similar, why these approximations to fill data gaps appear permissible.

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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 major results of the model, but helps establishing estimates for transfer coefficients for minor elements like beryllium, silver, selenium, and thallium.

6 If a data-gap-filling proxy was required, the geometric mean over the values from appropriate proxy material samples was taken, which was used as the proxy mean value. No weighting or hierarchy amongst the different proxy sample materials was employed. Thus the proxy value is also based on a *large sample size* from various sources, which decreases the risk of incorporating outlier data into the model.

A large data compilation of 1150 measurements of waste composition in Swiss excavation landfills could be accessed in (AWEL 2016a). The authors emphasize that these samples represent measurements of "suspicious waste" tested for landfill compliance and are not representing average waste. However, analysis shows that while some samples indeed are not compliant, the majority of over 88% is. Non-compliant are chiefly parameters for organic pollutants[7], which suggests suspicions were raised by perceivable discoloration or smell, while the inorganic composition is largely representative. The carbon composition from this source was not used, since it only measured some particular organic compounds, not all organic carbon. Furthermore, the median value of all samples is practically identical to the median of only compliant samples. The latter was used for the literature compilation on clean excavation material, which matches well with the values given from other literature, so no extreme or outlier data was introduced from this source.

[7] Hydrocarbon index (C10 – C40), PAH and BaP.

Tab. 3.2 Typical, average composition of landfilled mineral construction waste and clean excavation material and their respective leachate concentrations. From literature indicated in Tab. 3.1. n.a. = not available.

	Mineral construction waste	Leachate mineral construction waste	Clean excavation material	Leachate clean excavation material
	mg/kg	mg/l	mg/kg	mg/l
Oxygen	425'650	n.a.	484'030	n.a.
Hydrogen	66'667	n.a.	1331.5	n.a.
Organic Carbon	1222.5	11.347	3494.6	6.7668
Sulfur	4072.9	171.07	525.82	411.22
Nitrogen	n.a.	5.9353	64.149	0.7624
Phosphor	440	0.027334	534.43	0.038315
Boron	n.a.	0.23405	6	1.1055
Chlorine	335.38	52.573	120.6	460.16
Bromium	n.a.	0.19522	1.4355	0.4865
Fluorine	505.18	0.63442	414.06	4.333
Iodine	n.a.	n.a.	1.017	0.09552
Silver	n.a.	0.00005	0.16853	0.0002
Arsenic	4.4831	0.0064269	5.5295	0.0072884
Barium	41.801	0.085894	122.33	0.058484
Cadmium	0.18541	0.0011602	0.28536	0.000036354
Cobalt	7.6059	0.00171	7.6059	0.00019073
Chromium	27.477	0.0049423	24.447	0.00059702
Copper	16.336	0.0075925	13.256	0.0047605
Mercury	0.070909	0.00017156	0.094229	0.00026265
Manganese	422.07	0.13657	449.65	0.20351
Molybdenum	1.0764	0.0035227	2	0.0003381
Nickel	15.387	0.018043	20.606	0.00083911
Lead	24.753	0.0071702	10.354	0.00019375
Antimony	0.08	0.0013006	0.975	0.000067159
Selenium	3.3846	0.00074162	0.039872	0.00046201
Tin	2.9384	0.022865	2.5791	n.a.
Vanadium	34.573	0.0037279	74.936	0.00041981
Zinc	50.557	0.042929	39.19	0.0069413
Beryllium	1.0174	0.0005	0.5769	0.000009
Scandium	n.a.	n.a.	9.4681	n.a.
Strontium	259.51	1.1937	166.84	3.8223
Titanium	2182.7	0.041007	3765.5	0.0009
Thallium	0.42096	0.0005	0.43481	0.000005
Tungsten	n.a.	n.a.	1.3551	0.000007
Silicon	136'460	n.a.	301'350	15.495
Iron	9143.8	0.099142	23'378	11.369
Calcium	173'630	167.82	15'201	131.61
Aluminium	6454.4	0.20054	53'617	0.029622
Potassium	14'480	42.616	11'394	19.574
Magnesium	12'000	4.828	5878.2	39.207
Sodium	7989.2	61.854	7749.8	686.22

Tab. 3.3 Further average leachate concentrations in landfilled mineral construction waste and clean excavation material landfills. From literature indicated in Tab. 3.1.

	Leachate mineral construction waste mg/l	Leachate clean excavation material mg/l
DOC dissolved organic carbon	14.74	5.338
BOD biological oxygen demand	6.921	n.a.
COD chemical oxygen demand	50.72	n.a.
Carbonate Carbon (CO_3-C)	51.86	62.38
Sulfide-S	0.0354	0.02902
Sulfate-S (SO_4-S)	179.3	411.1
Nitrate-N (NO_3-N)	3.055	0.5259
Ammonium-N (NH_4-N)	0.472	0.05241
pH	7.868	8.43

3.2 Rainwater infiltration

The leachate carrying pollutants out of the landfill as water emissions consists of rainwater percolating though the landfill. In the landfill model the amount of leachate is calculated based on the difference of precipitation and evapotranspiration, both which are parameters depending on climate and location.

In Switzerland the typical infiltration would be 500 mm (difference between 1000 mm precipitation and 500 mm evapotranspiration). Data for Swiss inert landfills indicate that the leachate volume corresponds to about 300 mm, not 500 mm.[8] For the inert material landfill models the leachate volume calculation is modified to a lower value.

In order to keep the climate-dependency of the model, but also include the observed reduced infiltration in inert waste landfills, the calculated infiltration is reduced by 40% (multiplied with 0.6). With 1000 mm precipitation and 500 mm evapotranspiration, this leads to the 300 mm raw infiltration figure observed in the field. The factor 0.6 is employed in all climates.

As introduced in (Doka 2017) the user can specify a climate for the landfill using mean annual precipitation (MAP), actual evapotranspiration (ETa), and mean annual temperature (MAT). The infiltration rate is then calculated from the difference of precipitation and evapotranspiration.

Eq. 3.1 $$I_p = |MAP - ETa| \cdot 0.6$$

where

I_p Preliminary water infiltration rate (mm/m^{2}a)

MAP Mean Annual Precipitation, user-defined for a location (mm/m^{2}a)

ETa Actual evapotranspiration, user-defined for a location (mm/m^{2}a)

As a further refinement, in (Doka 2018:8) a "soft capping" of the infiltration rate was introduced. In some extreme climates the assumption of zero surface water runoff would lead to unrealistically high infiltration rates. To avoid this, for preliminary infiltration rates larger than 1000 mm/m^{2}a the value

[8] Most recent measurements for inert material landfill in (AWEL 2018b) indicate a mean annual leachate volume of 330 mm (i.e. liters per m^2). This was calculated from data given for total annual pollutant releases per area (t per ha*year) and pollutant concentrations in leachate (in mg per litre). Also (AWEL 2008) indicates an "effluent coefficient" of 0.3 to convert annual precipitation into annual leachate volume. This corresponds to a 300 mm infiltration figure in the landfill model.

was exponentially corrected to values always below 2000 mm/m²a, which is the maximal natural infiltration rate observed in the field (see Doka 2018:Fig. 3.1). The preliminary infiltration rate I_p from the above equation is therefore modified by following formalism:

Eq. 3.2
$$I = \begin{cases} if\ I_p \le 1000mm: & I = I_p \\ if\ I_p > 1000mm: & I = 1000 \cdot \left(2 - e^{\left(-0.001 \cdot \left[I_p - 1000\right]\right)}\right) \end{cases}$$

I	Infiltration rate (mm/m²a), as used in Eq. 3.3
I_p	Preliminary Infiltration rate (mm/m²a), from Eq. 3.1

3.3 Transfer coefficients

The transfer coefficients of the working point model are calculated based on three pieces of information: the initial average landfilled waste content, the initial average leachate concentration (both shown in Tab. 3.2) and the infiltration water generating landfill leachate (see Eq. 3.2).

Leachate generation

The available water from infiltration carries pollutants out of the landfill which leads to emissions. The model heeds a part of preferential water flow and assumes that it has little contact with the landfill body and does not lead to emissions (Doka 2003-III:30). Landfill height and the waste density in the landfill determines how much water is available per kilogram landfill waste. The effective leachate volume per kilogram waste V_{eff} is calculated heeding these modelling choices. The parameter V_{eff} then used to calculate a pollutant's transfer coefficients.

Eq. 3.3
$$V_{eff} = \frac{1 - w\%}{\left(\frac{h \cdot \delta}{I \cdot \alpha_t} - \frac{T_p \cdot \frac{h}{15} \cdot w\%}{v\%}\right)}$$

where,

V_{eff}	Effective leachate volume per kilogram waste (liters/kg·yr)
w%	Share of preferential flow in leachate output (22%)
h	Landfill height (m)
δ	Average waste density (kg/m³); for excavation material and mineral construction waste a value of 2000 kg/m³ is used.
I	Rain infiltration rate (mm/m²a)
α_t	Factor expressing the effect of permafrost conditions (-), cf. (Doka 2017:6)
T_p	Residence time of preferentially flown water (0.17a)
v%	Water content in average waste (20 w-%)

For generic values of type A and B landfills (height 11 m, density 2000 kg/m³,) and a Swiss climate with an annual infiltration I of 300 mm, a value of V_{eff} of 0.0106 liters per kilogram waste and year results. Depending on the user-specified landfill height and climate data, different values can result for V_{eff}.

End of the carbonate phase

Initially, the landfilled waste is buffered at a neutral to slightly alkaline pH of around 7.5–8.5. The material is buffered by the presence of carbonate minerals. Those minerals can be washed out over time and the pH then drops to acidic values. The point in time when the carbonate buffer will be washed out is determined site-specifically depending on the climate parameters. The end of the carbonate phase is expected when all calcium carbonate in the landfill is leached. This point in time is estimated based on the initially available calcium in the landfill, the available infiltration water and the calcium concentration in leachate:

Eq. 3.4 $t_e = {m_{Ca}}/{c_{o,Ca} \times V_{eff}}$ corrected to $t_e \leq 60'000$ years

where,

t_e Time in years when the pH buffer is washed out = carbonate phase [yr]

m_{Ca} Initial concentration of calcium in average landfilled waste [kg/kg]

$c_{o,Ca}$ Concentration of calcium in average leachate [kg/ℓ]

V_{eff} Effective annual leachate volume per kilogram of waste [ℓ/kg.yr]

Leachate concentrations and changes

The effective leachate carrying pollutants out of the landfill is initially assumed to have the average pollutant concentration shown in Tab. 3.2. This reflects the assumption that most relevant pollutants are in a solid/solution equilibrium (solubility-controlled) and that larger or smaller water volumes will not change that. A change in leachate concentration occurs however when the pH buffer is washed out of the landfill over time. As explained above, initially the waste material is well buffered and has high pH values – especially in construction waste landfills. If the buffer is washed out, the carbonate phase of the landfill ends and the pH drops and changes the solubility-controlling phases. For common cations the leachate concentrations increase by two orders of magnitude; for oxianions the concentrations *decrease* (As, Sb, Se, Mn, Mo, Cr, V, W). The factors employed to reflect these changes from pH drop are called x_e in the model and are given in (Doka 2003:51).

The transfer coefficients are calculated in two different temporal dynamics: linear and exponential. An exponential dynamic is assumed for well-soluble pollutants (Na, K, halogens, oxianions) and a linear dynamic for all others (Doka 2003:28).

The formula the exponential transfer coefficients depends on the variables derived above:

Eq. 3.5 $TK_e(t) = TK_\infty \cdot \left(1 - e^{-\left(V_{eff} \cdot c_{o,e} / m_e \cdot TK_\infty\right) \cdot t}\right)$ Exponential transfer coefficient

where,

t Time parameter after waste placement

$TK_e(t)$ Transfer coefficient for element e for time t [kg emitted / kg landfilled]

$TK_{\infty,e}$ Maximal long-term transfer coefficient of the element e, [kg emitted / kg landfilled]

V_{eff} Effective annual leachate volume per kilogram of waste [ℓ/kg.yr]

m_e Initial concentration of element e in average landfilled waste [kg e/kg waste]

$c_{o,e}$ Concentration of element e in average leachate [kg e/ℓ leacahte]

If time *t* surpasses the end of the carbonate phase (t_e) the leachate concentrations will be changed to a value of ($c_{o,e} \cdot x_e$), to reflect the changes of the lowered pH.

The maximal long-term transfer coefficient TK_∞ is set to 100% for all elements, as there is no essential stop to leaching (Doka 2003-III:14). [9]

The formula the linear transfer coefficients is simply:

Eq. 3.6 $$TK_e(t) = \frac{V_{eff} \cdot c_{o,e} \cdot t}{m_e} \quad ,with\ TK_e(t) \leq 100\%$$ Linear transfer coefficient

Transfer coefficients cannot exceed 100%, and in the case of linear transfer coefficients this needs to be procedurally enforced, while in the exponential formula 100% will never be exceeded for any magnitude of parameters.

Approximations

Some of the modelled elements are missing from the literature compilation. For those elements, the transfer coefficients are adopted from similar elements:

Transfer coefficient approximations for excavation material landfill:

- Oxygen and hydrogen is copied from calcium.
- Tin is copied from cadmium, based on based on comparison with construction waste landfills.
- Scandium is adoped from the arithmetic mean of other cations (Ag, Ba, Cd, Co, Cu, Hg, Ni, Pb, Sn, Zn, Be, Sr, Ti, Tl, Fe, Ca)

Transfer coefficient approximations for construction waste landfill:

- Oxygen and hydrogen is copied from calcium.
- Nitrogen is copied from sodium, based on comparison with excavation material landfill.
- Boron is copied from bromium, based on comparison with excavation material landfill.
- Bromium and iodine is copied from chlorine.
- Silver is copied from copper
- Scandium is adoped from the arithmetic mean of other cations (Ag, Ba, Cd, Co, Cu, Hg, Ni, Pb, Sn, Zn, Be, Sr, Ti, Tl, Fe, Ca)
- Tungsten is adoped from the arithmetic mean of other oxianions (As, Cr, Mn, Mo, Sb, Se, V)
- Silicon is copied from aluminium

3.4 One or two working point models?

An initial question was, if there should be two working point models for excavation material landfills (type A) and mineral construction waste landfills (type B) respectively, or if a single model for inert material waste is sufficient. In the following the transfer coefficients derived for a Swiss climate are discussed (infiltration 300 mm per year).

[9] The parameter was initially introduced in the model to heed the unusual stability of chromite in incinerator bottom ash to leaching.

As a helpful visualisation comparison of the short-term transfer coefficients (100 year) with those of a residual material landfill is shown for both landfill types in Fig. 3.1.

In each chart the diagonal line from bottom left to top right represent the location where pollutants are equally mobile as in a residual material landfill. There are some similarities in the mobility behaviour. In either landfill types A or B the oxianions (pink dots) are usually less mobile than in a residual material landfill. Additionally, it is observable that elements with anionic forms (Si, Al, P) are also less mobile. Both these findings can be rationalised with a higher anion exchange capacity (AEC) in construction waste and especially excavation material.[10] A higher anion exchange capacity binds anions better and therefore slows down mobility. Anion exchange capacity of a material depends on a range of characteristics, but can be associated with presence of iron and manganese oxides (McLean & Bledsoe 1992:6). As there is a factor 4–5 more manganese in excavation material and construction waste than in average residual waste material, this is in accordance with the hypothesis of a higher AEC in the former two materials.

