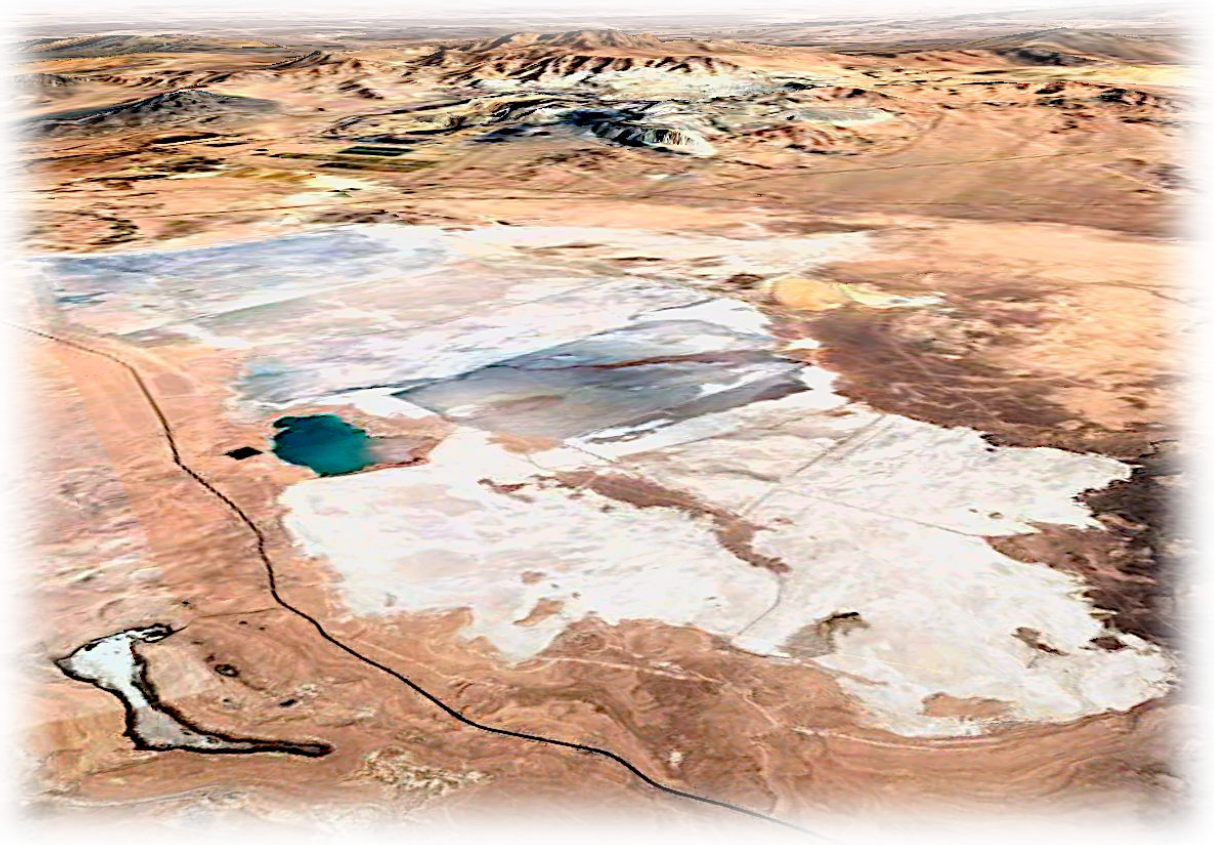
A model for waste-specific and climate-specific life cycle inventories of tailings impoundments



Technical report
Zurich, September 2017

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Title image Talabre tailings impoundment (image centre) in Chile viewed approximately from an altitude of 6 km above ground. Chile is the largest copper producer of the world and produced 30% of the primary copper in 2015. The Talabre tailings impoundment is located in the Atacama desert and measures approximately 9km by 12 km at its widest points and covers an area of approximately 60 km^2 or 8400 soccer fields. It receives the tailings of three nearby copper mines Chuquicamata, Ministro Hales and Radomiro Tomic, all belonging to Codelco, the largest copper producing company in the world. Chuquicamata is visible in the medium background of the image; it is the largest open cast copper mine of the world. 22°17' S lat, 68° 54' long W Image from Google Earth, © CNES/Airbus 2017, DigitalGlobe 2017. Imagery date 16 Oct 2016, accessed and edited (contrast, saturation, sharpness) 2 May 2017.

