

Life Cycle Inventory of the disposal of lignite spoil, coal spoil and coal tailings



Commissioner



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Contents

1	Introduction.....	3
2	Description of waste materials	5
2.1	Spoil from coal and lignite mining.....	5
2.2	Tailings from coal preparation	7
3	Disposal models	8
3.1	Leachate volume	8
3.2	Time frame for modelling	9
3.3	Future development of leachate concentrations	10
3.4	Fires.....	10
3.5	Failures.....	10
3.6	Other expenditures	11
3.7	Lignite spoil disposal model.....	11
3.8	Coal tailings disposal model	14
3.9	Coal spoil disposal model	19
3.10	Inventory tables	21
4	Result discussion	23
5	References	26

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March 2021. Corrected emissions of PO₄ in Tab 3.11 (all three datasets) and long-term NO₃ and SO₄ emission in coal tailings.

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Picture on title page: Spoil deposition in the German lignite mine Garzweiler II.

Original from <http://www.flickr.com/photos/22746515@N02/3281728778/sizes/o/in/set-72157607120764846/>

1 Introduction

This report inventories the environmental burdens of the disposal of waste materials generated from coal and lignite mining and processing. Overburden is the subsurface rock or sediment material that needs to be removed in order to gain access to a coal or lignite vein to be mined. After removal the overburden is termed spoil (Ger. *Abraum*). Especially in open cast mines large amount of spoil are generated. The raw coal from mining operations is milled, concentrated and purified in the processing step (also called preparation or milling). The non-coal minerals separated from the raw coal are called tailings (Ger. *Berge*¹). In accordance with the mining inventories in ecoinvent, only processing of coal, but not lignite is considered here².

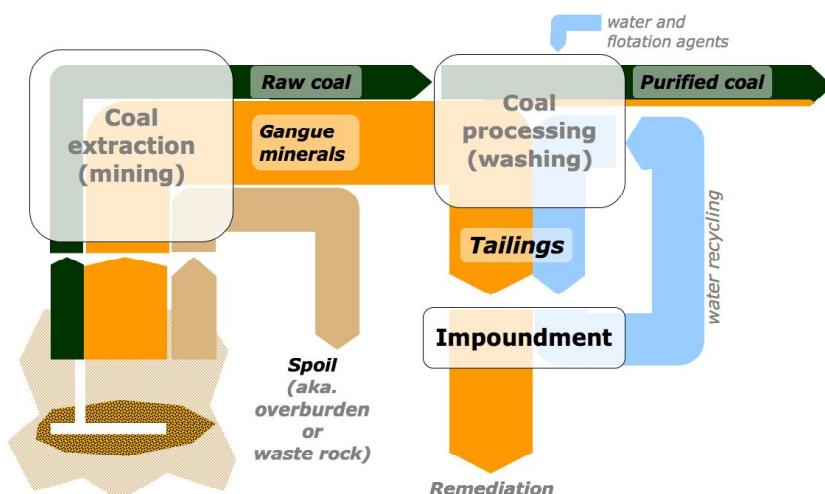


Fig. 1.1 Schematic of the origins of spoil and tailings waste in coal mining.

Spoil and tailings waste need to be disposed. Both types of materials pose environmental problems, due to the fact that the original rock or mineral material is mechanically disrupted by the mining and milling process. Mixing with water and exposure to air oxygen promotes the microbial oxidation of sulfide minerals, like pyrites, to sulfuric acid, which in turn can lower pH and increase the solubility of many heavy metals. This process is known as Acid Rock Drainage ARD or Acid Mine Drainage AMD and is a chief environmental issue of mining operations, especially where sulfide content is high and the content of pH buffering mineral is low.

In this report an emissions model of spoil and tailings disposal is developed based on the landfill disposal models presented in ecoinvent report No. 13 (Doka 2009), which were also used to create similar disposal emission models for tailings from non-iron metal mining (Classen et al. 2009) and tailings from uranium mining (Dones et al. 2009). Following the lead in those efforts, the goal is to attain a good estimate for a *global average burden* of the disposal of tailings and spoil. The goal is not to assess one particular disposal site or to optimise the disposal site performance, but to obtain a typical global average of burdens from currently practiced disposal processes in order to complement

¹ In coal mining literature the distinction between spoil and tailings is not always as clear cut as defined here and various nomenclatures abound. For example Reichelt & Site (2004:29) consider 'Berge' to be the summary rock waste material from coal processing operations *and* from coal mining, which would include spoil. As spoil and tailings can be disposed together, a separation by name is plausible to be abandoned for such deposited materials, with either of the terms prevailing.