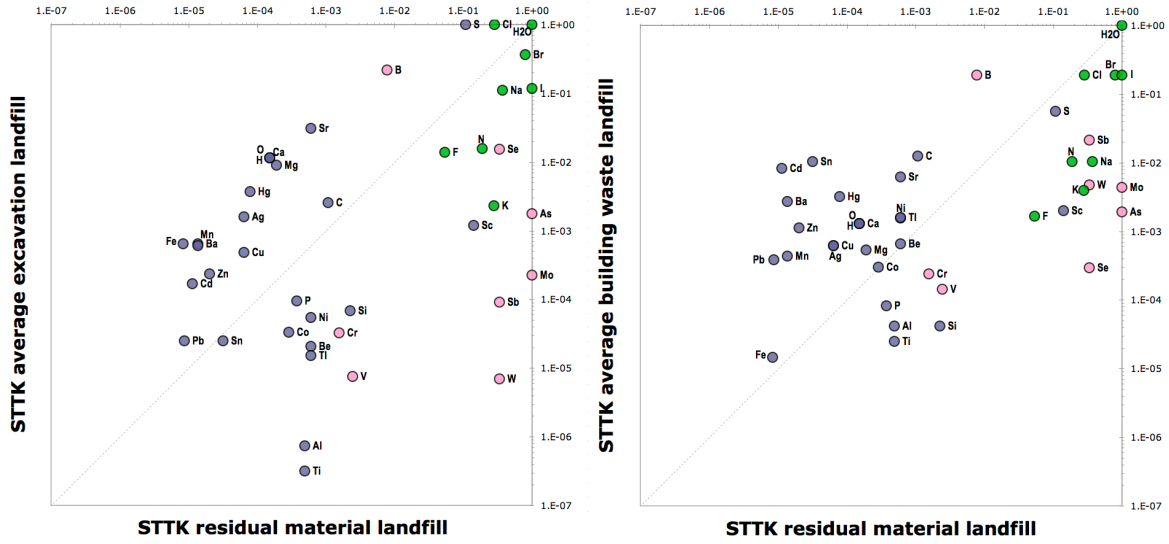

Fig. 3.1 Comparison of the average short-term transfer coefficients of a excavation material landfill (type A, left) and of a mineral construction waste landfill (type B, right) with a residual material landfill (in a Swiss climate). Green dots are well-soluble elements, pink dots are oxianions.

A different observation in Fig. 3.1 is that heavy metal cations (Zn, Pb, Cd, Cu etc.) form a cluster which is typically a bit more mobile than in the residual material landfill. All those findings are an indication that the compiled amount of literature data is sufficient to identify the typical behaviour characteristics of inert material landfills, all inherent variability notwithstanding. Otherwise such tendencies could not be observed and would be drowned out in data noise.

Looking at long-term transfer coefficients some relevant differences occur for oxianions (As, Cr, Mo, Sb, V, W) which are factors more mobile in a construction waste landfill than in an excavation landfill. This is the effect of the large pH buffer present in construction waste, which keeps the pH at high values throughout the modelled timeframe (in a Swiss climate). Conversely, in an excavation landfill the pH buffer is limited and will be washed out within about 8680 years, followed by a pH drop and a decrease of oxianion mobility. Thus on average, excavation landfills are better suited to retain oxianions than construction waste landfill. Since arsenic plays an important role in the average

[10] The pH for all three landfills is high during the short-term phase as all materials are well carbonate buffered. The observed differences in short-term oxianion mobility are therefore unlikely to be an effect of pH.

LCIA results, this is a relevant difference. LCIA results depend on the inventoried specific waste composition and thus for certain wastes, other differences can become relevant.

For the reasons listed above it seems sensible to make two separate models, one for excavation landfill (type A) and one for mineral construction waste material (type B).

3.5 Speciation and categories of emissions

3.5.1 Speciation

For sulfur and nitrogen, there are more than one emitted compound per element. The average leachate concentrations from the literature survey are used to calculate a species profile for sulfur and nitrogen. Also weight increases occur in the inventory because some compounds are inventoried as such, and not as elements (eg. phosphorus P as phosphate PO_4^{2-}, or nitrogen N as nitrate NO_3^-).

Carbon

No single organic compounds are modelled or inventoried, only the total organic carbon emissions (TOC). Dissolved organic carbon (DOC) is assumed to be equal to TOC. The biological oxygen demand (BOD) is derived from the TOC value and a BOD/TOC ratio of 0.61, calculated from the literature survey for construction waste. The chemical oxygen demand (COD) is derived from the TOC value and a COD/TOC ratio of 4.47, calculated from the literature survey for construction waste. No data BOD and COD data is available for excavation material, which is why the ratios for construction waste were also used for excavation material model.

Sulfur

Sulfur in leachate is mostly encountered as the aerobic species sulfate (SO_4). A small fraction is emitted as the anaerobic species hydrogen sulfide (H_2S). The average leachate concentrations from the literature survey are used to calculate a species profile. As data is available for either landfill model, the specific species profiles are employed, although the differences are probably not very significant.

Tab. 3.4 Average leachate concentrations for sulfur species in construction waste landfills and excavation material landfills. From literature indicated in Tab. 3.1.

	Construction waste landfill mg/L	speciation profile	**Excavation material landfill** mg/L	speciation profile
Sulfur in sulfate (SO_4-S)	179.3	99.98%	411.1	99.993%
Sulfur in sulfide (S^{2-}-S)	0.0354	0.0197%	0.029	0.007 %

Nitrogen

Nitrogen in leachate is mostly encountered as nitrate. Some nitrogen is emitted as ammonia (NH_4^+/NH_3). The average leachate concentrations from the literature survey are used to calculate a species profile. As data is available for either landfill model, the specific species profiles are employed, although the differences are probably not very significant.

Tab. 3.5 Average leachate concentrations for nitrogen species in construction waste landfills and excavation material landfills. From literature indicated in Tab. 3.1.

	Construction waste landfill mg/L	speciation profile	**Excavation material landfill** mg/L	speciation profile
Nitrogen in nitrate (NO_3-N)	0.5259	90.94%	3.0546	86.62%
Nitrogen in ammonia (NH_4^+-N)	0.052	9.06%	0.472	13.39%

Chromium

Chromium in leachate is inventoried as the soluble species (Cr-VI). This is based on measurements in inorganic landfills (Cf. Doka 2003-III:77).

3.5.2 Emission categories

For a common wet climate location (with downward leaching) the short-term leachate from the landfill is assumed to be drained via the short sewer pipe to a river or surface water recipient. In the long-term the sewer is assumed to be broken and leachate emissions go directly into groundwater beneath the landfill, as in other landfill models.

For an arid location where leachate is drawn *upward* to the surface, the direct emissions from the landfill occur in the categories low population density air and industrial soil. Leachate flow direction depends on the user-defined climate parameters (see chapter 3.2 'Rainwater infiltration' on page 13). Leachate flow direction is discussed in more detail in (Doka 2017:17).

3.6 Hydrogen sulfide H_2S in inert material landfills

Swiss inert material landfills are operated with the intent of not allowing any significant microbial biodegradation reactions to take place. For that reason the input of degradable organic carbon is severely limited. Generally inert material landfills therefore possess no air emissions (e.g. carbon dioxide CO_2 or methane CH_4 from degraded carbon).

In the past and abroad instances have been found where input of degradable carbon into inert material landfills led to degradation reactions (see e.g Grauwiler 1992). These reactions in turn use up the available free oxygen in the landfill body, which leads to *anaerobic conditions*. In anaerobic conditions the microbial constitution of the landfill changes and different chemical reactions can occur. In anaerobic conditions sulfur in sulfate (SO_4^{2-}) can be converted into sulfide (S^{2-}) and hydrogen sulfide (H_2S). The latter is a volatile gas with a smell of rotten eggs at a very low odour threshold of 0.566 µg/m^3. Under normal aerobic conditions, sulfate in a landfill might be emitted with leachate, which is of only small environmental concern.[11] In anaerobic conditions volatile hydrogen sulfide H_2S might form, which can be a local smell nuisance, but also be oxidises in air to sulfur dioxide SO_2 and contributes to secondary particulate matter formation.

Landfills can be quite heterogeneous and despite best engineering efforts it is possible that anaerobic pockets do form in a landfill body. For the modelling of a inert material landfills it is important to know whether this is a rare occurrence or not. Are the intended aerobic conditions generally met and is

[11] Common LCIA methods do not feature a characterisation or damage factor for emissions of sulfate into water.

the formation of anaerobic species like H_2S, ammonia NH_3 and CH_4 a rare occurrence? One possible indication of this would be the speciation observed in inert material landfill leachate. How large is the fraction of sulfur present as sulfide, the precursor to hydrogen sulfide H_2S? Several literature sources indicate the abundance of sulfide in inert material landfills leachate.

Tab. 3.6 Literature data on occurrence of sulfur in sulfide and sulfate in leachate measurements of inert material landfills and construction waste landfills.

Leachate description	Source	Sulfide-S	Sulfate-S	Sulfide fraction
		mg/l	mg/l	-
Median of 66 responding Swiss inert material landfills queries in December 2007	Hermanns & Moser 2012	0.01	172.3	**0.0058%**
Median from annual leachate measurements from one Austrian building waste (C&D) landfill operated between 1992 and 2008	Laner 2011	0.05	286.7	**0.0174%**
Mean of leachate measurements in 12 building waste (C&D) landfills	Laner 2011:Tab.3-13	0.09	560	**0.0161%**

Pratt (2018) reports a median sulfide fraction of 0.188% for a cleanfill in New Zealand, that also accepted water treatment sludge in the past.

These findings seems to indicate that sulfide generation is a rather *uncommon occurrence* for inert material landfills. But hydrogen sulfide is volatile and its equilibrium mass is concentrated in the gas phase, not the water phase. The leachate concentrations given in Tab. 3.6 above might therefore underestimate the occurrence of overall sulfide generation. As a different possible verification, one can calculate how large a typical conversion rate to sulfide must be, in order for hydrogen sulfide building up to concentrations that will lead to odour nuisance *outside* the landfill area. A coarse estimate[12] yielded that a conversion rate of 0.2% of sulfur to gaseous sulfide, the odour threshold was surpassed outside the perimeter of the landfill i.e. at 200 m distance. Switzerland is a densely populated region and it is safe to assume that such odour effect outside the landfill would not go unnoticed by the public, especially if it were a common occurrence. So it is likely that the actual sulfide conversion rate in inert material landfills is below 0.2%.

Would a sulfide conversion rate of 0.2% be a relevant burden in the process inventory? For average construction waste a maximal conversion rate of total sulfur to sulfide and H_2S air emissions of 0.2% would increase the burden from direct emissions by only 0.6% (calculated with ReCiPe'13 endpoint LCIA[13]). So at most the additional burden from the maximal estimated H_2S emissions would be usually very small. A typical and realistic conversion rate to sulfide in inert material landfills cannot be derived at this time, but it seems likely that this omission is not relevant to LCA results of inert material disposal. A realistic conversion rate might be much below the estimated worst case value of 0.2%.

[12] Assuming an inert material landfill of 40'000 m^2, a depth of 12 m, a gypsum content of 4.6 w% (based on recent statistical data for Switzerland), a rather low average wind speed of 1 m/s, and a simplified plume angle of 45°.

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

3.7 Carbonation of waste building materials

Following their production, certain building materials like concrete or mortar can take up carbon dioxide (CO_2) from the atmosphere and create solid carbonate minerals. This process is called carbonation. Carbonation is the reversal of the reaction taking place when producing cement or calcinated lime/quicklime from solid carbonate minerals; a process called decarboxylation. This decarboxylation results in gaseous CO_2 emissions. Carbonation therefore can effectively revert some of the process CO_2 emissions created during the production of these building materials.

$$(Ca,Mg)O\ \text{(solid)} + CO_2\ \text{(gas)} \underset{\text{Decarboxylation}}{\overset{\text{Carbonation}}{\rightleftharpoons}} (Ca,Mg)CO_3\ \text{(solid)}$$

If the carbonation reaction continues to take place beyond the use phase of the building material, some CO_2 uptake would need to be included in the process inventory of the end-of-life and disposal phases. In the following sections, it is explored whether this is necessary.

3.7.1 Degree of carbonation in construction waste materials

By checking the composition of building material after their use phase, i.e. as waste material, it can be determined, whether further CO_2 uptake in waste is possible. The composition data required for such a investigation is the calcium content (Ca), the magnesium content (Mg) and the total inorganic carbon content (TIC). Those figures from the literature compilation for landfilled construction waste are shown in Tab. 3.7. Carbonate minerals are mostly associated with calcium and magnesium. So by assuming those elements are present as carbonates, a potential maximal TIC content in waste can be calculated stoichiometrically; see last row in Tab. 3.7.

Tab. 3.7 Composition data of average landfilled construction waste regarding its actual and maximal carbonate content. From literature survey, see Tab. 3.2 on page 12.

Average landfilled construction waste composition			
Calcium (Ca)	kg Ca/kg waste	0.174	
Magnesium (Mg)	kg Mg/kg waste	0.012	
Actual Total Inorganic Carbon (TIC)	kg C/kg waste	0.0557	96.3% of maximal TIC
Potential maximal TIC	kg C/kg waste	0.0579	[1]

1 Calculated from calcium and magnesium content as $(m_{Ca}/40.1+m_{Mg}/24.3)\cdot 12$. The magnesium content only contributes 10% of the potential maximal TIC.

This maximal TIC can then be compared with the actually measured TIC content, to determine how far carbonation in the waste material has already progressed. For average construction waste composition the carbonation is virtually complete (96.3% of maximal amount). This is an indication that construction wastes are in all likeliness already *completely carbonated* when arriving at the landfill and any additional CO_2 uptake will be minor. So it seems not necessary to model any CO_2 uptake in building material landfill process inventories – at least in the average case.

Also a study from Japan finds that most of the calcium in concrete rubble is mostly present as carbonate ($CaCO_3$), i.e. has been converted from the originally present hydroxide or oxide ($Ca(OH)_2$ or CaO) (Kikuchi et al. 2010:Tab 4). The authors consider the aggregates in concrete (gravel, sand) to be not reactive. In the reactive hardened cement part of crushed concrete from a ~46 year old building, already initially 70% of the calcium is present as carbonate. After 91 days of dry conditions, this

fraction increases to 84%–89%, and for alternating wet and dry conditions, as they would likely occur in open storage heaps, the fraction can be estimated to be at or close to 100%.

3.7.2 Carbonation of building materials during the use phase

If as observed above, waste building materials are usually already close to fully carbonated. That means that carbonation is prevalently an effect of the *use phase* of the building material, and possibly the demolition phase. In common Swiss LCA of building products, carbonation and the accompanying CO_2 uptake from atmosphere is usually not considered.[14] This might prove to be a relevant assessment gap of building products.

Cement has a typical carbon footprint in the range 0.75 – 0.89 CO_2eq per kilogram (cradle-to-gate results for Portland cements from ecoinvent v3.6). Carbonation can absorb CO_2 in the vicinity of 0.5 kg per kilogram cement. So full carbonation might offset a significant part of the cumulated carbon footprint of the cement production (56% – 67%). The extent of carbonation depends on several circumstances which are summarised in the text box below.

Conditions of carbonation of building materials

Carbonation of building products during their use phase can be quite variable and depends on several circumstances. In *steel-reinforced concrete* carbonation is unwelcome, because it can lead to corrosion of reinforcement steel/rebar due to a lowering of the pH below 9. The increased volume of the rusted steel can cause mechanical pressure and break-off of surface concrete (spalling). Carbonation in reinforced concrete is therefore prevented or limited if possible. Carbonation depends on several conditions: (a) the porosity of the concrete, which is commonly smaller in concrete with a large cement content (small water/cement ratio) and larger in low-cement concrete or poorly produced concrete, (b) humidity of the environment, with rain exposed concrete carbonating more slowly, but also concrete in very arid climates carbonating more slowly, (c) age of the concrete, (d) the air exposure, i.e. material with large surface-volume-ratios or fine grain size will carbonate quickly, but buried surfaces will be less accessible for carbonation, and (e) the CO_2 content of air. Concrete surfaces sheltered from rain or indoors carbonate faster than surfaces exposed to rain. Therefore the extent of carbonation in concrete over time depends on the *exact product and application* and cannot be determined in a generic fashion independent of the application.