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.

Suggested citation: Doka G. (2017) A model for waste-specific and climate-specific life cycle inventories of tailings impoundments. Doka Life Cycle Assessments, Zurich, Switzerland. Available at http://www.doka.ch/publications.htm

Contents

1 Introduction

Tailings are the unused leftovers from the refining of primary resources (beneficiation). In beneficiation the target ore is separated from the unwanted minerals accompanying the material removed from mined vein (cf. Fig. 1.1 below). Tailings consist largely of these unwanted minerals and low quality ore minerals. In beneficiation processing the materials are ground to a fine powder and intensely mixed with oxygen and water. This renders the tailings minerals orders of magnitude more reactive than in their original underground surroundings. Contained pollutants like heavy metals are therefore much more mobile than before and more likely to be emitted over time. If in addition the tailings contain sulfide (S^{2-}) the processing can prompt the reaction of sulfide and oxygen forming sulfuric acid. The resulting low pH then accelerates leaching of heavy metals even more. This process is known as Acid Rock Drainage (ARD).

Large masses of tailings are produced per kilogram metal, especially if ore grades are low. For instance the worldwide average of the currently mined copper ore grade (2017) is approximately 0.6%, which means for every kilogram of copper *ore* a tailings mass of 167 kilogram needs to be disposed, even more for each kilogram of smelted copper *metal*.

Tailings impoundments often represent a considerable environmental risk in the primary metal process chain. Tailings contain mobile toxic metals and often large masses of tailings are produced. Although the environmental importance of tailings burdens for primary metal production was known, the ecoinvent database from 2003 to 2009 contained some critical assessment gaps by not including

tailings disposal for several primary resources. [1] In 2009 the ecoinvent database received an update in v2.1 which included the assessment of long-term emissions from tailings of primary metal production. [2] A dedicated tailings model was created, largely based on the modelling principles of landfill models already employed in ecoinvent, but specifically based on information for tailings. That model was based on a large literature survey of tailings composition and leachate concentrations (Doka 2008a). As with other tailings disposals only a *global average* was inventoried. A global average was sufficient as the metals consumption in LCA refers to a generic global average as well.

To allow these disposal assessments become more regionalised, an extended model is developed here that allows the creation of even more specific tailings disposal inventories, that can heed a specific tailings composition and local site-specific climate.

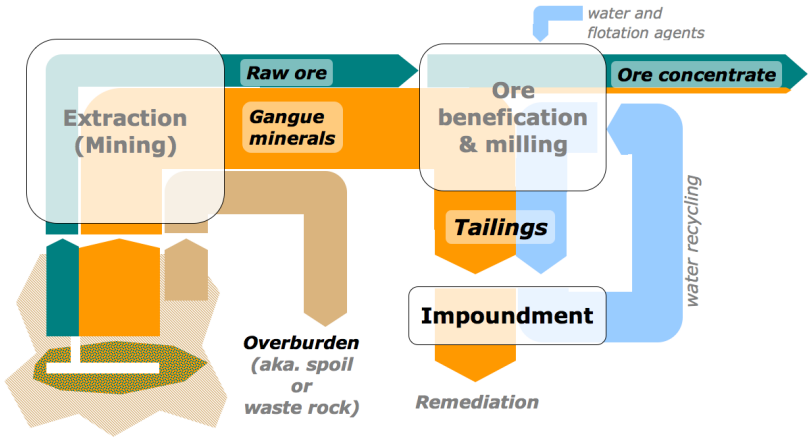

Fig. 1.1 Generation of tailings within the mining & beneficiation process.

2 Tailings impoundment model

The model for the tailings impoundment is based on the modelling framework applied in existing landfill process models used in ecoinvent, see ecoinvent report 13 (Doka 2003-III, 2017). These models are based on a top-down approach. Literature data on measurements of deposited waste compositions (solids) and the currently measured concentrations in leachate (liquid) form the basis to characterise pollutant mobility and calculate the currently occurring rate of emissions of the landfill. Heeding chief parameters like hydrology, pH development and elemental chemistry these emissions are then projected into the future. Thus these models incorporate physical and chemical conditions, while being rooted in the currently observed emission behaviour. The inventories estimate the expectable, anticipated magnitude of the emissions a particular disposal design is *bound* to produce. They represent the foreseeable pollution potential of a disposal technology. More details on general

[1] The direct emissions from tailings impoundments were however included for *some* commodities, like primary production of aluminium (bauxite), phosphor, and potassium. For uranium mill tailings the direct emissions were only assessed partially with a coarse model for short-term emissions, while radioactive emissions from uranium mill tailings were already included with a long-term time horizon of 80'000 years. Tailings from non-ferrous metal ores were only assessed with a land use, while all direct emissions from tailings were left out, due to a lack of a specific tailings model.