² Definitions of coal vs. lignite are internationally not unanimous. The term "coal" is used here in accordance with ecoinvent for all hard coal types, anthracite, bituminous and subbituminous coals with heating values approximately above 20 MJ/kg. Lignite (soft and hard brown coal) is used for fossil carbonaceous materials with poorer heating values, excluding peat.

the burdens from other life cycle stages of coal and lignite. As in other landfill datasets of ecoinvent, the functional unit is one kilogram of deposited waste.

2 Description of waste materials

2.1 Spoil from coal and lignite mining

Spoil is the non-target rock or sediment material, which is excavated during mining operations to gain access to the targeted coal or lignite vein. The annual global production of spoil is estimated to be 22'240 million tons from coal mining (Tab. 3.9) and 8600 million tons from lignite mining (Tab. 3.2).

The undisturbed original rock or sediment material above a vein is called overburden. Although overburden in its original state can generally be considered rather environmentally harmless, the mechanical disruption by the mining excavation renders it more reactive by creating fractures. In coal mining the excavated depths can be hundreds of meters and the spoil will then additionally suffer disintegrating action simply by the relief of pressure from superimposed rock³. Also the commencement of hydration and crystallisation processes can tear up spoil material further than from man-made interventions alone. Although the chemical composition as such is not altered, these processes increase the surface-to-volume ratio and expose previously well sheltered mineral phases to surface weather, and thus increase reactivity and emission potentials. Typical problems of lignite spoils are high salt wash-off and acid rock drainage, see Fig. 2.2. Another possibility to express the hazard of these materials is to test them against permissible values. The average concentrations found for lignite spoil leachate, as collected from a literature survey (see Tab. 3.1) would for example be judged as a contaminated site according to the Swiss contaminated sites ordinance⁴.

The majority of spoil material is either impounded or stacked on the surface, or in the case of open cast or pit mines backfilled. Some material can be reused in construction, e.g. dams. In underground mines backfill of spoil in disused shafts is possible and desirable, because it diminishes problems of cave-ins, which can lead to surface damages. Backfilling in underground mines is however considered an expensive disposal method and in Germany only 3% of the coal spoil and tailings are backfilled in underground mines (Reichelt & Sitte 2004:29).

³ To illustrate: the pressure from overlaying rock in 200 m depth is approximately 40 bars or the weight of over 400 metric tons, equivalent to a good sized railway train composition, per square meter.

⁴ The Swiss contaminated sites ordinance (CSO, *Altlasten-Verordnung, AltLV*) of 2009 establishes threshold limit values for protection of groundwater for various inorganic pollutants. Direct porewater or leachate concentrations serve as criteria to test a site for compliance. The leachate concentrations in Tab. 3.1 (p.12) and Tab. 3.4 (p.15) would overshoot some of those thresholds (SCL 2009).



Fig. 2.1 The spoil pile Haniel, a facility of the active underground coal mine Prosper-Haniel near Bottrop, Germany. Parts of the spoil pile are still being filled, while parts unused for stacking spoil are remediated to recreational areas, here an amphitheatre (Binlasch 2007).



Fig. 2.2 The former lignite mine Zwenkau, Germany, six and nine years after closure. Note the red water in the foreground, indicating visibly ongoing Acid Rock Drainage processes from pyrite oxidation (Hannusch 2005, Petersen 2008)

2.2 Tailings from coal preparation

Mined raw coal is purified in the coal preparation step (also called coal milling, coal washery, or coal refining) and mechanically separated from non-coal minerals using milling and flotation steps. The ground-up, non-coal mineral remains produced in raw coal preparation are called tailings (or variably reject, refuse, slurry, or discard). The liquid waste prior to settling is variably called coal sludge, coal slurry or black water). Processing of lignite is uncommon and not considered here, in accordance with the mining inventories in ecoinvent. Only tailings from coal preparation are inventoried. Coal tailings are usually impounded in tailings ponds or impoundments to recycle the water. The annual global coal tailings production is estimated to be 1167 million tons per year (see Tab. 3.5).

The tailings material poses similar environmental problems as spoil, as the milling and flotation renders the formerly secluded minerals more reactive. Due to the even smaller grain size of tailings these problems are further amplified. Again, as an expression of the toxicity of this waste, the average coal tailings leachate concentrations found in the literature survey (Tab. 3.4) would prompt a coal impoundments site to be judged as a contaminated site according to the Swiss contaminated sites ordinance (SCL 2009).



Fig. 2.3 Sampling procedures at an operational coal tailings impoundment and close-up of impounded coal slurry.
Pictures from (CIP 2004)

3 Disposal models

Based on the principles outlined in (Doka 2009) three new landfill models for the disposal of lignite spoil, coal spoil and coal tailings are created. Data of waste material elemental compositions and elemental concentrations in leachate is collected from a literature survey⁵.