For other products like *mortar* from cement or hydrated lime, carbonation is an intended process for the applied product. There carbonation and CO_2 uptake is intended as it leads to desirable product characteristics.

[14] For instance in the Swiss building assessment procedures of eco-bau and the associated LCA data in the KBOB/UVEK database, the topic of carbon uptake by building materials is not addressed (Kellenberger 2020, eco-bau 2020:14).

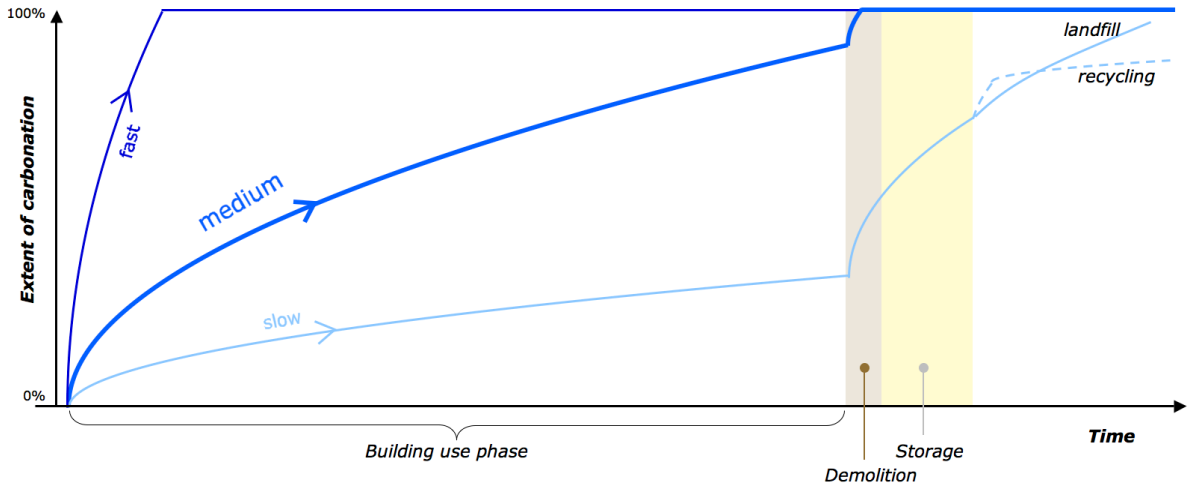

Fig. 3.2 **Theoretical scheme of the degree of carbonation of building materials over time. Shown are possible variations in carbonation speed from slow to fast depending on building material and application. Carbonation speed is proportional to (time)$^{1/2}$. Demolition increases surface area and thus increases carbonation speed. In the medium case carbonation will be close to complete at the end-of-life stage. For the slow case, after demolition, the possibilities for recycling (first with further grain reduction, followed by incorporation into a new compact building product) or landfilling (continued carbonation in a wet environment, with higher CO_2 concentration in pore air) are shown.**

3.7.3 Carbonation of building products in national greenhouse inventories?

A framework where carbonation of building products would need to be considered are the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2006, 2019). There is currently no general IPCC guideline for including carbonation effects in building products. In the 2006 version of the IPCC inventory guideline, the carbonation effect is mentioned in the chapter on cement production and lime production (IPCC 2006-3, page 2.15 and 2.24 respectively). There, neither for cement nor lime production inclusion of carbonation is considered good practice by the guidelines, but an issue of further research. IPCC also considers lime mortars to take a long time to carbonate (IPCC 2006-3, page 2.24). In deviation to this, page 2.26 seems to suggest inclusion of carbonation processes is permissible as long it is reported separately and not as a net figure; the example of precipitated calcium carbonate is given. Page 2.27 is suggesting that decarboxylation emissions from a mineral usage need not be reported, if the mineral production included a carbonation (CO_2 absorption) equalling the CO_2 decarboxylation, i.e. the net figure is zero, and the example of a carbonate mineral produced from a hydroxide is given, i.e. R-(OH)2 → R-CO3.

The 2019 refinement of the IPCC guidelines has hardly extended these few earlier remarks on carbonation. Carbonation is additionally mentioned as a reaction of bauxite/nepheline residues resulting from alumina production (primary aluminium ore refining), which however is not to be included in the inventory methodology (IPCC 2019-3, page 4.77). On page 4.68 of the same volume it is however suggested that CO_2 absorption in bauxite can be included, if it can be quantified.

The takeaway from this is that the IPCC Guidelines for National Inventories regrettably do not offer neither a consistent approach nor a calculation model relating to the estimation of carbonation of building products.

3.7.4 Suggestions for inclusion of carbonation in process inventories

In the landfilling inventory:

As outlined above, the extent of carbonation present at the end of life of a certain building product depends on the exact application/product composition, the use phase conditions and use phase duration. Whether any carbonation of the waste product is still possible in the landfill therefore depends on the use phase conditions. In the course of demolition and sorting of construction waste further carbonation might take place, accelerated by an increased surface-volume-ratio. Also the possibility of longer or shorter intermediate storage periods of demolition rubble and/or crushed and sorted rubble need to be considered.

The extent of carbonation of a landfilled construction waste product should ideally be *measured* as TIC content. From this any further carbonation potential and CO_2 uptake can be calculated – heeding the conditions of the landfill. For buried materials it is not guaranteed that enough atmospheric CO_2 is available to carbonate a material as efficiently as on the surface. So further modelling might be required, to derive accurate figures for CO_2 uptake in the landfill. A further practical problem is that TIC measurements in waste materials are not frequently performed, so it might be difficult to obtain data on the actual extent of carbonation in waste materials. If these problems can be overcome, it would be possible to inventory an additional, product-specific CO_2 uptake for further carbonation during the landfilling process. As shown in Tab. 3.7, on average the carbonation of waste building products being landfilled is close to complete already, so in most cases no further CO_2 uptake need be inventoried.

In the building product use and end-of-life phase:

The extent of carbonation of building products depends on several application-specific conditions as outlined in the text box above (see page 22). The observation that on average building products are almost fully carbonised when they arrive at the landfill (Tab. 3.7) implies that carbonation occurs to a major extent during the use phase and/or the demolition and sorting phase. This is certainly the case for most mortars which by design should carbonate. In reinforced concrete, carbonation during the use phase is being avoided as far as possible in order to prevent corrosion of steel reinforcements.[15]

If most construction waste *arriving* at a landfill is carbonated, but carbonation of reinforced concrete – an important building product by mass – is not complete at the end of its service life, this would suggest two possibilities:

A. Reinforced Concrete carbonates much faster during demolition, triage, sorting phases and any intermediate storage before it arrives at a landfill, and/or

B. Rubble from reinforced concrete arrives not at the landfill, but is rather re-used and recycled.

The continuing carbonation of reused concrete (recycled aggregate) is under investigation (see e.g. Leeman & Loser 2016) and would have to be included in the production and use phase of recycled aggregates. It is relevant to discern carbonation during the use phase and during end-of-life operations in order not to reward a process for a CO_2 uptake that is not really occurring in that process. If the details of carbonation dynamics of specific building product applications are known, the inventories of building processes can be augmented accordingly.

[15] (Leemann & Hunkeler 2016:Tab 2) lists total carbonation depths for various concrete types at the end of their service life in the range of 20 to 80 mm, and a common typical value of 55mm, which corresponds to a carbonation ratio of 27.5% (10% – 40%).

3.8 Landfill infrastructure

The required infrastructure expenditures based on a generic inert material landfill are established. For each kilogram of landfilled waste material a share of the infrastructure is added in the process inventory. Infrastructure of type A and type B landfills is largely similar, but some differences occur.

Area and height

A typical median area of an inert material landfill is assumed to be 63'000 m^2 with a depth of 11 m. This is based on values of 13 existing or planned inert material landfills in Switzerland, taken from (AWEL 2008, Burkhardt et al. 2015, AWEL 2013). The median area is used as default value for both types of landfills investigated here, landfills for excavation material (type A) and mineral construction waste landfills (type B). The user can override the default value with own figures and this is integrated into the inventory calculations (see calculation manual).

Construction

Some inert material landfills are created at former mineral extraction sites, like gravel mines. Some inert material landfills are created on open land, for instance meadows. If built on meadows it is assumed, that initially 1 meter of soil cover is removed. If built on disused mineral extraction sites no such operation is assumed. For excavation material landfills (type A) it is assumed they are built 50% on mineral extraction sites and 50% on meadows. For mineral construction waste landfills (type B) a 100% siting on meadows is assumed. The land transformation exchanges are inventoried accordingly.

At the base of the landfill pit a natural barrier is required by law or alternatively an additional mineral layer with a hydraulic conductivity K of at least $1 \cdot 10^{-8}$ m/s (VVEA Appendix 2). It is assumed that an additional mineral layer of 60 cm is required on 50% of the sites of either landfill type and is assumed to be sand (density 1600 kg/m^3).

Tab. 3.8 Assumptions for infrastructure expenditure raw data for two types of inert material landfills.

		Excavation material landfill (type A)	Mineral construction waste landfills (type B)
Area	m^2	63'000	63'000
Siting		50% on mineral extraction sites, 50% on meadows	100% on meadows
Removed soil	m^3	31'500	63'000
Sand base layer	kg	30'240'000	30'240'000
Base drainage pipe at bottom	m	1004	1004
Sewer pipe to discharge point	m	300	300

At the bottom of the landfill pit a base drainage pipe is installed to collect leachate water. Polyethylene tubes are assumed with a diameter of 10 cm and a wall thickness of 1cm. The length of the tubes is assumed to correspond to the circumference of the area, which is 1004 m.[16]

The collected leachate is not treated, but it is assumed to be lead to a discharge point by a short sewer pipe. An estimated length of 300 m concrete sewer pipe is assumed with a diameter of 17 cm and a thickness of 10 cm (based on Burkhardt et al. 2015:23). Additional materials for sewer pipe are

[16] The circumference is derived in a simplified manner from the surface area (63'000m^2) by assuming a square shape with an edge of 251m and neglecting any flank slopes.

estimated from a class 5 (small scale) sewer system from (Doka 2003-IV:12), and in proportion to the calculated concrete pipe mass.

Access street

For construction, operation and aftercare of the landfill, a short access road is assumed. The road length is assumed to be 50 m with a width of 6 m. For road construction a diesel consumption of 0.2 litre per m^2 of road is assumed. For the road a gravel bed of 0.5 m thickness is assumed, covered with a bituminous concrete layer of 0.1 m and 2500 kg/m^3 density, consisting of 15w% bitumen and 85w% gravel.

Recultivation

For recultivation a gravel layer of an assumed thickness of 0.5 m is placed on top of the finished landfill. An additional soil layer of 0.8 m is the final top layer[17]. The material is assumed to be distributed by loaders and diggers.

Material transport

In EcoSpold2 the transport of required infrastructure materials is inventoried via the use of market datasets. In EcoSpold1 the dataset author has to apply standard transport distances and transport modes to calculate the resulting transport service requirements (in tonne·kilometres) and include them in the process inventory. For the EcoSpold1 inventory the transport services required for infrastructure materials are calculated and included in the inventory.[18] They are not included in the EcoSpold2 inventory.

Land use

As explained in section 'Construction' on page 25, construction waste material landfills (type B) are assumed to be built on former meadows, while for excavation material landfills (type A) a split of 50% former mineral extraction site and 50% meadow was assumed. The area is transformed to a "dump site, inert material landfill".

The landfill depth is a user-specific parameter and can be varied. With a landfill area of 63'000 m^3, a generic depth of 11 m and a waste density of 2000 kg/m^3 a total waste mass of 1'386 million kilograms per landfill unit results; and correspondingly for 1 kg of waste a land transformation of 0.0000455 m^2/kg is required. But the inventoried figure will depend on the user-defined landfill depth.

The landfill operation phase duration is also user-specified, but a period of 15 years is suggested as a default. The length of the operation phase determines the landfill's land use occupation in m^2.yr. It is assumed that the whole landfill area is a dump site during this time (i.e. CORINE class 132), although it is possible in reality for the landfill to be operated in sequential zones and batches.

After operation an active recultivation to a meadow is assumed for both types of landfill (cf. section 'Recultivation' on page 26). The according land transformations from "dump site, inert material landfill" to "meadow" are inventoried.

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[17] "Soil" is not available as a technosphere product in either versions of the ecoinvent databases. In EcoSpold02 the input of soil should be inventoried explicitly. But no processes with soil outputs (e.g. in building construction) are inventoried as such. In EcoSpold1, the soil can be assumed to be recycled and received from a cut-off. Spreading of soil with diggers and loaders is inventoried in ES1 and ES2 inventories. In ES1 additional the transport of soil is inventoried (20 km with lorry).

[18] The standard distances are 20km lorry for gravel & sand, which is also adopted for transport of recultivation soil; plastic materials are transported 50km by lorry and 200 km by train; steel/metal materials are transported 50km by lorry and 600 km by train.

Added to the above land uses is the land transformation and occupation for the access road. The access road is assumed to be needed for the landfill use phase (user-defined) plus a period of aftercare lasting 25 years.

3.9 Landfill operation

For the operation phase of inert material landfills only the placement and compaction of the waste is inventoried. The functional unit of 1kg waste is converted to a volume, using an average density of 2000 kg/m^3 and the resulting 0.005 m^3 is inventoried as an excavation work with diggers and loaders. This assumes to cover waste placement and compaction.

As with all other ecoinvent disposal datasets[19], the delivery of the waste at the disposal facility is not included in the inventory of disposal.

3.10 New exchanges for environmental scarcity LCIA

The method of ecological scarcity (a.k.a. eco-scarcity, or MOeK, or UBP) is a Swiss LCIA method (ÖBU 2013). 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.** This is *not the carbon emitted* from a landfill, but the organic carbon in the waste material as it was initially *deposited into a landfill*. This is a mass flow that is within the technosphere and not at the technosphere-biosphere boundary like conventional LCI emissions.[20] Until now it was not possible to apply this characterisation factor correctly in database LCIA calculations.[21] A similar characterisation factor for the **total waste mass placed in a landfill** will be introduced in the ecoscarcity method update planned for 2020.

In order to asses these two material flows for ecoscarcity accurately, new exchanges are introduced, which accurately represent the targeted mass flows. These exchanges were discussed and defined with the ecoscarcity authors (Fredy Dinkel, Thomas Kägi, Rolf Frischknecht) and are introduced in the disposal inventories created in the updated tools.

The LCIA calculation itself is performed in either databases or LCA software. Therefore the databases or LCA software administrators need to do two things:

step 1: Introduce/define the new exchanges in the database/software.

step 2: Change LCIA implementation of ecoscarcity to include the new exchanges with their characterisation factors

Define new exchanges in databases

The owners of the ecoinvent database agreed in April 2020 to create the new exchanges (step 1). Also the managers of the Swiss KBOB database for building assessment at treeze have agreed in March

[19] The only exception being wastewater treatment, where the sewer transport of wastewater to municipal wastewater treatment plants is included in the wastewater disposal inventories.

[20] Since the landfill is clearly a man-made artefact and as such firmly within the technosphere. Although a landfill is *exposed* to influences from the biosphere, it should not be considered a part of the biosphere.