[2] In the same ecoinvent version 2.1 of 2009 also data gaps for non-radiological emissions from uranium mill tailings were closed (Doka 2008b). In version ecoinvent 2.2 emissions from coal tailings and overburden of coal and lignite mining (spoil) were included (Doka 2009).

modelling assumptions of these models can be taken from (Doka 2003-III). The tailings model presented here and its transfer coefficients are not applicable to other waste types.

In the present extension of the model, the large body of compiled literature data on tailings composition and leachate data is used to derive a *working point model* of the impoundment that describes the average behaviour of chemical elements in such a type landfill, expressed in transfer coefficients for 41 chemical elements (see middle part in Fig. 2.1) . Already included in that data is site-specific information on precipitation, which influences the mass of available water trickling through the tailings impoundment body or site-specific information on impoundment height (see top part in Fig. 2.1).

This working point model is then applied to a *specific* tailings composition provided by the user (see bottom part in Fig. 2.1). Therefore the direct emissions of the tailings impoundments are made specific to a certain waste composition. Having used local conditions to derive the working point model, the inventory results also reflect these local circumstances.

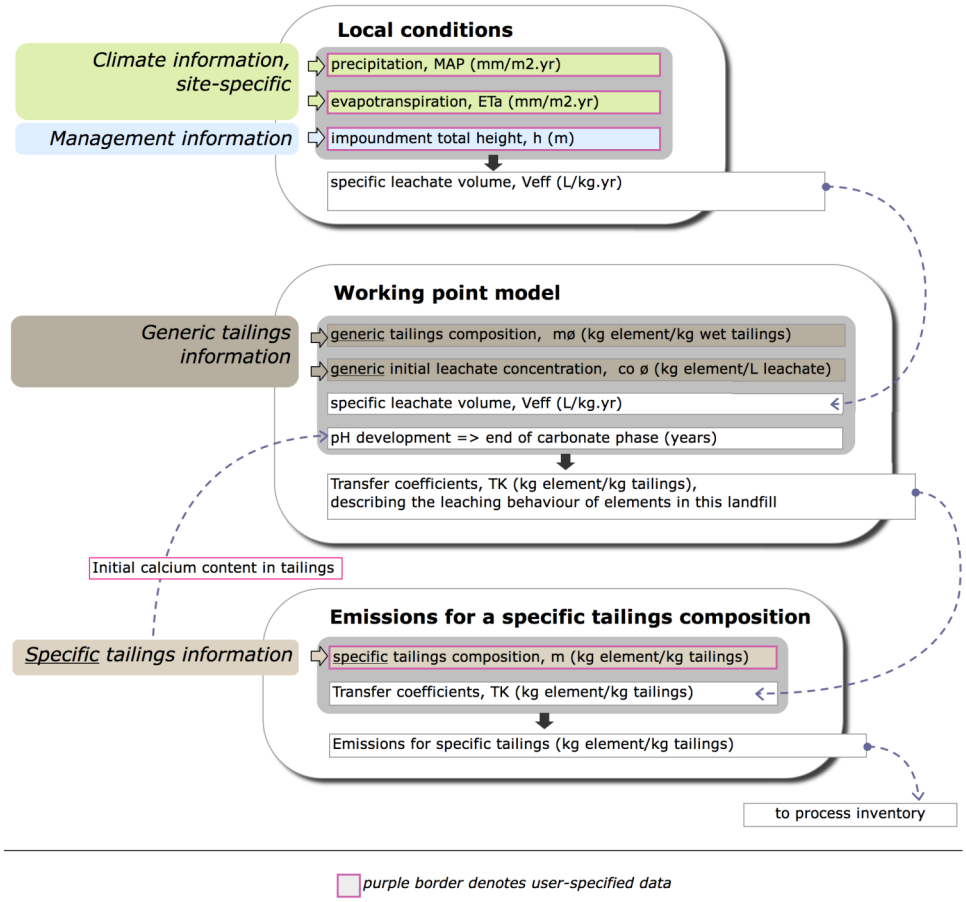

Fig. 2.1 Simplified scheme of the information flow in the tailings impoundment model, combining site-specific information (green), disposal management information (blue), and tailings composition information (brown).