The goal of this report is to create a generic or typical worldwide average emission inventory for the disposal of such materials. The inventory is derived from landfill models, which are based on data observed in the field, to determine realistic emissions. To this end, the literature values on compositions from various regions are combined to result in an unweighted geometric mean value, which is taken to be the typical generic composition value; a procedure used in previous landfill models of ecoinvent. This sampling procedure yields results with a sound chemical behaviour, see result discussion, chapter 4 on page 23.

As an intermediate result of the modelling so called transfer coefficients are calculated. These parameters indicate for each chemical element present in the deposited waste how much of it will be leached out of the landfill over time. Transfer coefficients are given in percentages, which translate as "kg element emitted per kg element deposited", 100% being the maximum value. The calculation of transfer coefficients is a helpful intermediate result, which allows to characterise landfill behaviour and to make meaningful comparisons with other landfill types (see result discussion).

It is assumed that the landfill models are generally applicable to spoil and tailings impoundments, backfills or heaps. In former landfill models leaching with rainwater infiltration from the surface alone was heeded as a pollutant transport media. This is well applicable to spoil heaps on the surface and to tailings impoundments. Spoil backfills however can attain large dimensions; in German lignite mines depths of several hundred meters are established, meaning that the spoil backfill is several hundred meters thick. After restoration the waste material is therefore not only exposed to infiltration water from rainfall on surface area, as established in previous landfill models, but also to lateral *groundwater flow*. For large landfill bodies like backfills the groundwater flow is likely to bring more water to the spoil per year compared to the direct surface infiltration water alone. With larger water availability larger mobility of pollutants and increased emissions are feasible. Conversely however, in the deep underground the oxygen availability is generally lower, thus pyrite oxidation and Acid Rock Drainage is less likely to occur (see also chapter 3.3 on page 10). The quantitative details however depend on climate, the groundwater basin size, aquifer depths and its characteristics, landscape topography, and are not quantified in this generic case. In the landfill models pollutants are transported only by water directly infiltrating from the surface. Groundwater in-flow and flow-through across the deposit and resulting pollutant removal is not considered. This results in conservative⁶ estimates on emissions.

3.1 Leachate volume

The natural precipitation and thus water availability at the disposal site is an important aspect of leaching speed. The exchange of water determines transfer of pollutants out of the tailings body, i.e. emissions. While the deposited composition represents the pollutants initially present in the landfill body, the porewater concentrations represent the tendency of these pollutants to be transferred into the

⁵ Data is taken from following sources: Berrios & Kolitsch 2001, Brüschke 2001, Cesnovar 2004, Das et al. 2006, EPA 1999, Güdelhöfer et al. 1988, Hecht et al. 2003, IPPC 2004, Kind 2007, Maiti et al. 2004, McSpirit 2005, Meuser et al 2005, Oehmig 2003, Reichel & Schauer 2007, Reichelt & Sitte 2004, Rinker 2001, Rose & Cravotta 1998, Schöpel 1985, Schüring 1996, Singh & Gupta 2007, Singh 1990, SSP 2009, Tauber 1988, Van Rensburg & Morgenthal 2003, Wang 2004, Weiler 2000, Wiggering et al. 1991, Wisotzky & Lenk 2007, Wisotzky 1999, Wisotzky 2003, Wisotzky 2004, WVDEP 2009.

⁶ The word 'conservative' is used here with the meaning that emissions are likely to be higher than inventoried.

solution phase and being mobilised. It is water availability and the exchange rate of water in the landfill that ultimately determines the actually occurring emissions out of the landfill.

In order to estimate an appropriate leachate volume, the climate at the world's current major coal and lignite mining sites was analysed. The locations of the most important mining sites for coal and lignite were heeded, representing 90% of coal and lignite production, respectively, based on production figures 2000-2006. Average precipitation data for these mining sites were derived from (FAO 2000). A part of the precipitated water will evaporate and not become available to the landfill body. Active or passive remediation of landfills are aimed at increasing the vegetation cover, which in turn will increase evapotranspiration. In Germany it is noted that coal spoil heaps, due to their dark colour, can attain daytime temperatures of up to 60°C. The model heeds evapotranspiration for the leachate volume calculation. Actual evapotranspiration rates at mining sites are established from data in (UNEP GRID 1994). A generic infiltration rate is established, by weighting in different mining sites according to their annual spoil or tailings production (see Tab. 3.2 ff.).