[21] Attempts to overcome this by the ecoscarcity authors involved valuing the long-term *emissions* of carbon from the landfill, i.e. a part of the *output* from the landfill (ÖBU 2013:191). Due to variable behaviour of carbon in different waste materials and in different landfills this introduced distortions compared to the true "input carbon" value.

2020 to create the new exchanges. As ecoinvent works with EcoSpold2 and treeze works with EcoSpold1, this covers already a wide range of applications. In other applications or databases, the managers introducing the new exchanges for ecoscarcity should be using the information compiled below for consistency.

Tab. 3.9 The two new exchanges for EcoSpold2.

Name English	Waste mass, total, placed in landfill	Organic carbon, placed in landfill
Category	natural resource	natural resource
Subcategory	in ground	in ground
Unit	kg	kg
Comment	This exchange should only be used to apply ecofactors for the Swiss LCIA method of Ecological Scarcity and not any other purpose. The exchange represents a flow that is within the technosphere	This exchange should only be used to apply ecofactors for the Swiss LCIA method of Ecological Scarcity and not any other purpose. The exchange represents a flow that is within the technosphere
UUID	6bc06a91-ae35-4a2b-ab39-da4dd36b621a	4044e84c-26c5-4cef-b76c-8c660d60bcfe

Tab. 3.10 The two new exchanges for EcoSpold1.

Name English	Waste mass, total, placed in landfill	Organic carbon, placed in landfill
Name German	Abfallmasse, gesamt, einer Deponie zugeführt	Organischer Kohlenstoff, einer Deponie zugeführt
Category	resource	resource
Subcategory	in ground	in ground
Unit	kg	kg
Comment	This exchange should only be used to apply ecofactors for the Swiss LCIA method of Ecological Scarcity and not any other purpose. The exchange represents a flow that is within the technosphere	This exchange should only be used to apply ecofactors for the Swiss LCIA method of Ecological Scarcity and not any other purpose. The exchange represents a flow that is within the technosphere

The comments to the new exchanges are intended to assure a *proper application* of these exchanges. The proper application is only for applying ecofactors of the Ecological Scarcity LCIA method. The corresponding flows, especially for carbon, should neither be seen as carbon sequestration figures, nor as carbon emissions from landfills.[22]

Implementation of ecoscarcity LCIA

Step 2 "LCIA implementation of ecoscarcity" is straightforward for the *ecoscarcity 2020 version*. As of this writing (June 2020), the magnitudes for characterisation factors of the ecoscarcity update are not known yet.

In addition to the new ecoscarcity version, the ecofactors could also be used with advantage in *older ecoscarcity versions*. Versions 2008 and 2013 had already ecofactors for valuation of organic carbon

[22] The flows do not represent a carbon sequestration, as they represent merely the *initial* amount of carbon stored in a landfill, including the carbon that will be emitted afterwards in the form of landfill gas or as leachate. A significant amount of carbon placed in landfill can therefore be released and not retained. Counting carbon placed in landfill as a sequestration also would – in the case of biomass materials – *double count* the already inventoried uptake of CO_2 from air during biomass growth. The new exchanges are also not suitable to represent emissions, as this would double count the proper emissions figures already given in the inventory separately.

in landfills (15 and 5.5 UBP per gram C, respectively). The new exchange for organic carbon placed in landfill allows a much more accurate assessment of this burden than the previously used solution of characterising long-term TOC emissions. The latter ecofactors needs to be removed from ecoscarcity calculation, when the new exchanges are employed in LCIA.

Valuation of total mass in landfills was already employed in the (legacy) ecoscarcity versions of 1990 and 1997 (220 and 500 UBP per kilogram waste, repectively). The solution employed previously for these factors was a valuation via the landfill surface area, which was available from the inventories. If still needed, the new exchange for waste mass placed in landfill allows an alternative and more flexible implementation of those ecofactors.

3.10.1 Augmentation of already existing datasets

The above refers to newly generated datasets. For consistency, already existing datasets with landfilling activities in them should additionally be augmented to include in their process inventories the two new exchanges for waste mass and organic carbon placed in landfill. 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".

The new exchanges shall be added where the actual waste-specific landfilling process is inventoried with its emissions and process expenditures. This location is not always absolutely clear from process names alone, as there have been instances, where authors made inventories consisting only of a request to existing landfilling processes, but using the naming conventions of a landfill inventory.[23] To avoid double counting, exchanges shall only be added in the "final sink" landfill processes, but not in processes merely *requesting* those processes.

In 2003 for several landfills, datasets containing constant, *process-specific* expenditures independent of waste composition were created for ecoinvent (e.g *"process-specific burdens, residual material landfill"*). These service datasets contained constant information applicable to all waste compositions and were then requested in the *waste-specific* landfilling datasets. It is not advisable to add the new waste mass exchange in those process-specific service datasets. As not all landfilling processes refer to the old "process-specific burdens", e.g. unsanitary landfill, it is advisable to have the augmentation in a uniform pattern in the process that includes the waste-specific landfilling expenditures.

[23] For instance in the KBOB database contains a process called "disposal, render carrier board, resin bound, to inert material landfill". The name of this process has the exact structure of a inert landfilling process, but this is misleading. Its inventory contains merely links to other processes, like inert waste to landfill, plastic plaster to landfill, and glass to landfill. Such a process must not receive any of the new exchanges again, as they will be properly added in the requested datasets.

4 EcoSpold1 Export

Since 2017 the suite of Excel waste tools (Doka 2017) is able to create inventories directly in the file format **EcoSpold2**, to be used in the ecoinvent database v3+ (versions 2011 and later). Since their first inception in 2003, the Excel tools were able to create **EcoSpold1** files, the file format of the ecoinvent 2000 project (2003-2010), but only via ecoinvent's Windows add-on for Excel called EcospoldAccess, which by now is largely deprecated software. As EcoSpold1 is a comparatively simple file format, the tools are now given the additional possibility to create EcoSpold1 files directly. Instructions on how export EcoSpold1 files are given in the updated calculation manual (Doka 2020).

4.1 Implementation remarks

Many of the EcoSpold1 contents can be obtained directly from the EcoSpold2 contents that had to be already provided in the Excel waste tools to produce valid EcoSpold2-XML-Files. In a few instances EcoSpold1 (ES1) has different information requirements not matching those of EcoSpold2 (ES2). The handling in those instances is explained below.

4.1.1 Element productionVolume

The name "productionVolume" notwithstanding, in this element of EcoSpold1 the **annual consumption** in the geographic area shall be stated, **not the production**.[24] Considering imports and exports this is not equal to the production volume as it is used in EcoSpold2. The contents of this element cannot be easily derived from the ES2 production volume. The element has no quantitative function in the ES1-LCI calculation; and even if an influence were desired: it's contents are a text string, can contain free text, and cannot dependably be parsed into a number. The element is not required within ES1. A figure for such a consumption volume would be nice to have, but requires the author to provide that figure with little benefit to the LCI result. For all these reasons it was decided to skip this element for the ES1-file generation.

The actual production volume is a required figure in ES2. In order not to loose this information and not to produce any confusion, in the generated ES1 file any production volume given by the user is added in the ES1 *technology comment*. The added text template is the following:

- *The annual production volume (APV) of this dataset is ## kg/yr. APV comment: xxxxxx*

4.1.2 Element Representativeness percent

The 'percent' element of the Representativeness tag of EcoSpold1 is intended to indicate how representative the inventoried process is for the *market supply of indicated region and time period* (Ecoinvent 2002). This shall be given as a percentage of the total market.

This figure of ES1 is informative only and not a required element. It has no bearing on LCI calculations of ES1 files. The contents of this element cannot be easily derived from the ES2 production volume. To do this, a figure for the local market supply volume would be needed and a second figure indicating the fraction of production being exported, and thus not entering the market

[24] The EcoSpold01 Schema Documentation states: *"productionVolume indicates the market area consumption volume (NOT necessarily identical with the production volume) in the geographical area indicated of the product/service at issue. The market volume should be given in absolute terms per year and in common units. It is related to the time period specified elsewhere."* The bracketed phrase is in the original text (Ecoinvent 2002).

supply of indicated geographic region. For all these reasons it was decided to skip this element for the ES1-file generation.

As noted above, a given ES2 annual production volume will be written as an additional text into the ES1 technology comment.

4.1.3 Tag validation (review)

The EcoSpold1 definition contains a tag which carries the text and information of the reviewer of the dataset.[25] The dataset author is not – or should not be – the same person as the dataset reviewer. While in the ecoinvent 2000 project (2003-2010) all published datasets would be validated and reviewed by a neutral person, clear procedures for review in EcoSpold1-datasets are missing now. Presumably a reviewer would have to include his or her review text to the dataset after review.

In common use many EcoSpold1-datasets are published that have no validation tag.[26] This is not ideal, as it is not clear whether these datasets have been reviewed at all or whether merely the specific tag was not filled out or is missing. But it indicates that a missing validation tag does not lead to import errors in LCA software. This means usable EcoSpold1-files can under circumstances be lacking the validation tag.

A review is preferable, although in the review practices of the past, the validation tag hardly ever contained really valuable information. If a review is performed in a project, it is advantageous if the resulting review comments can be included. As lean solution, a reviewer's name can be selected in sheet "General Data" of the Central Repository workbook (cell F26). Also a review comment text can be entered.[27] If a reviewer name is lacking, no validation tag will be created. If a reviewer name is provided, it is added in the proofReadingValidator element of the ES1 validation tag. If a reviewer text comment is given, it is entered in the proofReadingDetails element. If only a reviewer name is given but the review comment is empty, the text in proofReadingDetails will read "[no review comment provided]".

4.1.4 "Data Generator" and "Data Entry By" person

The EcoSpold1 Scheme documentation (Ecoinvent 2002) features two separate and mandatory elements for the persons involved in the dataset creation:

1. The "Data Generator" is the person who compiled or authored the inventory. The ES1 documentation describes this as "the person that prepared the dataset". The person is referenced in the tag <dataGeneratorAndPublication>.

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[25] The tag contains three elements 1) proofReadingDetails: Comment of the reviewer, 2) proofReadingValidator: a number identifying the reviewer, 3) otherDetails: further text added later, e.g. feedback from users. The first two items are compulsory (Ecoinvent 2002)

[26] For instance none of the 6801 datasets in the KBOB-UVEK database carry any review or validation information (KBOB et al. 2018). Similarly, in the LCI repository of ESU services at www.esu-services.ch/data/public-lci-reports many of the EcoSpold01 inventories carry no review or validation information. Datasets converted with the EcospoldAccess/Excel2EcoSpold tool seem to have a validation tag.

[27] This is a simple lean solution, as there is only one reviewer and one text to be entered. A more elaborate solution would be to allow for a separate reviewer and text for each of the hundreds of datasets defined in sheet "DS info" of Central Repository workbook. In view of the limited information usually carried in the EcoSpold1 validation tag, this seemed an excessive solution, even though it would be the more precise.

2. The "Data Entry" person is commonly understood as the one that entered or uploaded the created inventory into the database. But according to the ES1 documentation it is "the person that prepared the dataset *and* enters the dataset into the database" [sic, emphasis added]. The person is referenced in the tag <dataEntryBy>.

The **given EcoSpold1 definitions** seem to imply an *overlap* between the two performed functions: The generator is the one who prepared the dataset, while the entry person has prepared *and* entered the dataset. This overlap in definition does not seem to make much sense, unless one assumes a dataset is always created by two persons, where *both* are preparing the dataset, but only *one* of them enters the dataset into the database therefore becoming an author and entry person.

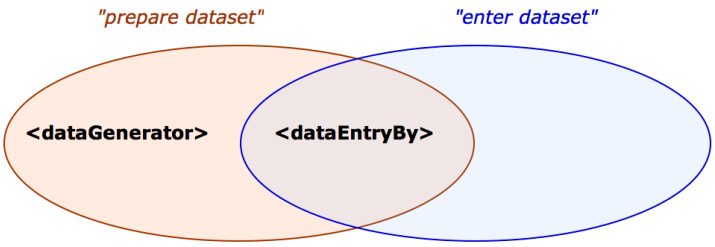

Fig. 4.1 Euler diagram of the peculiar logical structure of the definitions of the "dataset generator person" and the "dataset entry person" according to the EcoSpold1 Scheme Documentation (Ecoinvent 2002).

In actual practice of the **ecoinvent 2000 project** (2003-2010) the data generator was usually the *author* of the dataset, compiling and preparing the inventory, while the data entry person had no inventory authorship, but was merely performing a database service or secretarial function (this in violation of the formal definition of the DataEntry person as an author *and* secretary). So in practical terms the two functions were usually separated. It was also possible to have the same person referenced in both elements.

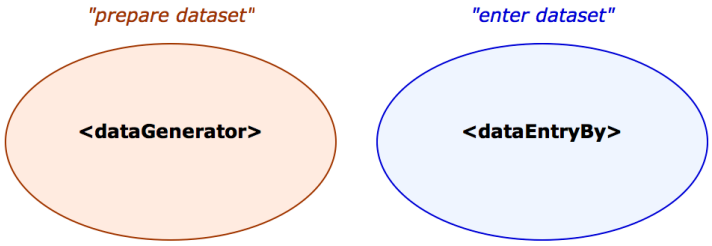

Fig. 4.2 Euler diagram of the actual practice of using the ES1 tags for "dataset generator person" and the "dataset entry person".

In the Excel waste tools the author of the datasets has to identify him/herself (Central Repository workbook in sheet "General Data"). For the present EcoSpold1 export function, it was decided to use the setting placed there in *both elements* for "Data Generator" (author) and "Data Entry By" (author, secretary).

4.1.5 Country information for persons

In EcoSpold 1 and 2 persons are identified with the person's name, address, phone number, e-mail, and company affiliation. In EcoSpold1 person definitions are also required to contain a *country code* element (2-letter ISO, like CH, DE, US, NL etc.). In EcoSpold2 the countryCode field has been discontinued, but a country may or may not be specified in the field "address" in free text. It is

therefore not possible to reliably parse a country code from the ES2 address field. Since in ES1 the country code element is listed as a required element in a person's definition, it must be present for a strictly valid ES1 file (Ecoinvent 2002).

As lean solution, the country code to a data generator person can be given in sheet "General Data" of the Central Repository workbook as a 2-letter ISO code (cell F25). To avoid creation of invalid ES1 files, the country code for Switzerland (CH) is entered there, but can be overwritten. The same country code is also used for the reviewer person.

4.1.6 Database version

In the tag <dataSetInformation> is an element for the version of the database the dataset is contained. This is a required field. Originally this was intended to contain the ecoinvent database version, the dataset belongs to (ecoinvent used EcoSpold1 from 2002 to 2010, in versions 1.0–2.2). Since the tools are free to users and can be used *outside* of work with ecoinvent, it does not make sense to put any kind of ecoinvent version number there.[28] Since this is a required field, it was decided to put the version number "0.00" as encountered in several other more recent EcoSpold1 files.

4.2 Multi-functional datasets

Ecospold1 allows the creation of multi-functional datasets. Those are for processes which produce *more than one useful product*. Recycling processes are for instance often multi-functional in reality in as they produce a reusable recyclate, but also provide the service of waste material disposal with the input material. Or municipal waste incinerators perform a disposal service, which is their main source of revenue, but can also produce a usable net energy output.

The datasets from this report on construction waste and excavation materials landfills are mono-functional in as they only "produce" the service of waste disposal, and as such do not require any allocation. But the EcoSpold1 export functionality is also implemented in other waste treatment models, some of which can be multi-functional, e.g. municipal incineration. For those datasets, multi-functionality must be addressed. How multi-functionality is treated in the EcoSpold1 export routines is presented below.