Please note that in ecoinvent's models of landfills with organic degradable waste, like residential waste dumps and municipal sanitary landfills, additional issues are heeded like differing degradability of waste materials, generation of landfill gas, retention/re-precipitation of degraded waste material (Doka 2003-III, 2017). In the tailings impoundment model there is no degradability parameter. These are

chiefly inorganic waste materials and zero landfill gas generation is assumed (as already done for bottom ash/slag compartments and residual material landfills).

Intended use of the tailings impoundment model

The goal of LCA in general is to depict the likely environmental burdens of human activities in a circumspect manner. Ecoinvent is a backgrounds database that contains such information for a large range of different processes and products. These backgrounds processes are usually generic or typical averages. It is not the goal of ecoinvent to cover all possible or imaginable product variants or production methods, only an *average* or the *most frequent ones* in a generic manner. This translates also to inventories of disposal processes: the goal is to capture the *typical burden* for a certain waste product disposal in a generic manner. This has been the main driver behind the efforts to make disposal inventories dependent on specific waste compositions. With these models it is possible to heed the differences in waste composition at the end-of-life of a product, and not being forced to attribute identical burdens to each kilogram of waste, only because it is managed in a similar way, like incineration or landfilling. The ecoinvent disposal models do not only heed the particular characteristics of a treatment technology, but also the particular characteristics of a certain waste product.

The present model extends the specificity of the tailings disposal model to heed local conditions and become more site-specific, especially regarding hydrology, instead of having merely one generic global average. Still the intended use of the resulting inventories is to *complement* life cycle data of products and production. The goal is not to judge for example mine siting or optimise mining operations, but to provide founded estimates on the expectable, anticipated emissions from tailings impoundments. The models presented here shall not be confounded with procedures to do Environmental Impact Assessment (EIA) or environmental auditing for a particular single impoundment, as EIA has quite different scope and purposes than LCA.

3 Local conditions

The local conditions employed in the model are the same as already utilised in the new landfill models (Doka 2017).

- **Mean Annual Precipitation MAP:** determines how much precipitation water is available.
- **Actual Evapotranspiration ETa:** determines evaporated precipitation water and therefore determines water entering the impoundment body.
- **Mean Annual Temperature MAT:** will be used here to determine permafrost conditions which slow down landfill weathering.
- **Landfill height h:** determines over which mass of tailings the leachate water will be distributed.

To correctly estimate the leachate volumes of a landfill it is important to consider the *actual* evapotranspiration rates *ETa* which are based on the available water, and not *potential* evapotranspiration rates *PET*, which are based on a theoretical unlimited supply of water.

These local conditions are influencing the effective annual amount of leachate V_{eff} produced per year per kilogram of tailings. The equation 5.20 of (Doka 2003-III) is employed which heeds preferential flow pathways.

Eq. 3.1 $$V_{eff} = \frac{1 - w\%}{\left(\frac{h \cdot \delta}{I} - \frac{T_p \cdot w\%}{v\%}\right)}$$

V_{eff}	Effective leachate volume (liter/kg·yr)
w%	Share of preferential flow in leachate output (5%)
h	Impoundment height (m)
δ	Tailings density (2200 kg/m^3)
I	Infiltration rate (mm/m^{2}a)
T_p	Residence time of preferentially flown water (0.346a = 18 weeks)
v%	Water content in tailings (30 w-%)

A few parameters of the generic global tailings model (Doka 2008a) are adopted here as well. As tailings are a finely grounded mineral matrix the occurrence of preferential flow is diminished over a common municipal waste landfill. The share of preferential flow w% and the residence time of preferentially flowing water T_p are adjusted accordingly. Water content v% is assumed at 30%. Tailings density δ is assumed to be 2200 kg/m^3. Impoundment height h is a figure provided by the user.