The annual leachate volume that is effectively leaching out pollutants, is modelled with the variable V_{eff} with the unit litres of leachate per year and per kilogram of deposited material [$\lambda/a\cdot kg$]. Details of the calculation and variables are explained at length in (Doka 2009). Details for the three landfill models developed here follow below.

3.2 Time frame for modelling

As in other landfill models of ecoinvent, long timeframes need to be heeded in order to record the relevant emissions caused by a landfill. The soil and the underground in general is a rather slow environmental media compared to air or water. In order to capture the processes occurring in landfills a long timeframe needs to be considered. Adopting a mere short-term perspective is not correct for assessing landfills, since that fails to detect many important and characteristic environmental burdens from landfills. The importance of a long-term perspective when it comes to sub-surface processes is well established e.g. in the performance assessment of final repositories for nuclear waste.

In the ecoinvent database a modelling period of 60'000 years for Swiss landfills is chosen (Doka 2009). This represents a coarse 'ecological planning horizon' for Switzerland: by that time the Swiss midlands will probably be covered again with glaciers that remodel the Swiss landscape. The Swiss ecosphere will be redefined, creating new boundary conditions for environmental goals. The emissions over 60'000 years therefore represent a rough mean expectation value of the burden inflicted on the ecosphere as we know it now and are concerned about also for the sake of our long-term descendants. In other regions this ecological planning horizon might be dissimilar. Especially in tropical regions very long periods of climatic stability can be expected⁷. Despite these variations, the timeframe for the landfill models of this report is set to 60'000 years, i.e. identical to modelling periods for Swiss landfills and for tailings from non-iron metal ores, and similar to the modelling period for uranium tailings of 80'000 years (Dones et al. 2009).

In the landfill model two separate sets of transfer coefficients are produced. The transfer coefficients for a 100 year time horizon (short-term) and the transfer coefficients for the full 60'000 year modelling period. In the inventory emissions during the first 100 years are distinguished from the subsequent period up to 60'000 years. The goal of this distinction in LCI is to be able to produce contribution analyses in LCIA, and *not* to exclude long-term emissions in LCIA. Exclusion of long-term emissions is in violation of good LCA practice and ISO standards, as the relevant burdens of landfilling are ignored.

⁷ In such regions ecological planning horizons might be defined by very rare events like climate changes after meteorite impact (cf. Cretaceous-Tertiary impact winter 65 million years ago) or by very slow processes (glaciation after continental drift). For example, about 400 million years ago most regions that are tropical today were close to the south pole. However, currently tropical regions drift towards the equator and not towards the poles.

3.3 Future development of leachate concentrations

Solubility of elements chiefly depends on pH and redox potential. A decrease of pH, like observed from Acid Rock Drainage, can lead to increased concentrations of pollutants. Such changes in pH and the effects on leachate concentrations are heeded in other landfill models (Doka 2009). Whether a pH drop actually occurs depends on the buffering characteristics of the landfilled material, especially carbonates. The waste materials from coal and lignite mining and preparation tend to have low buffering capacities compared to the acidifying potential of contained sulfide minerals; thus a drop in pH is typical to occur for these materials. However, these types of landfills are very large in scale and only a part of the landfilled material is close to the surface. This means that for a large part of the landfilled material the oxygen availability for sulfide oxidation is low. A significant part of the landfill will likely be submersed in groundwater, furthering anoxic conditions and also making pyrite oxidation less likely.

For the models of this report it was therefore decided not to include any pH decrease in the future. This is a conservative approach, likely to underestimate the realistic long-term emissions. Pyrite oxidation can nevertheless occur near the surface and is observed. Therefore measurements of low-pH leachates were also included in the data population used to derive the landfill models. The established mean values for waste composition and leachate concentration are therefore a mixture of non-acidic and some acidic conditions. The average pH of all considered data points remains at approximately 6. In the field, pH drops down to 2 are observed.

3.4 Fires

Depending on the inhomogeneity of the mined vein, the spoil produced from coal and lignite mining can contain burnable organic carbon. This carbon can spontaneously ignite and lead to fires on heaps or smouldering within heaps. Tailings from milling of raw coal contain also significant coal fractions. When sufficiently dry this can also pose a fire or smoulder risk. Such fires are hard or impossible to control or terminate, can reach temperatures of 500°C and are known to continue for decades. In the German Ruhr region at least seven coal spoil heaps are known to smoulder (Mrasek 2005), some of which are still being used for coal mining operations, e.g. the spoil heap Grosses Holz in Bergkamen, Germany. Environmental impacts of smouldering heaps are air emissions from oxygen-deficient, incomplete incineration. Emissions of carbon monoxide and coking smells are detrimental and a nuisance to nearby residents.