On a *technical* level, it would be possible to create EcoSpold1 process inventories with multiple functions and accompanying allocation factors for exchanges, as the EcoSpold1 definition allows for allocation (tag <allocation> with the elements *referenceToCoProduct* and *fraction*). On a *practical* level, this solution is not ideal for two reasons:

1. To be useful in a relational database, a multifunctional EcoSpold1 inventory needs to be *split up* into its allocated parts using the defined allocation factors. For instance from one unallocated dataset, *three* different datasets could be created. It is not clear, whether the recipient software of the files created here, such as SimaPro or OpenLCA, are able to perfom this allocation procedure of multifunctional inventory files. Ordinary monofunctional datasets are less cumbersome in this respect.

[28] If a dataset is elaborated outside of ecoinvent, it would be misleading to imply an ecoinvent version number. And if a dataset was created for ecoinvent, the *current* ecoinvent database uses EcoSpold2, not EcoSpold1, so a version number of 3.+ does not make sense for an EcoSpold1 file. And entering a *previous* database version number like 2.2 carries the objectionable deception that this dataset was part of the ecoinvent v2.2 database, when it was not. The ecoinvent 2.2 database was published in 2010 and retro-active additions to it were neither planned, nor are they formally correct. So not any version number makes sense for datasets created *now*.

2. In ecoinvent v2.2 and databases referring to its guidelines (like the KBOB/UVEK databases) which use EcoSpold1, allocation in disposal processes was usually handled simply by allocating 100% of the burden to the chief function of waste disposal, and 0% to the co-products like generated net energy or reusable materials. In that way, these multifunctional datasets were in essence recorded like *monofunctional* disposal datasets.[29] A 100%–0% allocation is largely superfluous.

For these two reasons it is decided that the EcoSpold1 files exported from the calculation tools are monofunctional in nature and carry 100% of the process burden which is attributed solely to the disposal service as the reference product. Any co-product datasets are *not created* as they would carry no burden information and serve little use in a database and make no difference to ultimate LCA results. 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 provided only as additional text information[30]. The comment is added, if applicable, in municipal incineration and sanitary landfills with landfill gas utilization. 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. 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 and all solid recyclates amount to less than 1%. This alternative allocation is close to alternative economical allocation calculated in (Doka 2003:22) where burden on disposal with economical allocation would have been 93%, but the 100%–0% allocation was employed in the inventory.

[29] Another practical advantage of this approach is that the database is not needlessly filled with over-granular datasets like "heat, from waste polyethylene incineration" and "heat, from waste paper incineration" etc. with little utility to the average practitioner. An exception was made in 2003 for average municipal solid waste mixture in incineration, which was designed as a multi-functional inventory with the two energy products heat and electricity as co-products, which alas were allocated 0% of the burden by default. But alternative allocations factors were possible and could be introduced in that dataset.

[30] 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. 4.1 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
Disposal service	kg	0.3225	1
Produced net heat	MJ	0.00412	2
Produced net electricity	kWh	0.04352	2
Steel scrap	kg	0.175	3
Aluminium scrap	kg	0.806	3
Copper scrap	kg	2.84	3
Zinc concentrate (hydroxide sludge)	kg	0.0042	3

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 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.

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

5 New datasets for EcoSpold1

5.1 Introduction

In 2003 several datasets were created in the ecoinvent 2000 project to inventory waste disposals in what was then called inert material landfills. These were largely for unpolluted, inorganic materials like concrete, or alternatively for small amounts of "carry-along" materials, like for instance biomass fibres as part of reinforced gypsum boards. As the direct emissions of inert material landfills were not yet inventoried in 2003, all datasets had the same burdens from process-specific expenditures only (infrastructure & operation).

With the new models new datasets can now be created including waste-specific, direct landfill emissions. As the landfill type of inert material landfills are now distinguished into type B landfills (mineral construction waste landfill) and type A landfills (excavation material landfill) the datasets obtain a *new dataset name*. Also the time period of the datasets is updated from (1994-2000) to (2006-2012). The correspondence between old dataset names and new dataset names is shown in Tab. 5.1 below.

Up until now the waste composition did not play a role in the inert material landfill datasets. With the new model, waste-specific emissions are calculated depending on the waste composition. Therefore the waste composition of the waste materials has to be specified, and in some cases new compositions had to be researched. The used compositions are outlined in the following chapters.

The origin of the waste materials is described below and in (Doka 2003-I). The origin is especially relevant for production specific waste, for instance zeolite, which is a purified inorganic catalyst carrier material from oil refining and cracking.

For most of the new datasets a disposal in a *construction waste landfill* is assumed. As the wastes are mostly building materials and process waste, this seems the most appropriate choice. The other

possible "inert material landfill" would be an excavation material landfill, but in ecoinvent 2000 excavation material was usually assumed to be recycled and was not inventoried. It is therefore unlikely that the inert materials are for excavation material.

Additionally to the datasets for wastes based on already existing "inert material landfill" inventories, also three new datasets are created (bottom three lines in Tab. 5.1).

All datasets are sited in Switzerland and use the average Swiss climate parameters to calculate leachate flow.

Tab. 5.1 Correspondence table for old datasets (without landfill emissions, "inert material landfill") and the new datasets (with landfill emissions, "construction waste landfill" or "excavation landfill") for both EcoSpold1 (Ecoinvent 1.0-2.2) and EcoSpold2 (ecoinvent v3+).

Old ecoinvent 2000/ EcoSpold1 name	Old eiv3+ EcoSpold2 name	New EcoSpold1 name EN	New EcoSpold1 local name DE	New eiv3+ EcoSpold2 name
disposal, concrete, 5% water, to inert material landfill	treatment of waste concrete, inert material landfill	disposal, concrete, 5% water, to construction waste landfill	Entsorgung, Beton, 5% Wasser, in Bauabfall-Deponie	treatment of waste concrete, construction waste landfill
disposal, emulsion paint, 0% water, to inert material landfill	treatment of waste emulsion paint, inert material landfill	disposal, emulsion paint, 0% water, to construction waste landfill	Entsorgung, Dispersionsfarbe, 0% Wasser, in Bauabfall-Deponie	treatment of waste emulsion paint, construction waste landfill
disposal, glass, 0% water, to inert material landfill	treatment of waste glass, inert material landfill	disposal, glass, 0% water, to construction waste landfill	Entsorgung, Glas, 0% Wasser, in Bauabfall-Deponie	treatment of waste glass, construction waste landfill
disposal, gypsum, 19.4% water, to inert material landfill	treatment of waste gypsum, inert material landfill	disposal, gypsum, 19.4% water, to construction waste landfill	Entsorgung, Gips, 19.4% Wasser, in Bauabfall-Deponie	treatment of waste gypsum, construction waste landfill
disposal, inert waste, 5% water, to inert material landfill	treatment of inert waste, inert material landfill	disposal, inert waste, 5% water, to construction waste landfill	Entsorgung, Inertstoff, 5% Wasser, in Bauabfall-Deponie	treatment of inert waste, construction waste landfill
disposal, limestone residue, 5% water, to inert material landfill	treatment of limestone residue, inert material landfill	disposal, limestone residue, 5% water, to construction waste landfill	Entsorgung, Kalksteinrückstand, 5% Wasser, in Bauabfall-Deponie	treatment of limestone residue, construction waste landfill
disposal, mineral wool, 0% water, to inert material landfill	treatment of waste mineral wool, inert material landfill	disposal, mineral wool, 0% water, to construction waste landfill	Entsorgung, Mineralwolle, 0% Wasser, in Bauabfall-Deponie	treatment of waste mineral wool, construction waste landfill
disposal, natural gas pipeline, 0% water, to inert material landfill	treatment of decommissioned pipeline, natural gas, inert material landfill	disposal, natural gas pipeline, 0% water, to construction waste landfill	Entsorgung, Erdgasleitung, 0% Wasser, in Bauabfall-Deponie	treatment of decommissioned pipeline, natural gas, construction waste landfill
disposal, packaging cardboard, 19.6% water, to inert material landfill	treatment of waste paperboard, inert material landfill	disposal, packaging cardboard, 19.6% water, to construction waste landfill	Entsorgung, Verpackungskarton, 19.6% Wasser, in Bauabfall-Deponie	treatment of waste paperboard, construction waste landfill
disposal, paint, 0% water, to inert material landfill	treatment of waste paint, inert material landfill	disposal, paint remains, 0% water, to construction waste landfill	Entsorgung, Anstrichstoff Reste, 0% Wasser, in Bauabfall-Deponie	treatment of waste paint, construction waste landfill
disposal, plastic plaster, 0% water, to inert material landfill	treatment of waste plastic plaster, inert material landfill	disposal, plastic plaster, 0% water, to construction waste landfill	Entsorgung, Kunststoffputz, 0% Wasser, in Bauabfall-Deponie	treatment of waste plastic plaster, construction waste landfill
disposal, polyurethane, 0.2% water, to inert material landfill	treatment of waste polyurethane, inert material landfill	disposal, polyurethane, 0.2% water, to construction waste landfill	Entsorgung, Polyurethan, 0.2% Wasser, in Bauabfall-Deponie	treatment of waste polyurethane, construction waste landfill
disposal, slag from MG silicon production, 0% water, to inert material landfill	treatment of slag from metallurgical grade silicon production, inert material landfill	disposal, slag from MG silicon production, 0% water, to construction waste landfill	Entsorgung, Schlacke aus MG-Silizium Prod., 0% Wasser, in Bauabfall-Deponie	treatment of slag from metallurgical grade silicon production, construction waste landfill
disposal, steel, 0% water, to inert material landfill	treatment of scrap steel, inert material landfill	disposal, steel, 0% water, to construction waste landfill	Entsorgung, Stahl, 0% Wasser, in Bauabfall-Deponie	treatment of scrap steel, construction waste landfill
disposal, zeolite, 5% water, to inert material landfill	treatment of waste zeolite, inert material landfill	disposal, zeolite, 5% water, to construction waste landfill	Entsorgung, Zeolith, 5% Wasser, in Bauabfall-Deponie	treatment of waste zeolite, construction waste landfill
–	–	disposal, gravel, 0.2% water, to construction waste landfill	Entsorgung, Kies, 0.2 Wasser, in Bauabfall-Deponie	treatment of waste concrete gravel, construction waste landfill
–	–	disposal, cement, 5% water, to construction waste landfill	Entsorgung, Zement, 5% Wasser, in Bauabfall-Deponie	treatment of waste cement in concrete and mortar, construction waste landfill
–	–	disposal, excavation material, clean, 20% water, to excavation landfill	Entsorgung, Aushub, sauber, 20% Wasser, in Aushub-Deponie	treatment of excavation material, clean, excavation landfill

5.2 Waste compositions for datasets

Concrete

In the literature compilation for this study, concrete waste is recorded separately (cf. Tab. 3.1). From that compilation a waste concrete composition is compiled. For missing data, a synthetic concrete composition from 66% crustal material (for aggregate) and 33% cement is used as approximation. The resulting composition is corrected to 100% by adjusting the oxygen content. A water content of 5% is used, as specified by the EcoSpold1 waste name.

Inert waste, mineral building waste

In the literature compilation for this study, average, generic mineral building waste is recorded separately (cf. Tab. 3.1). That compilation represents the mixture of waste landfilled in construction waste landfills (VVEA type B) and is used as the unspecified, generic "inert waste in an inert material landfill". The composition is corrected to 100% by adjusting the oxygen content. A water content of 5% is used, as specified by the EcoSpold1 waste name.

Natural gas pipeline

In the dataset for high-pressure underground pipelines for natural gas transport, the disposal of the used pipeline materials assumes that at most 50% of the pipeline is removed and the rest *is left in the ground*. To inventory the pipeline materials left in the ground on site, a disposal in an inert material landfill was inventoried as an approximation by the dataset authors (see the ecoinvent v3 dataset "pipeline construction, natural gas, high pressure distribution network/CH"). The dataset authors assume transfer of 50% of the end-of-life pipeline materials (cast iron, polyethylene) to separate disposals off-site. The same is done for 50% of the concrete, reinforcing steel and pitch. Left in the ground are therefore 50% of those materials plus also 100% of materials not moved off-site, which are cement and sand.[31]

The corresponding complex waste consisting of cast iron, cement, concrete, pitch, polyethylene, steel and sand is inventoried as the decommissioned pipeline materials left in the ground. Composition of pitch (bitumen), polyethylene, and steel (scrap) is available from Doka 2003. A new composition for cast iron is derived from data for ductile cast iron used in pipes (Wikipedia 2020). The composition is corrected to 100% by adjusting the iron content. A cement composition was derived from the literature survey (cf. Tab. 3.1). The cement composition is corrected to 100% by adjusting the oxygen content and a water content of 5% is used. Sand is approximated with data on crustal abundance from the literature survey (cf. Tab. 3.1). The sand composition is corrected to 100% by adjusting the water content.

[31] The correct sum weight of the materials left in to ground is 812'536 kg per km pipeline (see Tab. 5.2). The dataset authors however inventory only an amount of 11'860 kg, representing only 1.5% of the actually mass left in the ground. The 11'860 kg represent the masses for cast iron, polyethylene, steel pipes (= 385.8 + 1650 + 9824). I.e. the authors neglected to heed the amounts of cement, concrete, pitch and sand left in the ground. The correct, much higher sum mass of waste left in the ground should be inventoried in the natural gas pipeline dataset. This will affect the burden of natural gas supply not significantly (ReCiPe LCIA results).

Tab. 5.2 Used materials left in ground from high-pressure natural gas pipeline per kilometre.

	Used pipeline materials left in ground (corrected figures) kg per km	weight distribution
Cast iron	385.785	0.047%
Cement	3900	0.480%
Concrete	3276	0.403%
Pitch/bitumen	300	0.037%
Polyethylene	1650.45	0.203%
Steel pipes	9824	1.209%
Sand	793'200	97.620%
total	812'536.24	100%

Limestone residue from paper production

From pulping mills for paper production stems a limestone-rich a residue (also called lime mud). A composition including traces is derived from (Gaskin et al. 2009, Morris et al. 2000:Tab.2). The resulting composition is corrected to 100% by adjusting the oxygen content. A water content of 5% is used, as specified by the EcoSpold1 waste name.

Slag from metallurgical-grade silicon production

During smelting of silicon metal a silicon-rich slag is produced, which is not very polluted. A composition with traces is derived from data given in (Naess et al. 2014:Fig.10+Tab.II and Jungbluth et al. 2012:22). The resulting composition is corrected to 100% by adjusting the oxygen content. The water content is 0%, as specified by the EcoSpold1 waste name.

Zeolite

During the refining of oil catalysts with various metals (Co, Mo, V, Ni, W, platinum group metals) are used. The catalyst metals are on a carrier matrix, usually zeolite. Spent catalysts are recycled for their metal content and the zeolite is landfilled. A composition of the zeolite is derived from (Vaiciukyniene et al. 2020:Tab.1, Gao & Owens 2012: Tab.1,2). The resulting composition is corrected to 100% by adjusting the oxygen content. A water content of 5% is used, as specified by the EcoSpold1 waste name.

Gravel (new dataset)

If a specific concrete composition is landfilled, it is possible to inventory this as two separate masses of cement and gravel. Composition of gravel is assumed to be natural rock and is approximated with crustal concentrations compiled in the literature survey (cf. Tab. 3.1). This disposal can also be used for landfilled loose gravel, sand or natural rock. However, if unpolluted it is more likely that this material is recycled into new building products.

Cement (new dataset)

If a specific concrete composition is landfilled, it is possible to inventory this as two separate masses of cement and gravel. The cement composition is taken from the literature survey (cf. Tab. 3.1). This waste fraction might also be a good approximation for fine fractions from building waste crushers landfilled in a construction waste landfill.