The infiltration rate I is derived from data on precipitation MAP and actual evapotranspiration ETa. Water not evaporated is assumed to enter the landfill body. The evapotranspiration data refers to naturally vegetated surfaces (Doka 2017:Fig 4.3). As tailings impoundments represent a difficult ground to allow vegetation, the provided site-specific ETa value is divided by two to determine impoundment hydrology.

No surface runoff of precipitation water is assumed. It is assumed optimistically that the impoundment is constructed in a way that no surface water leaves the impoundment. Such surface creek formation on impoundments would also mean surface erosion which would pose a threat to the mechanical long-term stability of the impoundment. It is assumed that the impoundment is constructed in a way that this does not happen.

Eq. 3.2 $$I = \left| MAP - \frac{ETa}{2} \right| \cdot \left(1 - e^{-ft \cdot [MAT + 15°C]}\right)$$

MAP	Mean Annual Precipitation (mm/m^{2}a)
ETa	Actual evapotranspiration (mm/m^{2}a)
ft	Factor for temperature-dependence, constant = 0.3 [-]
MAT	Mean Annual Temperature [°C], please note addition of 15°C

An additional extension is heeding the risk of permafrost conditions. In (Doka 2017) a coarse estimate of permafrost occurrence based on mean annual temperature MAT was derived, ibid. Eq. 2.2. During freezing leachate flow is halted and this effectively translates to a reduction of available water over time to produce emissions. The term in the right bracket of Eq. 3.2 expresses this reduction of water flow dynamics based on the locality's mean annual temperature.

3.1 Weathering in wet and dry climates

There are distinct differences in the fate of impoundment leachate in wet and dry climates. In wet climates leachate transports pollutants *downward* through the impoundment body towards

groundwater. This is the normal case for temperate climates. In dry climates, evaporation at the surface is so strong that an *upward* pressure gradient results, which draws groundwater through the impoundment to the surface. On the surface the water evaporates and leaves behind solid, but brittle precipitates from the dissolved elements, called evaporites or efflorescent salts (see Fig. 3.2) . Evaporites are prone to surface erosion by wind. Wind erosion is a relevant emission pathway in arid climates.

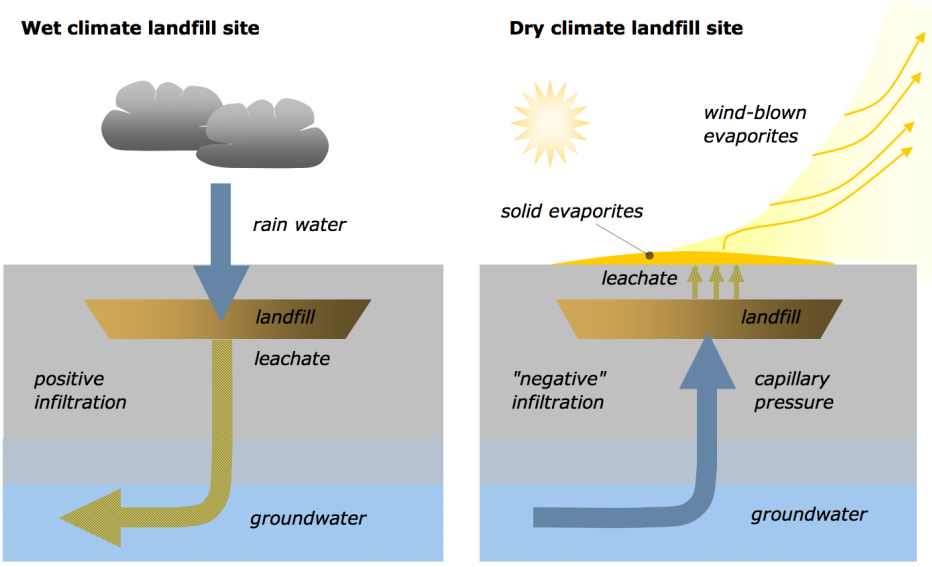

Fig. 3.1 Conceptual differences in emissions from landfill sites in *wet* climate and in *dry* climate.