The topic of burning waste heaps is comparable to burning coal or lignite veins. While being of relevance for local surroundings, vein fires are deemed hardly relevant in a life cycle perspective of the whole fuel chain for most mining regions, except China, see chapters 6.5 and 15.3 in (Dones et al. 2007). It is presumed that burning waste heaps in lignite or coal production are even less relevant than vein fires, that they are vastly reduced with current spoil management and are therefore not quantified here.

3.5 Failures

In case of extreme weather spoil heaps and tailings impoundments can become unstable and be eroded mechanically, thus emitting part of the contents into the environment (see e.g. Kite 2009). In one of the largest such disasters in U.S. history, a coal impoundment dam failure in 2000 released 1.16 million cubic meters of coal sludge into nearby waterways, polluting 400 km of rivers, killing 1.6 million fish and contaminating water supply for 27'000 residents in the Martin County, Kentucky. Between 2000 and 2008 in total 43 coal impoundment failures and incidents were reported in the USA (CIP 2004).

As this is an issue of risk assessment and it is deemed that such failures are comparatively rare, impacts of failures are not included in these inventories. It is assumed – as in the inventories for other

landfills and tailings impoundments in ecoinvent – that the landfill is *mechanically stable* over the whole modelled period and that only chemical weathering and leaching occurs.

3.6 Other expenditures

Energy demand for the handling of waste and dump site land use is already inventoried in the lignite and coal mining datasets of the ecoinvent database (Dones et al. 2007) and not repeated in the disposal datasets. The disposal datasets only contain emission exchanges.

3.7 Lignite spoil disposal model

Data for lignite spoil composition and leachate concentrations are collected in the literature survey. Over 1100 data points were analysed to derive a landfill model specific to the behaviour of lignite spoil observed in the field.



Fig. 3.1 Backfilling of spoil in the German open cast lignite mine Hambach (Billaudelle 2008).

Literature data

Tab. 3.1 lists the geometric means of the lignite spoil solids composition and of the leachate concentrations as established from the literature survey.

Tab. 3.1 Mean values of the literature survey for lignite spoil composition (in mg/kg) and lignite spoil leachate concentrations (in mg/λ).

Element	Solids composition (mg/kg)	Data points	Geometric standard deviation GSD	Leachate concentration (mg/λ)	Data points	Geometric standard deviation GSD
C org	7879	11	253%	27.54	15	144%
S	4832	49	366%	960.1	21	370%
N	-	0	-	-	0	-
P	275	33	208%	339.4	2	163%
Cl	-	0	-	61.79	6	132%
F	-	0	-	-	0	-
Ag	-	0	-	-	0	-
As	10.06	2	156%	0.0106	18	439%
Ba	218.5	30	203%	0.05951	19	268%
Cd	0.1619	1	-	0.001755	16	3392%
Co	3.476	33	274%	0.06947	22	854%
Cr	46.61	31	183%	0.02245	21	1807%
Cu	1.802	26	239%	0.03651	21	1527%
Hg	-	0	-	-	0	-
Mn	142	38	322%	10.55	21	447%
Mo	3.358	1	-	0.01561	16	387%
Ni	16.08	33	193%	0.391	25	878%
Pb	4.358	27	269%	0.001032	14	465%
Sb	0.5731	1	-	0.00111	12	324%
Se	-	0	-	-	0	-
Sn	2.258	1	-	0.0003378	9	232%
V	35.64	26	253%	0.01163	12	382%
Zn	25.02	37	295%	0.3126	21	1728%
Be	1.081	1	-	0.004316	8	558%
Sc	4.448	26	183%	0.005785	13	158%
Sr	61.92	26	167%	5.856	17	140%
Ti	2259	31	163%	0.02616	7	202%
Tl	0.3327	1	-	0.0002496	12	173%
W	0.035	0 (est)	-	0.0005275	12	431%
Si	380300	25	113%	27.88	2	476%
Fe	7108	25	223%	4.656	24	7289%
Ca	3227	48	386%	555.8	26	136%
Al	22240	31	155%	1.4	17	2394%
K	5981	25	192%	21.99	22	197%
Mg	1779	25	287%	221.6	23	251%
Na	2441	25	153%	62.59	22	197%

Leachate volume

For the calculation of the effective specific annual leachate volume V_{eff} , a spoil density of 2200 kg/m³ was assumed. The average height of deposited lignite spoil bodies was established using a worldwide weighted mean of open cast vs. underground mines, with a height of 150 m for open cast mines and 30 m for underground mines, estimated from photographs of various mines in the world. The generic height for lignite spoil heaps, weighted according to annual lignite spoil generation, is 147 meters. The rainwater infiltration is calculated as a weighted mean of the infiltrations at major lignite mining sites, weighted according to lignite spoil production (see Tab. 3.2). A value of 347 mm/m²·a infiltration results for generic lignite spoil. Other parameters for the landfill model were adopted from residual material landfills. The resulting effective leachate volume V_{eff} for lignite spoil landfills is 0.000845 λ/(a·kg).