Excavation material (new dataset)

Usually excavation material is recycled if possible, which is the default assumption in ecoinvent inventories. But separate landfills for this material exist in the real world (VVEA type A) and a separate model for this type of landfill was created in this project. The average excavation material composition is taken from the literature survey (cf. Tab. 3.1). This dataset might be useful for sensitivity analysis.

Further waste compositions

For all the remaining waste materials, waste compositions are already researched in Doka 2003. (emulsion paint, glass, natural gypsum, rock wool, paperboard, paint remains, plastic plaster, polyurethane, steel). Those compositions were used without alterations.

6 Results

6.1 Results for construction waste landfill

Seventeen different disposal datasets for waste in a construction waste landfill are created. A Swiss climate (1000 mm precipitation, 500 mm evaporation) and 11 m landfill height are assumed. The LCIA results of those disposals are shown in Fig. 6.1.[32] The results are grouped into building materials, mostly biogenic materials (cardboard), inorganic bulk materials, materials with synthetic polymers, and various process-specific waste materials. The datasets for cardboard and plastics are meant for parts of materials enclosed in other materials, e.g. cardboard fibres in gypsum, or PU foam on brickwork.

As expected the waste-specific emissions give rise to clear differences in the burdens from the disposal of these waste materials. The least burdening materials have burdens only little over the infrastructure and processing burdens (which are indicated by a small horizontal line in Fig. 6.1). The more burdening materials can in contrast have one or two orders of magnitude more burdens. This underlines the importance in being able to have differentiation between materials by heeding their actual composition, and not having identical burdens for waste in inert material landfills as was the case previously.

The contributions to the LCIA burdens is shown in Fig. 6.2. Contributions are from infrastructure & processing, from all short-term emissions, from three particular long-term emissions (arsenic, barium, manganese), and from all remaining long-term emissions. Arsenic, barium, and manganese show to be relevant for a number of different waste materials. Lead is of large relative importance for paint remains, and of some importance for plastic plaster, polyurethane and glass. Zeolite is special in as its

[32] 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.

burdens come from vanadium and nickel emissions, which are amongst the catalyst metals the waste zeolite was a carrier matrix for.

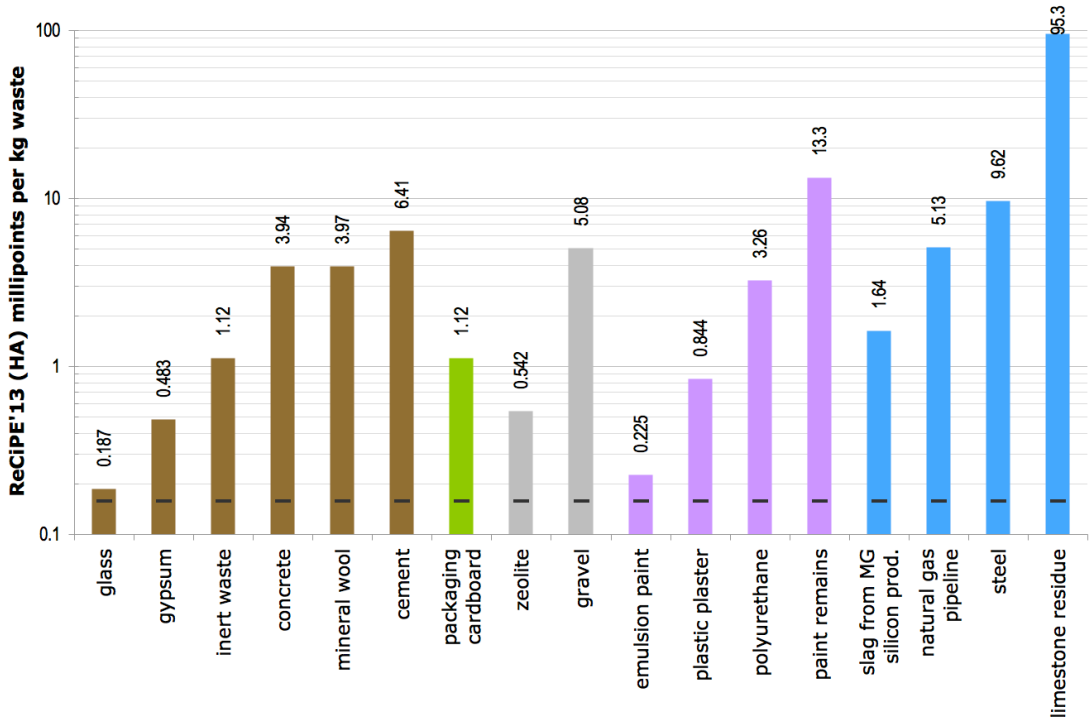

Fig. 6.1 **LCIA results for 17 disposal datasets in construction waste landfill in ascending order and grouped into building materials (brown), mostly biomass materials (green), inorganic bulk materials (grey), materials with synthetic polymers (purple), and various process-specific waste materials (blue). The small horizontal line at 0.157 millipoints represents the constant burden from infrastructure and processing. The column above that mark therefore represents the burdens from the waste-specific direct landfill emissions to water.**

The short-term emissions (0–100 years) end up to be negligible contributions in all datasets (blue bar in Fig. 6.2). In contrast, the long-term emissions (100–60'000 years) are usually the dominant burdens, except in the least burdening processes, like glass or emulsion paint, 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.[33]

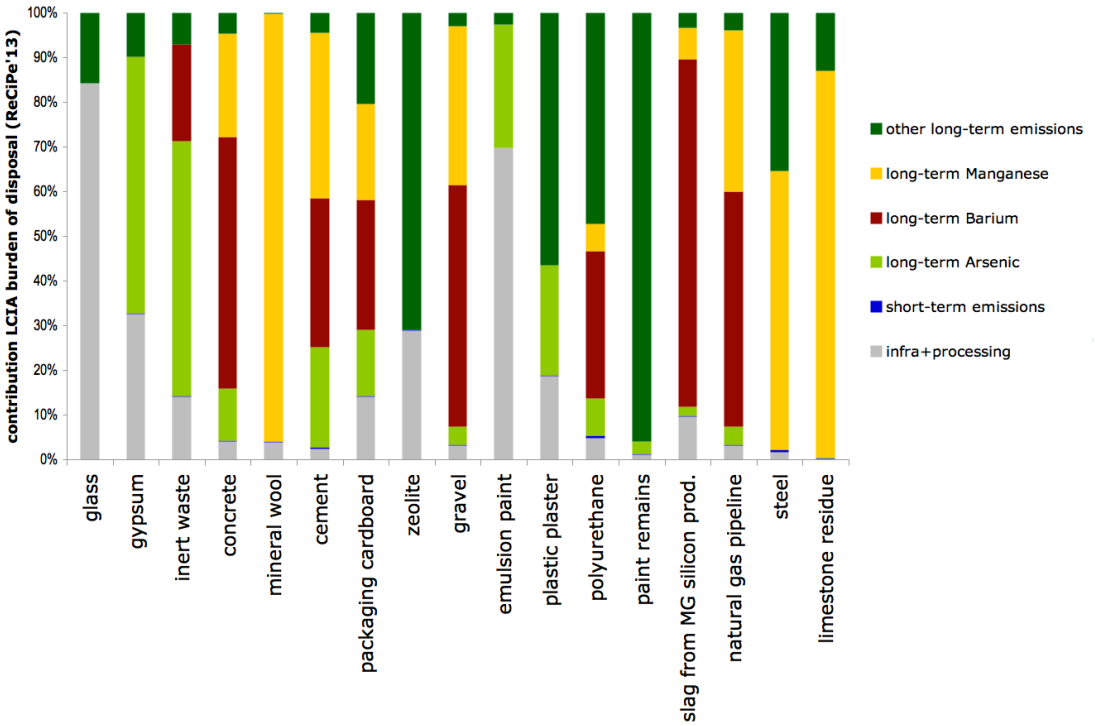

Fig. 6.2 Contribution analysis of the LCIA results for 17 disposal datasets in construction waste landfill. In the same ordering sequence as in the previous chart.

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 17 waste materials, typically only 60%±15% of the total landfilled waste mass is emitted in the long-term. The remainder 40% mass stays in the landfill even after 60'000 years. The emitted mass typically represents only 40%±20% of the toxicity potential of a waste material.[34] The established burdens therefore do not represent extremes or worst cases – which would be 100% emission. The various landfill models in (Doka 2017) 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.

6.1 Results for excavation landfill

For the time being only one single dataset for disposal in an excavation landfill was created. An average composition of clean excavation material was used (cf. page 40). Per kilogram an LCIA

[33] 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).

[34] 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.

burden of 4.44 millipoints (ReCiPe'13 HA) is calculated. In comparison, this is in the range of values found in construction waste landfills in Fig. 6.2. The main differences are larger emissions from manganese and smaller emissions from arsenic; both those emissions make up the majority of the burden of excavation material. Although manganese occurs only in traces in the waste (360 ppm) and this is similar to the content in average building waste (337 ppm), it is more mobile in an excavation landfill than it would be in an construction waste landfill. Arsenic in contrast is less mobile in an excavation landfill. The carbonate-buffered phase is shorter in the excavation landfill (8680 years) than in the construction waste landfill (>60'000 years). The shorter carbonate phase and the subsequent drop in pH increases the mobility of manganese and decreases the mobility of arsenic compared to a construction waste landfill.

6.2 Differences of landfill models for identical waste materials

To illustrate the differences in landfill modelling and the value of a specific inert material landfill model, identical waste materials in different landfills can be compared. As an example a **cement-fibre slab** (consisting of wood and cement) into a Swiss construction waste landfill is compared with the same material in a sanitary landfill. As a second example, a disposal of **gypsum** in both those landfills is shown.

Tab. 6.1 LCIA results for the disposal of 1kg cement-fibre slab or gypsum in a Swiss construction waste landfill and a sanitary landfill.

Waste material	Disposal	LCIA score per kilogram In ReCiPe'13 millipoints	ratio sanitary / construction waste
cement-fibre slab	construction waste landfill	0.9	
cement-fibre slab	sanitary landfill	3.88	4
gypsum board	construction waste landfill	0.483	
gypsum board	sanitary landfill	27.2	56

The disposal in a construction waste landfill has significantly lower LCIA scores than in a sanitary landfill, here a factor 4 and 56, respectively. This is caused on one hand by the larger infrastructure expenditures for a sanitary landfill, and on the other hand by the different pollutant mobility behaviour in the landfills. The emission burden for cement-fibre slab is dominated by emissions of arsenic and manganese, and in the construction waste landfill only 68% and 23% of the landfilled amount of those elements are released, while in the sanitary landfill either release is 100%.[35] For gypsum in the construction waste landfill, the waste-specific emissions are dominated by arsenic and to a lesser degree mercury. In the sanitary landfill however the emissions are dominated by sulfur dioxide SO_2 emissions to air which come from the conversion of a small part (6.5%) of the sulfate in gypsum to hydrogen sulfide H_2S in the anaerobic conditions of the sanitary landfill and its subsequent oxidation to SO_2 in the landfill gas combustion, flare, or atmosphere. These results exemplify the importance of applying a specific landfill model, since even identical materials can have very different behaviour and fates in different environments.

[35] These transfer coefficients are for the assumed Swiss climate only. In other conditions other TKs can result.

6.3 Relevance of construction waste landfills

How do the LCIA results of the ecoinvent database change with these new landfill models? This can at the moment not yet be answered precisely, as the new landfilling datasets are not yet included in ecoinvent. But an approximate answer can be given by looking at the inventoried waste disposed in inert material landfills. The cumulated amount of inert waste can be calculated from the cumulated land use transformation to the area type "dump site, inert material landfill": per kilogram of inert waste a land transformation of $4.44 \cdot 10^{-5}$ m^2 is inventoried. From this relation, the cumulated amount of inert waste per dataset can be calculated. By assuming this waste is mostly waste concrete, the changes in LCIA results that would occur, if the new dataset including landfill emissions were included, can be estimated.

The outcome of this analysis for the ecoinvent v3.6 database (Sept 2019, cut-off, model) that for the majority of the 18'121 datasets the increase in burden is negligible – typically below 0.1% (with ReCiPe'13(HA)). So very often for many datasets the emissions from inert material landfills do not play a relevant role, which is an expected result. On the other hand for some datasets the emissions from inert material landfills do become relevant. Naturally, the inert landfill disposal processes themselves are affected and can exhibit burden increases of factors 3 to 5. But also infrastructure processes can be affected and for instance a composting facility infrastructure exhibits a burden increase by 36%. Also electricity transmission infrastructure is significantly affected with burden increases of around 24%. But also products and goods themselves can be affected: for instance sodium nitrate product ($NaNO_3$) has a burden increase of 43%; feldspar production has an increase of 11%; or lithium carbonate ($LiCO_3$) has an increase of 4%.[36] Electricity production from alpine hydropower reservoirs increases by 4.4%, for run-of-river plants by 3.5%.

For these processes, and several others, the emissions from inert material landfills become perceptible and conversely their burdens were underestimated until now. With the new models, the burdens can be included and differentiated according to the specific waste material. So for these processes the new models are of consequence.

7 Glossary

Cleanfill — A landfill receiving relatively unpolluted and unproblematic waste like clean excavation material. This excludes e.g. landfills for common municipal waste, putrescibles, polluted industrial wastes. Nomenclature in waste disposal is frequently not unanimous and some use the term 'cleanfill' not for the disposal facility, but for the deposited *material*. Also some use cleanfill to mean landfills for excavation material including also building material waste. In this report the term cleanfill is generally used for excavation material landfills (Type A).

Inert material landfills — Summary term for landfills receiving excavation material and/or mineral building material waste. In the current Swiss Waste Ordinance of 2016, this would be type A and type B landfills. In the former Swiss waste ordinance TVA of 1990 these two ware not distinguished and the term inert material landfill (Ger. *Inertstoffdeponie*) was used for either. In this report the term will be used to refer both types of landfills collectively.

[36] For sodium nitrate and lithium carbonate productions the inert waste is likely misnomeric: the wastes are likely to be very *soluble* materials, and as such not be materials that would be allowed in an inert material landfill.

C&D waste	(also CDW, or building waste) Construction and demolition waste. When unsorted this can include metals and burnable materials like wood and plastics. When sorted this will mainly consist of inorganic materials like concrete, bricks, gypsum, pane glass and can also include excavation material.
Inert material	Inert waste materials are usually inorganic wastes with a relatively low pollutant content, like excavation subsoil, or waste concrete, bricks, stones, glass etc. The term 'inert' must be seen in contrast to more reactive waste materials like food waste, that starts to decompose quickly and produce emissions to air and water, or solvent waste, that evaporates, leads to air emissions and is potentially ignitable, or hazardous or corrosive wastes that are acutely detrimental and reactive, or other combustible or explosive wastes. In contrast to such *reactive* waste materials, inert wastes are much less reactive and require much less safety measures for their transport, handling and storage. They are merely *comparatively slow* to react and transform. The term 'inert' does not mean they are not reactive at all, but are relatively sluggish to do so.
Type A landfill	In the current Swiss Waste Ordinance of 2016, a type A landfill is receiving mostly clean excavation material that could not be reused or recycled (VVEA 2016). This can correspond to excavation material landfills or cleanfills in other countries. Prior to 2016 the excavation material landfills were lumped together with construction waste landfills (new Type B) as "inert waste material landfills" in the former Swiss waste ordinance TVA of 1990.
Type B landfill	In the current Swiss Waste Ordinance of 2016, a type B landfill is receiving mostly inorganic construction waste material and polluted excavation material that could not be reused or recycled (VVEA 2016). This can correspond to C&D waste landfills in other countries, when this excludes large masses of burnable waste materials, like wood. Prior to 2016 the construction waste landfills were lumped together with excavation material landfills (new Type A) as "inert waste material landfills" in the former Swiss waste ordinance TVA of 1990.
ES1	EcoSpold1 (sometimes also named "EcoSpold 2000"). A file format for Life Cycle Assessment data. Within the context of this project, ES1-files refer to process inventory files. EcoSpold1 files were defined and used in the ecoinvent 2000 project (2003-2010) which released ecoinvent versions 1.0-2.2. Later ecoinvent versions (3+) used Ecospold2 (ES2). After 2010, non-canonical versions or dialects of the EcoSpold1 format (i.e. not strictly adhering to the original ecoinvent nomenclature) were created by various consulting firms (e.g. Pre, ESU-services, treeze).
ES2	EcoSpold2. A file format for Life Cycle Assessment data. Within the context of this project, ES2-files refer to process inventory files. ES2 is the updated format of EcoSpold1 (ES1).