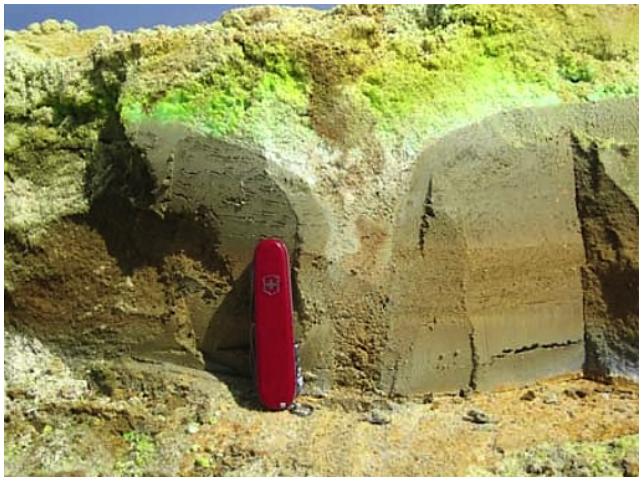

Fig. 3.2 Evaporites forming on tailings from copper mining at Bahia de Ite, Peru. Photo taken from (Diaby et al. 2006)

For the present tailings impoundment model following assumptions are made:

- Net annual infiltration is considered to decide if a landfill site has reversed leachate flow.
- Net annual infiltration is calculated from Eq. 3.2 on page 7.
- With a negative net annual infiltration, the site is considered to have reversed leachate dynamics ("reversed dry site")
- On reversed dry sites leachate solids accumulate on top of the site as evaporites

- Of the accumulated evaporites 10% are considered to be eroded by wind and inventoried as emission to air of each chemical element. This percentage can be altered. 10% was already used in the ecoinvent database for uranium tailings in dry locations like Namibia (Doka 2008b).
- Of the windblown evaporites the sum mass is also inventoried as particulate matter emission (PM).[3] A conservative profile of 20% $PM_{<2.5}$, 30% $PM_{2.5-10}$, and 50% $PM_{>10}$ is assumed (Doka 2008b). [4]
- Of the accumulated evaporites 90% are inventoried as emission to industrial soil.
- In sites with positive net annual infiltration and downward leachate flow, leachate emissions are inventoried as groundwater emissions.

The impoundment weathering is chiefly a function of the available infiltration water. A small amount of infiltration water will slow down landfill weathering and move the point in time when the carbonate buffer is washed out further into the future. If this is the case the carbonate phase lasts longer, which means that the pH remains longer non-acidic and more time is available to efficiently wash out oxianions. Oxianions are well soluble at non-acidic pH values, while many heavy metal cations are better soluble at acidic pH. A longer carbonate phase can therefore increase inventoried emissions of oxianions. Because of this chain of reasons, the inventoried emissions of oxianionic elements like chromium or antimony can *increase* with *less* infiltration water. This might seem paradoxical, but is a sensible result from the model. In turn, the familiar cationic heavy metals (zinc, lead, cadmium, copper etc) have the unsurprising behaviour of less emissions when infiltration water is scarce.

3.2 Working point model of the impoundment

The local conditions determine the available water to produce emissions. How much emissions actually occur is derived from a transfer coefficient model. The transfer coefficients define the behaviour of each chemical element in the landfill, i.e. how much of it is leached out of the landfill body. Transfer coefficients are between 0 and 100%, so not more can be emitted than is present in the landfill.

For the working point model the local conditions are employed to derive the effective annual leachate volume V_{eff} , see Eq. 3.1 on page 7. Data of the large literature survey on tailings composition and leachate concentrations from (Doka 2008a) is used to describe the initial mobility of elements in a generic tailings body. Transfer coefficients are then derived in the common manner employed in other inorganic landfill models (Doka 2003-III:19ff).

The resulting working point model then contains the information on how a chemical element typically behaves in a *tailings* body specifically. Additionally the tailings model includes various leaching dynamics over time appropriate to chemical elements, not only exponential, heeds potential buffer washout and the effects of a pH drop.

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[3] This is in accordance to methodological choices in the ecoinvent database, where particulate emissions to air can be inventoried as PM figures (where mass and size is the dominant aspect) and additionally as toxic compounds contained in the particulates (where toxicity is the relevant aspect).

[4] Data from field measurements of wind blown aerosols agree well with the recently formulated brittle fragmentation theory BTF (see Kok et al. 2012:Fig 4.3b). This data suggests that typically around 77% of total aerosol mass is PM_{10} and around 9% is $PM_{2.5}$. The assumptions made for the tailings model (with 50% PM_{10}, 20% $PM_{2.5}$) are conservative, as impact scores with the BTF ratios would be 30–50% *higher*, depending on the LCIA method.