Tab. 3.2 Calculation of the generic water infiltration rate for the most important lignite spoil sites, weighted according to annual spoil production.

	Average annual lignite production Mio. t / a 1)	overburden-lignite ratio m ³ /t	Spoil production Mio. m ³ / a	Annual water infiltration mm/m ² ·a 4)
Germany	174.00	5.06	880.30	450
China	78.80	4.29	338.40	450
USA	76.60	4.29	328.90	0
Russia	78.77	1.55	122.10	450
Australia	69.33	4.29	297.70	350
Greece	65.70	4.94	324.60	450
Turkey	59.40	5.30	314.80	50
Poland	60.70	3.60	218.50	150
Czech Republic	49.23	3.49	171.60	550
Serbia	39.73	4.29	170.60	450
Romania	31.20	6.30	196.60	550
India	29.23	4.29	125.50	300
World	911.00	4.29	3'900.00	349.3

1: Data for 2000, 2005 and 2006 from BGR 2008

2: Calculated from Ballisoy & Schiffer 2001

3: Tab. 13.1, in Dones et al. 2007

4: Data from FAO 2000, UNEP GRID 1994

avg: generic world average (recursive calculation)

Transfer coefficients

For some elements like bromium, iodine, silver, mercury, selenium and tungsten data for lignite spoil is missing to calculate transfer coefficients. For the halogens (Br, I) the transfer coefficients of fluorine were assumed. For silver, mercury, and selenium composition and leachate data from coal tailings were employed for transfer coefficient calculation. For tungsten only leachate compositions were available; additionally a tungsten solids content of 0.035 mg/kg was assumed, which leads to reasonable transfer coefficients for this element.

Tab. 3.3 Transfer coefficients of the lignite spoil landfill model.

Element	Short-term transfer coefficients	Long-term transfer coefficients
	0 - 100 years kg emitted / kg deposited	0 - 60000 years kg emitted / kg deposited
C org	0.0295%	17.7%
S	1.68%	100%
N	1.22%	99.9%
P	10.4%	100%
B	-	-
Cl	2.75%	100%
Br	6.71%	100%
F	6.71%	100%
I	6.71%	100%
Ag	11.7%	100%
As	0.00891%	5.2%
Ba	0.0023%	1.38%
Cd	0.0916%	55%
Co	0.169%	100%
Cr	0.00407%	2.41%
Cu	0.171%	100%
Hg	0.206%	100%
Mn	0.628%	100%
Mo	0.0393%	21%
Ni	0.205%	100%
Pb	0.002%	1.2%
Sb	0.0164%	9.35%
Se	0.191%	68.2%
Sn	0.00126%	0.758%
V	0.00276%	1.64%
Zn	0.106%	63.3%
Be	0.0337%	20.2%
Sc	0.011%	6.38%
Sr	0.799%	100%
Ti	0.0000978%	0.0587%
Tl	0.00634%	3.8%
W	0.127%	53.4%
Si	0.000619%	0.372%
Fe	0.00553%	3.32%
Ca	1.45%	100%
Al	0.000532%	0.319%
K	0.031%	17%
Mg	1.05%	100%
Na	0.216%	72.7%

3.8 Coal tailings disposal model

Data for coal tailings composition and leachate concentrations are collected in the literature survey. Over 1300 data points were analysed to derive a landfill model specific to the observed behaviour of coal tailings.

Literature data

Tab. 3.4 lists the geometric means of the coal tailings solids composition and of the leachate concentrations as established from the literature survey.

Tab. 3.4 Results of the literature survey for coal tailings composition (in mg/kg) and coal tailings leachate concentrations (in mg/λ).