8 References

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

AWEL 2008 AWEL (2008) Richtplan Teilrevision Ver- und Entsorgung - Deponiestandorte Fakten, Argumente. AWEL Amt für Abfall, Wasser, Energie und Luft, Sektion Abfallwirtschaft. Aug 2008 https://awel.zh.ch/content/dam/baudirektion/awel/abfall_rohstoffe_altlasten/abfall/dokumente/verschiedenes_spezialthemen/deponiestandorte_fakten_und_argumente_2008.pdf (17 Nov 2018)

AWEL 2013 AWEL (2013) Standortspezifische Einleitbedingungen für Sickerwässer aus Deponien. By GEO Partner AG, Zürich, commissioned by AWEL Amt für Abfall, Wasser, Energie und Luft, Sektion Abfallwirtschaft. Feb 2013. http://www.awel.zh.ch/internet/baudirektion/awel/de/betriebe_anlagen_baustellen/abfallanlagen/deponien/_jcr_content/contentPar/downloadlist_1/downloaditems/standortspezifische_.spooler.download.1368610863828.pdf/einleitbedingungen_sickerwasser_deponien_2013.pdf (17 Mar 2016)

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 2016a AWEL (2016a) Proben von unverschmutztem Aushubmaterial Gesammelt im Kanton Zürich ab 2014 - Aushubproben Details. Stand der Daten: Mai 2016. Amt für Abfall, Wasser, Energie und Luft (AWEL) Baudirektion Kanton Zürich), Abteilung Abfallwirtschaft & Betriebe, Sektion Abfallwirtschaft. http://www.tinuschneider.ch/Software/charts_aushubproben/ → Details (19 Nov 2019)

AWEL 2016b AWEL (2016b) Proben von unverschmutztem Aushubmaterial Gesammelt im Kanton Zürich ab 2014 - Aushubproben Übersicht. Stand der Daten: Mai 2016. Amt für Abfall, Wasser, Energie und Luft (AWEL) Baudirektion Kanton Zürich), Abteilung Abfallwirtschaft & Betriebe, Sektion Abfallwirtschaft. http://www.tinuschneider.ch/Software/charts_aushubproben/ → Übersicht (19 Nov 2019)

AWEL 2016c 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 2017 AWEL (2017) "Deponiestatistik 2016". Amt für Abfall, Wasser, Energie und Luft (AWEL), Baudirektion Kanton Zürich, Sept 2017. https://awel.zh.ch/internet/baudirektion/awel/de/betriebe_anlagen_baustellen/abfallanlagen/deponien/_jcr_content/contentPar/downloadlist_2/downloaditems/bericht_zur_deponies.spooler.download.1461750991824.pdf/deponiestatistik_2015.pdf (17 Nov 2018)

AWEL 2018 AWEL (2018) "Deponiestatistik 2017". Amt für Abfall, Wasser, Energie und Luft (AWEL), Baudirektion Kanton Zürich, Nov 2018. https://awel.zh.ch/content/dam/baudirektion/awel/abfall_rohstoffe_altlasten/abfall/dokumente/statistiken/deponie_statistiken/deponiestatistik_2017.pdf (15 Okt 2019)

AWEL 2018a AWEL (2018a) Daten aus Deponiestatistik Kanton Zürich - Auswertung nach Standorten. 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/**googlechartsstandorte**/index.html via https://awel.zh.ch/internet/baudirektion/awel/de/abfall_rohstoffe_altlasten/zahlen_und_fakten/auswertung_3spaltig.html in tab → "Auswertung nach Standorten" (19 Nov 2019)

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)

Ayotte et al. 2011 Ayotte JD, Gronberg JM, Apodaca LE (2011) "Trace-Elements and Radon in Groundwater Across the United States, 1992-2003". U.S. Geological Survey Scientific Investigations Report 2011–5059, National Water-Quality Assessment (NAWQA), U.S. Geological Survey , U.S. Department of the Interior http://water.usgs.gov/nawqa/trace/pubs/sir2011-5059/ (31 Oct 2016)

Beesley & Dickinson 2011 Beesley L, Dickinson N (2011) Carbon and trace element fluxes in the pore water of an urban soil following greenwaste compost, woody and biochar amendments, inoculated with the earthworm Lumbricus terrestris. Soil Biology and Biochemistry, Volume 43, Issue 1, January 2011, p. 188-196. https://www.sciencedirect.com/science/article/abs/pii/S0038071710003688 (6 Oct 2019)

Bradford et al. 1996 Bradford GR, Change AC, Page AL, Bakhtar D, Frampton JA, Wright H (1996) "Background Concentrations of Trace and Major Elements in California Soils". University of California, Kearney Foundation of Soil Science, Division of Agriculture and Natural Resources, March 1996. http://www.waterboards.ca.gov/water_issues/programs/compost/docs/kearney1996.pdf (31 Oct 2016)

Burkhardt et al. 2015 Burkhardt M, Dietschweiler C, Schmidt S, Hemmann J, Sinnet B, Kägi R (2015) "Nanopartikel im Sickerwasser von Deponien - Erfassung von Titandioxid-Weisspigmenten als Leitsubstanz für Nanopartikel". HSR Hochschule für Technik Rapperswil, Institut für Umwelt- und Verfahrenstechnik (UMTEC) und EAWAG, Dübendorf. Bericht für das AWEL Amt für Abfall, Wasser, Energie und Luft, Zürich. Feb 2015.

	https://www.umtec.ch/fileadmin/user_upload/umtec.hsr.ch/Projekte/150210_Bericht_Nanodepo.pdf (20 Sept 2018)
Butera et al. 2014	Butera S, Christensen TH, Astrup TF (2014) "Composition and leaching of construction and demolition waste: inorganic elements and organic compounds". Journal of Hazardous Materials, 276, 302–311. doi:10.1016/j.jhazmat.2014.05.033 https://www.academia.edu/download/41625664/Composition_and_leaching_of_construction20160127-18528-hfxjcj.pdf (2 Feb 2020)
BUWAL 1996	Habersatter K, Fecker I, Dall'Aqua S, Fawer M, Fallscher F, Förster R, Maillefer C, Ménard M, Reusser L, Som C, Stahel U, Zimmermann P (1996) Ökoinventare für Verpackungen. Band I + II. Bundesamt für Umwelt Wald und Landschaft (BUWAL, Swiss EPA) (Hrsg.): Schriftenreihe Umwelt SRU Nr. 250/I + II. Berne 1996. http://www.buwal.ch/publikat/oekobila.htm (30 May 2001)
Dagan 2017	Dagan J (2017) Ermittlung relevanter Messparameter für die Sickerwasserbeprobung von Typ B Deponien. Bachelorarbeit. Zürcher Hochschule für Angewandte Wissenschaften ZHAW, Wädenswil. https://www.ar.ch/fileadmin/user_upload/Departement_Bau_Volkswirtschaft/Amt_fuer_Umwelt/Umwelt/Publikationen/Merkblaetter/Abfall/BA-Arbeit_Dagan_Sickerwasserbeprobung_Deponie-TypB_2017.pdf (6 Oct 2019)
Dettli et al. 2004	Dettli R, Steiner P, Baur M, Müller M (2004) Kosten und Entschädigung von Strom aus Kehrichtverbrennungsanlagen. Bundesamt für Energie BFE. Forschungsprogramm Energiewirtschaftliche Grundlagen. https://www.econcept.ch/de/projekte/kosten-und-entschadigung-von-strom-aus-kehrichtverbrennungsanlagen/ (4 Jul 2006)
Doka 2003-I	Doka G (2003) Life Cycle Inventories of Waste Treatment Services, part I: General Introduction Waste Material Compositions Municipal Waste Collection. ecoinvent report No. 13, part I. Swiss Centre for Life Cycle Inventories, Dübendorf, 2003. http://www.doka.ch/13_I_WasteTreatmentGeneral.pdf (15 Dec 2003)
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, Dübendorf, 2003. http://www.doka.ch/13_II_WasteIncineration.pdf
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 2003-IV	Doka G (2003) Life Cycle Inventories of Waste Treatment Services, part IV: Wastewater treatment. ecoinvent report No. 13, part IV. Swiss Centre for Life Cycle Inventories, Dübendorf, 2003. http://www.doka.ch/13_IV_WastewaterTreatment.pdf
Doka 2008a	Doka G (2008) Life Cycle Inventory data of mining waste: Emissions from sulfidic tailings disposal. For BAFU/FOEN (Swiss EPA), April 2008. http://www.doka.ch/SulfidicTailingsDisposalDoka.pdf (3 Apr 2008)
Doka 2008b	Doka G. (2008) Non-radiological emissions from uranium tailings: A generic, global model for Life Cycle Inventory data. Doka Life Cycle Assessments, Zurich, November 2008. http://www.doka.ch/PSIuraniumtailingsDoka.pdf (19 Feb 2009)
Doka 2009	Doka G. (2009) Life Cycle Inventory of the disposal of lignite spoil, coal spoil and coal tailings. Doka Life Cycle Assessments, Zurich, For the Paul Scherrer Institute PSI, Nov 2009. http://www.doka.ch/DokaCoalTailings.pdf (10 Apr 2010)
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	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	Doka G (2020) Calculation manual for LCI calculation tools for regionalised waste treatment. Doka Life Cycle Assessments, Zurich, Switzerland. July 2020. Available at http://www.doka.ch/publications.htm
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.
eco-bau 2020	eco-bau (2020) Methodik Baumaterialien eco-bau – Bewertung von Baumaterialien und Bauprodukten nach ökologischen Kriterien – Methodik Beschrieb und Anhang 1. Verein eco-bau, Zürich. https://www.eco-bau.ch/resources/uploads/Methodik_eco-bau/2020/Methodik_eco-bau_Bericht%2BAnhang%201_01_2020.pdf (25 Feb 2020)
Ecoinvent 2002	Ecoinvent (2002) EcoSpold01 Schema Documentation - EcoSpold01Dataset_Simple.html. Swiss Centre for Life Cycle Inventories. Version of 29. Oktober 2002 http://www.ecoinvent.org/download/EcoSpoldSchema_v1.0.zip (29 Oct 2008) and identical at https://www.ecoinvent.org/files/ecospold1_data_exchange_format.zip (1 March 2016, 27 Apr 2020)
Engelsen et al. 2010	Engelsen CJ, van der Sloot HA, Wibetoe G, Justnes H, Lund W, Stoltenberg-Hansson E (2010) Leaching characterisation and geochemical modelling of minor and trace elements released from recycled concrete aggregates. Cement and Concrete Research 40 (2010) 1639–1649. https://publications.tno.nl/publication/34633439/6W38Vt/w10044.pdf (4 Feb 2020)
EPA 1995	EPA (1995) Construction and Demolition Waste Landfills. Prepared for U.S. Environmental Protection Agency Office of Solid Waste by ICF Incorporated Contract No. 68-W3-0008, February 1995.

	http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.566.2617&rep=rep1&type=pdf (19 Nov 2018) also https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=9101Q651.TXT (19 Nov 2018)
Frischknecht et al. 1996	Frischknecht R., Bollens U., Bosshart S., Ciot M., Ciseri L., Doka G., Dones R.,Gantner U., Hischier R. and Martin A. (1996): Ökoinventare von Energiesystemen.ETH/PSI, Zürich/Villigen.
Gallert 2000	Gallert I (2000) Qualität des Grundwassers im Kanton Basel-Stadt Stand 1999 Basel, den 16. Juni 2000 AMT FÜR UMWELT UND ENERGIE BASEL-STADT Abteilung Labor und Rheinüberwachungsstation. Download of 5.7.04 from http://www.aue-bs.ch/de/gewaesser/_pdf/grundwasser/QualiGW1999alles.pdf (5 Jul 2004)
Gao & Owens 2012	Gao X, Owens WT (2012) Process for Metal Recovery from Catalyst Waste. UnitedStates Patent Application Publication US2012/0156116A1, 21 Jun 2012. https://patentimages.storage.googleapis.com/e8/01/b2/b6a75815787c9a/US20120156116A1.pdf (22 Jun 2020)
Gaskin et al. 2009	Gaskin J, Miller W, Morris LA (2009) Land application of pulp mill lime mud. University of Georgia, College of Agriculture and Environmental Sciences, Cooperative Extension Service, Bulletin 1249, Reviewed May 2009. Georgia , USA. https://www.researchgate.net/publication/277772121_Land_application_of_pulp_mill_lime_mud (22 Jun 2020)
Gimmi & Waber 2004	Gimmi T, Waber HN (2004) Modelling of tracer profiles in pore water of argillaceous rocks in the Benken Borehole: Stable water isotopes, chloride and chlorine isotopes. National Cooperative for the Disposal of Radioactive Waste (Nagra), WEttingen, Switzerland. NAGRA Technical Report 04-05. Dec 2004. https://www.nagra.ch/data/documents/database/dokumente/$default/Default%20Folder/Publikationen/NTBs%202001-2010/e_ntb04-05.pdf (9 Oct 2019)
Graf 1998	Dr. Graf AG (1998) Regelungen über die Abfallentsorgung in Zementwerken - Grundlagen, Methoden, Berechnungen. DR. GRAF AG Gerlafingen, Switzerland. Report No. 91.2018/A. Comissioned by Bundesamt für Umwelt, Wald und Landschaft, BUWAL July 1998. Download of 14.April 2004 from http://www.umwelt-schweiz.ch/imperia/md/content/abfall/zw_abb_s11.pdf and http://www.umwelt-schweiz.ch/buwal/de/fachgebiete/fg_abfall/zahlen/recht/index.html
Grauwiler 1992	Grauwiler M. (1992) Redoxpotential – einfacher Indikator für Zustand von Inertdeponien. Abfall Nr.1/92, p.23
Haxel et al. 2002	Haxel GB, Hedrick JB, Orris GJ (2002) Rare Earth Elements—Critical Resources for High Technology. USGS Fact Sheet 087-02, U.S. Geological Survey, U.S. Department of the Interior, 2002. http://pubs.usgs.gov/fs/2002/fs087-02/fs087-02.pdf (27 Jun 2002) http://web.archive.org/web/20080627153421/pubs.usgs.gov/fs/2002/fs087-02/fs087-02.pdf
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
Holcim 2010	Holcim (2000) "Umweltschutz in der Baustoffproduktion: Ziele, Massnahmen, Erfolge Umweltbericht HCB "Holderbank" Cement und Beton mit den Daten der Jahre 1996 bis 1998". Datiert 18. April 2000. Download vom 27 April 2006 von http://www.holcim.com/Upload/CH/Publications/de_media_press_publications_umweltbericht_1.pd (2020 unter https://www.yumpu.com/de/document/read/30816445/umweltschutz-in-der-baustoffproduktion-ziele-massnahmen-erfolge)
IAEA 2010	IAEA (2010) Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments. Technical Reports Series No. 472, ISBN 978–92–0–113009–9, International Atomic Energy Agency IAEA, Vienna, 2010. http://www-pub.iaea.org/MTCD/publications/PDF/trs472_web.pdf (28. Nov 2016)
IPCC 2006-3	Intergovernmental Panel on Climate Change (IPCC) 2006. IPCC Guidelines for National Greenhouse Gas Inventories – Volume 3 Industrial Processes and Product Use. Agyemang-Bonsu WK, Harnisch J (coordinating lead authors) and 86 international authors. Intergovernmental Panel on Climate Change (IPCC), IPCC/IGES, Hayama, Japan. https://www.ipcc-nggip.iges.or.jp/public/2006gl/vol3.html (2 Jun 2020)
IPCC 2019-3	Intergovernmental Panel on Climate Change (IPCC) 2006. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories – Volume 3 Industrial Processes and Product Use. Cai B, Ottinger DA (authors) ntergovernmental Panel on Climate Change (IPCC), IPCC/IGES, Hayama, Japan https://www.ipcc-nggip.iges.or.jp/public/2019rf/vol3.html (2 Jun 2020)
Jungbluth et al. 2012	Jungbluth N, Stucki M, Flury K, Frischknecht R, Büsser S (2012) Life Cycle Inventories of Photovoltaics – version 2012. ESU-services, Uster, comissioned by Swiss Federal Office of Energy SFOE. 5 Sept 2012. http://treeze.ch/fileadmin/user_upload/downloads/PublicLCI/jungbluth-2012-LCI-Photovoltaics.pdf (22 Sept 2014)
KBOB et al. 2018	KBOB (2018) KBOB-Ökobilanzdatenbestand 2016 und UVEK-Ökobilanzdatenbestand 2018. Koordinationskonferenz der Bau- und Liegenschaftsorgane der öffentlichen Bauherren KBOB, und eco-bau Nachhaltigkeit im öffentlichen Bau, and IPB Interessengemeinschaft Privater Bauherren, 7 Nov 2018. http://www.dflca.ch/inventories/Kommunikation_Oekobilanzdatenbestand_KBOB-2016_UVEK-2018.pdf (25 Nov 2018) see databases at https://db.ecoinvent.org/download/KBOB%20DQRv2_2016.zip?area=3e2c0806caa3c (25 Nov 2018) and