The use of transfer coefficients characterising the mobile fraction of a solid is somewhat comparable to the use of solid-liquid partition coefficients (Kd) for the fate and leachability of pollutants in soil of LCIA models, for instance USES-LCA. But while in LCIA soil fate models only one single generic set of partition coefficients is used for any type of soil, here data pertaining specifically to tailings impoundment environments is used.

4 Waste-specific information

The transfer coefficients of the working point model define the behaviour of chemical elements in the specified tailings impoundment, which includes local climate and disposal management information. Now lastly a specific tailings composition can be combined with the derived transfer coefficients in the tailings impoundment to calculate the waste-specific emissions of the tailings impoundment.

The specific tailings composition should relate to an average composition encountered at the specified locale. It should cover as many of the chemical elements as possible, most notably the toxic ones, like heavy metals (copper, lead, cadmium, chromium etc), arsenic, fluoride, in order to avoid assessment gaps. [5] For the purposes of the tailings model, it is assumed that the whole tailings impoundment is composed of the specified tailings composition.

If possible, a calcium content should also be specified. While calcium has no toxicity by itself, it is used to determine pH development of the landfill body. This is described in the next section.

4.1.1 Determination of the end of the carbonate phase

In the ecoinvent landfill models (Doka 2003-III) the washout of buffering materials was used to determine the point in time when the pH of the landfill would be decreasing (ibid. Eq. 5.15). This is the end of the carbonate phase t_e. The carbonate buffer is mostly associated with the presence of calcium. While in the previous landfill models in (Doka 2003, 2008, 2009) the working point model was created in a static manner and only one end of the carbonate phase per landfill type resulted, the present model must be made flexible to allow for a variety of situations, depending on waste composition and site climate/hydrology. This is achieved by looking at the calcium content of the specific tailings composition. It is assumed that the tailings body is made up from this material alone, i.e. is the waste body of the working point model. The magnitude of leachate volume V_{eff} and of the calcium concentration in leachate then determines how fast this buffer will be depleted.

Eq. 4.1 $$t_e = {m_{Ca}}/{c_{0,Ca} \cdot V_{eff}}$$

t_e	End of the carbonate phase (years)
m_{Ca}	Specific calcium content of the tailings (kg Ca/kg tailings)
$c_{0,Ca}$	Calcium concentration in leachate (kg Ca/liter leachate)
V_{eff}	Effective leachate volume (liters/kg·yr)

If the user has not specified a specific calcium content the generic value of the literature survey for global tailings is used (see Tab. 6.1 on page 12).

[5] It is misleading to determine that a tailings composition A is more burdensome than an other composition B, when A has composition data for many more elements than B.

The determined end of the carbonate phase is used in the calculation of the transfer coefficients of the working point model. So if possible the specific tailings composition will be used to determine the pH development of the *whole* impoundment. This is correct as – unlike with other landfill models – the specific waste composition is not a subset of a larger average landfill body, but actually represents the contents of the whole impoundment body. The generic global data is chiefly used to determining typical elemental *behaviour* in tailings bodies and spare the user to research also specific leachate concentration data.[6]

5 Emission speciations

5.1.1 Sulfur in leachate

From the literature values compiled for leachate composition typical generic values of sulfur in sulfate (SO_4) and in sulfide (S^{2-}) are available (cf. Tab. 6.2 on page 13). These values suggest that sulfur in leachate is 46% present in sulfate form and 54% in sulfide form. This split us used for all sulfur emissions in leachate.

5.1.2 Nitrogen in leachate

From the literature values compiled for leachate composition typical generic values of nitrogen in nitrate (NO_3) and in ammonia (NH_3/NH_4) are available (cf. Tab. 6.2 on page 13). These values suggest that nitrogen in leachate is 33% present in nitrate form and 67% in ammonia form. This split us used for all nitrogen emissions in leachate.

5.1.3 Carbon in leachate

Emissions of carbon in leachate are only inventoried with summary carbon emissions, not individual hydrocarbon compounds. Carbon emissions to water are inventoried simultaneously as TOC, DOC, BOD, and COD in the ecoinvent database. All carbon in tailings leachate is assumed to be emitted as dissolvable organic carbon (DOC=TOC), identical to the assumption made in other ecoinvent landfill models. A COD/TOC ratio of 0.091 is inferred from tailings water measurements for a Cu-Zn mine (IPPC 2004:159). A BOD/COD ratio of 0.02 is inferred from measurements of a Au mine in (Acheampong et al. 2013: SI Tab.5). Assuming these figures are both typical for metal mine tailings leachate, a BOD/TOC ratio of 0.00182 is calculated. These ratios are used to calculate BOD and COD exchanges from the TOC exchange obtained from the impoundment model.