Element	Solids composition (mg/kg)	Data points	Geometric standard deviation GSD	Leachate concentration (mg/λ)	Data points	Geometric standard deviation GSD
C org	124000	2	102%	10.71	2	118%
S	5562	8	1040%	431.5	71	603%
N	42.66	1	-	6.207	22	843%
P	689.6	3	132%	0.07357	3	457%
Cl	445	5	135%	236.2	11	1094%
F	1.86	1	-	1.53	1	-
Ag	0.00354	1	-	0.004923	4	2454%
As	4.451	13	2273%	0.01813	28	518%
Ba	65.52	17	2368%	0.1693	11	688%
Cd	0.6397	16	584%	0.03738	26	2619%
Co	3.429	13	539%	0.07629	25	1220%
Cr	26.07	18	430%	0.01763	25	1278%
Cu	19.39	24	455%	0.03763	73	495%
Hg	0.0486	4	973%	0.001183	5	131%
Mn	72.54	20	965%	7.48	78	1700%
Mo	1.335	7	558%	0.04606	10	398%
Ni	16.29	16	758%	0.359	74	887%
Pb	18.38	25	910%	0.003011	25	1301%
Sb	0.1281	3	353%	0.004525	10	378%
Se	0.7467	7	1072%	0.01686	12	587%
Sn	-	0	-	-	0	-
V	8.684	13	1358%	0.004873	6	224%
Zn	54.27	24	833%	0.7484	73	1801%
Be	0.5928	7	205%	0.02697	8	840%
Sc	-	0	-	-	0	-
Sr	19.38	12	549%	1.25	20	273%
Ti	137.3	7	9741%	0.01112	2	282%
Tl	-	0	-	0.0002632	4	140%
W	0.035	0 (est)	-	-	0	-
Si	239.1	11	2047%	2.889	7	294%
Fe	6103	20	1341%	5.074	63	3250%
Ca	1101	14	826%	175.3	77	409%
Al	3130	15	2194%	1.352	56	2017%
K	1317	13	1139%	25.12	21	421%
Mg	858.6	14	1141%	84.96	64	683%
Na	685.5	13	464%	238.4	21	367%

Leachate volume

It is assumed that the tailings are disposed in a separate impoundment on the surface. A landfill thickness of 30 m is assumed. For calculation of the specific leachate volume V_{eff} , a tailings density of 2200 kg/m³ was assumed. The rainwater infiltration is calculated as a weighted mean of the infiltrations at major coal mining sites, weighted according to coal tailings production. A value of 420 mm/m²·a infiltration results for generic coal tailings impoundments. Other parameters were adopted from residual material landfills. The resulting effective leachate volume V_{eff} for coal tailings impoundments is 0.00497 λ/(a·kg).

Tab. 3.5 Calculation of the generic water infiltration rate for the most important coal tailings sites, weighted according to annual tailings production.

	Average annual coal production Mio. t / a 1)	Tailings-coal ratio t / t coal Dones et al. 2007, p.38	Annual tailings production Mio. t / a	Annual water infiltration mm/m ² ·a
China	1881	0.155	291.5	400
USA	948.5	0.415	393.6	550
India	369.2	0.271	100.1	550
Australia	283.1	0.309	87.48	250
South Africa	239.3	0.27	64.6	0
Russia	204.3	0.271	31.67	250
Indonesia	142.9	0.271	38.73	450
Poland	98.57	0.271	26.71	150
Germany	9.867	0.461	4.549	150
Czech Republic	13.87	0.271	3.758	550
World	4690	0.25	1'167	420

1: Data for 2000, 2005 and 2006 from BGR 2008

Hydrocarbon pollutants

Coal tailings are produced from mechanical separation of conglomerate minerals from fossilised carbon. The organic carbon content in coal tailings is therefore high. Schüring (1996) gives figures of the organic carbon content in coal tailings in the vicinity of 12% by weight. Limited information on hydrocarbon concentrations in the coal tailings leachate are available from (WVDEP 2009).

Tab. 3.6 Concentrations of organic pollutants in coal tailings liquid in West Virginia, USA, from (WVDEP 2009). Only values above detection limit shown.

Concentrations in coal slurry (=coal tailings) liquid in West Virginia, USA	ppm (mg/λ)
Oil and Grease	2.2
Total Petrol Hydrocarbons TPH, Oil Range	4.16
Total Petrol Hydrocarbons TPH, Diesel Range	0.685
2-Butanone	0.0684
Acetone	0.0167
2-Methyl 2-propane	0.0054
1-Butanol	0.0034
Toluene	0.0028
Benzene	0.0018
Methyl butane	0.0012
m,p-Xylene	0.0008
o-Xylene	0.0006

Compared to the inorganic pollutants in coal tailings leachate, the organic pollutants listed in Tab. 3.6 contribute only unnoticeably to the toxicity of average coal tailings leachate (with Eco-indicator'99 characterisation factors). Due to the low relevance of the quoted emissions they are not inventoried in the dataset. Apart from low data population and remaining gaps for other substances a sub-problem would be the modelling of organic pollutants and their possible microbial decomposition over long time frames in the landfill model. A potentially relevant data gap are concentrations of polyaromatic hydrocarbons (PAH) in leachate, which might have a perceptible influence on leachate toxicity. PAHs are present in the tailings solids (see Tab. 3.7), but no information on their transfer to tailings leachate is available.