https://db.ecoinvent.org/download/UVEK%20LCI%20Data_2018_20191003.zip?area=3e2c0806caa3c (2 Mai 2020)

Kellenberger 2020 Personal Communication with Daniel Kellenberger, senior manager at Intep Integrale Planung GmbH, Zürich, 25. Feb 2020

Kerndorff et al. 1993 Kerndorff, H., R. Schleyer and H. H. Dieter (1993): Bewertung der Grundwassergefährdung von Altablagerungen - Standardisierte Methoden und Maßstäbe. Berlin, Institut für Wasser-, Boden- und Lufthygiene des Bundesgesundheitsamtes: 145. Cited on p.91 of Laner D (2011) Understanding and evaluating long-term environmental risks from landfills. Dissertation TU Vienna.

Kikuchi et al. 2011 Kikuchi T, Kuroda Y (2011) Carbon dioxide uptake in demolished crushed concrete. Journal of Advanced Concrete Technology. Vol. 9, No. 1, 115-124, February 2011. https://www.jstage.jst.go.jp/article/jact/9/1/9_1_115/_pdf (26 Feb 2020)

Kolbas et al. 2018 Kolbas A, Kolbas N, Marchand L, Herzig R, Mench M (2018) Morphological and functional responses of a metal-tolerant sunflower mutant line to a copper-contaminated soil series. Environmental Science and Pollution Research, April 2018, DOI: 10.1007/s11356-018-1837-1. https://www.researchgate.net/profile/M_Mench/publication/324181642_Morphological_and_functional_responses_of_a_metal-tolerant_sunflower_mutant_line_to_a_copper-contaminated_soil_series/links/5af17138458515c283754f17/Morphological-and-functional-responses-of-a-metal-tolerant-sunflower-mutant-line-to-a-copper-contaminated-soil-series.pdf?origin=publication_detail (21 Nov 2019)

Laner 2011 Laner D (2011) Understanding and evaluating long-term environmental risks from landfills. Dissertation Vienna University of Technology. Technische Universität Wien, Fakultät für Bauingenieurwesen. Juni 2011 http://www.wien.gv.at/umweltschutz/nachhaltigkeit/pdf/laner.pdf (11 Jan 2012)

Langmuir et al. 2004 Langmuir D, Chrostowski P, Vigneault B, and Rufus Chaney R (2004) Issue Paper on the Environmental Chemistry of Metals. Submitted to U.S. Environmental Protection Agency Risk Assessment Forum. 19 Aug 2004 https://www.researchgate.net/profile/Rufus_Chaney3/publication/237330649_ISSUE_PAPER_ON_THE_ENVIRONMENTAL_CHEMISTRY_OF_METALS/links/00b49530b3de040dd3000000.pdf (31 Oct 2016)

Lechner & Dreier 1995 Lechner P, Dreier p (1995) Baurestmassen Vermeidung, Verwertung, Behandlung. Report UBA-95-110, Umweltbundesamt, Wien, August 1995. https://www.umweltbundesamt.at/aktuell/publikationen/publikationssuche/publikationsdetail/?pub_id=703 https://www.umweltbundesamt.at/fileadmin/site/publikationen/R110.pdf (20 Sept 2018)

Leeman & Loser 2016 Leeman A, Loser R (2016) Karbonatisierungswiderstand von Recyclingbeton. Co-financed by cemsuisse, association of the Swiss cement manufacturers, cemsuisse-project 201405. EMPA, Swiss Federal Laboratories for Materials Science and Technology, Dübendorf, Switzerand, and Cemsuisse, Swiss Cement Industry Association, Berne, Switzerland. Nov 2016. www.cemsuisse.ch/cemsuisse/forschung/forschungsberichte/berichte_2016/index.html?lang=de (24 Feb 2020)

Leemann & Hunkeler 2016 Leemann A, Hunkeler F (2016) Carbonation of concrete: assessing the CO_2-uptake. Co-financed by cemsuisse, association of the Swiss cement manufacturers, cemsuisse-project 201602. Cemsuisse, Swiss Cement Industry Association, Berne, Switzerland. 12 Aug 2016. http://www.cemsuisse.ch/cemsuisse/forschung/forschungsberichte/karbonatisierung/index.html?lang=de or www.cemsuisse.ch/cemsuisse/forschung/forschungsberichte/berichte_2016/index.html?lang=de (24 Feb 2020)

McLean & Bledsoe 1992 McLean J.E., Bledsoe B.E. (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. October 1992. http://www.earthwardconsulting.com/library/EPA_arsenic_in_soil_issue14.pdf (10 Oct 2002)

Morris et al. 2000 Morris LA, Nutter WL, Miller WP, Overcash M (2009) Treatment and use of pulp and paper and textile industry residues in southern U.S. forests. In: Henry CL, Harrison RB, Bastian RK (eds) The Forest Alternative Principles and Practice of Residuals Reuse. College of Forest Resources, University of Washington, Seattle, USA. https://www.researchgate.net/publication/284290788_Treatment_and_use_of_pulp_and_paper_and_textile_industry_residues_in_southern_US_forests (22 Jun 2020)

NABO 2017 NABO (2017) Schwermetallbelastung des Bodens (0-20 cm). Nationale Bodenbeobachtung NABO, Datenstand 31.12.2009. 4- Dez 2017. Bundesamt für Umwelt BAFU. https://www.ubst.bafu.admin.ch/opendata/UBD0066.ods (6 Oct 2019)

Naess et al. 2014 Naess MK, Kero I, Tranell G, Tang K, Tveit H (2014) Element Distribution in Silicon Refining: Thermodynamic Model and Industrial Measurements. JOM - The Journal of The Minerals, Metals & Materials Society (TMS), Vol. 66 (2014), No. 11, 2343–2354 https://link.springer.com/content/pdf/10.1007/s11837-013-0797-7.pdf (23 Jun 2020)

NAQUA 2019 NAQUA (2019) Zustand und Entwicklung Grundwasser Schweiz Ergebnisse der Nationalen Grundwasserbeobachtung NAQUA, Stand 2016. Bundesamt für Umwelt BAFU, Eidg. Departement für Umwelt, Verkehr, Energie und Kommunikation (UVEK), Bern, 2019. https://www.bafu.admin.ch/dam/bafu/de/dokumente/wasser/uz-umwelt-zustand/zustandundentwicklunggrundwasserschweiz.pdf.download.pdf/UZ-1901-D_NAQUA.pdf (22 Nov 2019)

ÖBU 2008 Frischknecht R, Büsser Knöpfel S, Flury K, Stucki M (2013) Ökofaktoren Schweiz 2013 gemäss der Methode der ökologischen Knappheit – Methodische Grundlagen und Anwendung auf die Schweiz. BAFU Reihe Umwelt-Wissen, Nr. UW-1330, Bundesamt für Umwelt BAFU, ÖBU – Netzwerk für nachhaltiges Wirtschaften Bern, 2013. http://www.oebu.ch/themen/nachhaltigkeit-in-unternehmen/umwelt/oekofaktoren-schweiz-2013-nun-erhaeltlich.html (14 Mar 2014) also at www.bafu.admin.ch/uw-1330-d

Pladerer et al. 2004 — Pladerer C, Ganglberger E, Funder B, Roiser-Bezan G (2004) Vermeidung von Baustellenabfällen in Wien - Erarbeitung von Vermeidungsstrategien für Baustellenabfälle basierend auf einer praktischen Durchführung der Probenahme, Nachsortierung und einer analytischen Untersuchung von Baustellenabfällen und einzelner Fraktionen in Wien - Endbericht Teil 3/4: Ergebnisse. Österreichisches Ökologie-Institut, Wien, Mai 2004. http://www.ecology.at/files/berichte/E10.760-3.pdf (30 Jan 2020)

Pratt 2018 — Pratt S (2018) Assessment of Ecological Effects Envirofert Ltd - Appendix F: Leachate monitoring results and loads. Tonkin & Taylor Ltd, Auckland, New Zealand, December 2018. https://www.waikatoregion.govt.nz/assets/WRC/Community/whats-happening/have-your-say/Envirofert/TandT-Eco-effects-appendices.pdf (10 May 2019)

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)

Rubli et al. 2017 — Rubli S, Brupbacher A, Rubli D (2017) Umweltleistungen von Bauschuttaufbereitungsanlagen (BSAA) Schad- und Störstoffentfrachtung bei trockener und nasser Aufbereitung von Mischabbruch. Energie- und Ressourcen- Management GmbH, im Auftrag des AWEL Amt für Abfall, Wasser, Energie und Luft, Abfallwirtschaft und Betriebe, Baudirektion Kanton Zürich. März 2017. https://awel.zh.ch/internet/baudirektion/awel/de/abfall_rohstoffe_altlasten/abfall/bauabfaelle/baustoffkreislauf/_jcr_content/contentPar/morethemes/morethemesitems/umweltleistungen_von.spooler.download.1499081760573.pdf/umweltleistungen_von_BSAA_final_web.pdf (15 Oct 2019)

Salminen et al. 2005 — Salminen R (Chief-editor), Batista M. J, Bidovec M, Demetriades A, De Vivo B, De Vos W, Duris M, Gilucis A, Gregorauskiene V, Halamic J, Heitzmann P, Lima A, Jordan G, Klaver G, Klein P, Lis J, Locutura J, Marsina K, Mazreku A, O'Connor P. J, Olsson SÅ, Ottesen R-T, Petersell V, Plant J.A, Reeder S, Salpeteur I, Sandström H, Siewers U, Steenfelt A, Tarvainen T (2005) "Geochemical Atlas of Europe FOREGS – Part 1: Background Information, Methodology and Maps - Appendix: Statistical data of analytical results". Association of the Geological Surveys of The European Union (EuroGeoSurveys) & the Geological Survey of Finland, ESPOO, Finland. Nov 2005. http://weppi.gtk.fi/publ/foregsatlas/article.php?id=15 (9 Feb 2020)

Schachermayer et al. 1998 — Schachermayer E, Lahner T, Brunner PH, (1998) "Stoffflussanalyse und Vergleich zweier Aufbereitungstechniken für Baurestmassen (STOVAUF)", UBA-Monographie, Band 99, Umweltbundesamt (Federal Environment Agency), Wien (Vienna), Austria. http://www.umweltbundesamt.at/fileadmin/site/publikationen/M099.pdf s.a http://www.umweltbundesamt.at/aktuell/publikationen/publikationsliste/?pub_category_id=1 (20 Sept 2018)

Scheibengraf & Reisinger 2005 — Scheibengraf M, Reisinger H (2005) Abfallvermeidung und -Verwertung: Baurestmassen - Detailstudie zur Entwicklung einer Abfallvermeidungs- und -verwertungsstrategie für den Bundes-Abfallwirtschaftsplan 2006. Umweltbundesamt, Wien. Report REP-0009, 2005. http://www.umweltbundesamt.at/fileadmin/site/publikationen/REP0009.pdf (20 Sept 2018)

Schmutz 2015 — Schmutz D (2015) "Untersuchung der Sickerwässer von Inertstoffdeponien und von Bachsedimenten der jeweiligen Vorfluter auf Schadstoffe". Amt für Umweltschutz und Energie, Kanton Basel-Landschaft. https://www.baselland.ch/politik-und-behorden/direktionen/bau-und-umweltschutzdirektion/umweltschutz-energie/abfall/publikationen/downloads/untersuchung_sickerwasser_inertstoffdeponien.pdf (20 Sept 2018)

Shacklette & Boerngen 1984 — Shacklette HT, Boerngen JG (1984) Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States - An account of the concentrations of 50 chemical elements in samples of soils and other regoliths. U.S. Geological Survey Professional Paper 1270, Geological Survey, United States Department of the Interior, United States Government Printing Office, Washington 1984. http://pubs.usgs.gov/pp/1270/pdf/PP1270_508.pdf (31 Oct 2016)

Vaiciukyniene et al. 2020 — Vaiciukyniene D, Mikelioniene A, Baltusnikas A, Kantautas A , Radzevicius A (2020) Removal of ammonium ion from aqueous solutions by using unmodified and H2O2-modified zeolitic waste. Nature Scientific Reports, volume 10, Article number: 352 (2020) https://www.researchgate.net/publication/338602277_Removal_of_ammonium_ion_from_aqueous_solutions_by_using_unmodified_and_H2O2-modified_zeolitic_waste (22 Jun 2020) https://www.nature.com/articles/s41598-019-55906-0 (22 Jun 2020)

van der Sloot et al. 2011 — van der Sloot HA, van Zomeren A, Meeussen JCL, Hoede D, Rietra RPJJ, Stenger R, Lang Th, Schneider M, Spanka G, Stoltenberg-Hansson E, Lerat A, Dath P (2011) Environmental Criteria for Cement Based Products ECRICEM - Phase I: Ordinary Portland Cement & Phase II: Blended Cements and methodology for impact assessment. Energy research Centre of the Netherlands (ECN), report ECN-E--11-0200, November 2011 http://www.ecn.nl/docs/library/report/2011/e11020.pdf (3 Feb 2020)

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)

Wikipedia 2020 — Various authors (2020) Ductile iron . Wikipedia. https://en.wikipedia.org/wiki/Ductile_iron (26 Jun 2020)

WVZ 2001 — WVZ Wasserversorgung Zürich (2001) Die Qualität des Zürcher Trinkwassers (Stand 2001) Download on 21. Aug 2002 from http://www.stadt-zuerich.ch/wvz/qualitaet/Qualitaetsdaten_2000.asp

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