These ratios suggest very low degradability of organic carbon in tailings leachate, which is an expected observation.

5.1.4 Other speciations

The split of particulate matter emissions to air (in the case of arid sites) is explained in chapter 3.1 'Weathering in wet and dry climates' on page 7.

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[6] This dynamic consideration of the end of the carbonate phase based on calcium content and leachate volume is different from the procedure used for the previously crated sulfidic tailings inventory of (Doka 2008a). There merely a *static* Acid Base Accounting (ABA) was performed using the initial contents of carbonate (CO_3) and sulfide (S^{2-}) present in tailings. For the tailings composition researched from literature values it was found that much *less* protons (H+) could be created from sulfide oxidation than could be neutralised by the carbonate buffer. The material therefore appeared well buffered and the end of the carbonate phase was thus assumed to not be reached within the modelling period. That procedure neglects however buffer washout over time. With the present model, buffer washout as the major mechanism of pH development is heeded. Whether the end of the carbonate phase is reached is influenced by the climate/hydrology of the site.

6 Appendix: compiled literature values

Tab. 6.1 Values for wet tailings composition and leachate composition. Geometric mean of found literature values. Used in impoundment model.

	Tailings composition mg/kg wet tailings	**Leachate composition** mg/L leacahte
O	428'200	n.a.
H	n.a.	n.a.
ORG-C	7266	3.383
S	14'620	2789
N	23.1	n.a.
P	493	290.6
B	310.3	42.66
Cl	2	n.a.
Br	0.2289	n.a.
F	826.4	2.418
I	n.a.	n.a.
Ag	14.42	0.0009861
As	284	0.02831
Ba	459.6	0.006821
Cd	7.158	0.02337
Co	18.59	0.05144
Cr	42.55	0.005719
Cu	277	0.1391
Hg	0.8819	0.0000418
Mn	875.3	2.085
Mo	7.024	0.2447
Ni	22.81	0.02231
Pb	468.4	0.01369
Sb	79.2	0.009127
Se	6.811	0.02603
Sn	11.63	0.01606
V	57.04	0.008005
Zn	790.9	0.9409
Be	1.453	0.006152
Sc	6.131	n.a.
Sr	112	0.879
Ti	6.287	0.01259
Tl	2.599	0.001658
W	10.93	n.a.
Si	154'000	2.106
Fe	45'430	6.492
Ca	11'890	225.5
Al	27'810	2.274
K	9603	13.54
Mg	7396	62.89
Na	1456	14.79

Tab. 6.2 Additional values for wet tailings composition and leachate composition. Geometric mean of found literature values. Not used in model (except S^{2-}/SO_4 and NO_3/NH_4 for speciation).

	Tailings composition mg/kg wet tailings	**Leachate composition** mg/L leachate
CO3-C	27'150	17.43
S2- -S	8340	482.7
SO4-S	268.3	410
NO3-N	n.a.	0.5349
NH4-N	n.a.	1.073
pH	7.056	4.712
AP *	53'070	22.36
NP **	13'250	15
Rb	92.2	n.a.
Th	6.052	0.015
U	2.878	n.a.
Zr	74.18	n.a.
Nb	10.58	n.a.
Y	15.87	n.a.
Ga	12.15	n.a.
La	36.48	n.a.
Ce	48.86	n.a.
Nd	43.03	n.a.
Au	0.1183	n.a.
Bi	25.57	0.0003347
Cs	10.5	n.a.
In	2.112	n.a.
Re	0.1095	n.a.
Ir	0.02131	n.a.
Li	23.58	n.a.
Yb	10.95	n.a.
Ge	1.575	n.a.
Te	2.216	n.a.
Pr	8.134	n.a.
Hf	3.793	n.a.
Ta	2.8	n.a.
Pt	0.1289	n.a.
Pd	0.05954	n.a.

* AP = Acidification Potential in mg $CaCO_3$

** NP = Neutralisation Potential in mg $CaCO_3$

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