Tab. 3.7 Concentrations of organic pollutants in coal tailings solids in West Virginia, USA, from (WVDEP 2009). Only values above detection limit shown.

Concentrations in coal slurry (=coal tailings) solids in West Virginia, USA	ppm (mg/kg)
Total Petrol Hydrocarbons TPH, Oil Range	159
Total Petrol Hydrocarbons TPH, Diesel Range	144
Naphthalene	0.259
Benzo(a)anthracene	0.036
Benzo(a)pyrene	0.07
Benzo(b)fluoranthene	0.082
Benzo(g,h,i)perylene	0.155
Chrysene	0.206
Dibenzo(a,h)anthracene	0.032
Fluoranthene	0.07
Fluorene	0.202
Naphthalene	1.5
Phenanthrene	0.903
Phenol	0.045
Pyrene	0.095
Naphthalene, 1,3-dimethyl	1.28
Naphthalene, 1,4-dimethyl	1.08
Naphthalene, 1-methyl	1.09
Benzene	0.166
Ethylbenzene	0.122
Isopropylbenzene	0.0302
Sec-Butylbenzene	0.0085
n-Propylbenzene	0.0455
1,2,4-Trimethylbenzene	0.216
1,3,5-Trimethylbenzene	0.0768
Toluene	1.04
m,p-Xylene	0.585
o-Xylene	0.284
2,4-Dimethylphenol	0.167
2-Ethyl hexanal	0.433
Methyl butane	0.107
Methyl pentane	0.133
Butane	0.135
Cyclohexane	0.333
Hexane	0.0735
Isobutene	0.0948
Methyl cyclohexane	0.58
Methyl cyclopentane	0.081
Pentane	0.0998
Butane, 1,1 -dibutoxy	1.07
Butanioc acid, butyl ester	1.32
Hexadecane	0.683
Pentadecane 2,6,10,14 tetramethyl	1.34
Propanoic acid, 2-methyl, 2-ethyl	1.05
Tetradecane	0.886
Tridecane	1.01

Radioactive pollutants

Only very limited data is available for radioactive species in coal tailings (Reichelt & Sitte 2004). The geometric means of Ra-226 and U-238 in coal tailings solids were found to be 856 and 212 Bq/kg, respectively. With additional proxy data from mine water concentrations it is estimated that these two radioactive species in tailings make up below 1% of the total long-term emission burden as measured with Eco-indicator'99. Due to the weak data population and the seemingly low relevance, these emissions are not inventoried here. This preliminary finding shall not prejudice the future investigation of radioactive pollutants in coal tailings for LCA, as the scope of valuation of radioactive water pollutants in Eco-indicator'99 is small.

Transfer coefficients

For some elements like bromium, iodine, tin, scandium, thallium, and tungsten data from coal tailings is missing to calculate transfer coefficients. For the halogens (Br, I) the transfer coefficients of fluorine

were assumed. For tin, scandium, thallium, and tungsten composition and leachate data from lignite spoil was employed for transfer coefficient calculation.

Tab. 3.8 Transfer coefficients of the coal tailings impoundment model.

Element	Short-term transfer coefficients	Long-term transfer coefficients
	0 - 100 years kg emitted / kg deposited	0 - 60000 years kg emitted / kg deposited
C org	0.00429%	2.57%
S	3.85%	100%
N	6.97%	100%
P	0.0053%	3.18%
B	-	-
Cl	23.2%	100%
Br	33.5%	100%
F	33.5%	100%
I	33.5%	100%
Ag	69.1%	100%
As	0.202%	70.3%
Ba	0.128%	77%
Cd	2.9%	100%
Co	1.11%	100%
Cr	0.0336%	18.3%
Cu	0.0964%	57.8%
Hg	1.21%	100%
Mn	5.12%	100%
Mo	1.7%	100%
Ni	1.09%	100%
Pb	0.00814%	4.88%
Sb	1.74%	100%
Se	1.12%	99.9%
Sn	0.00743%	0.758%
V	0.0279%	15.4%
Zn	0.685%	100%
Be	2.26%	100%
Sc	0.0646%	6.38%
Sr	3.2%	100%
Ti	0.00402%	2.41%
Tl	0.0378%	3.85%
W	0.746%	53.4%
Si	0.6%	100%
Fe	0.0413%	24.8%
Ca	7.91%	100%
Al	0.0215%	12.9%
K	0.943%	99.7%
Mg	4.92%	100%
Na	15.9%	100%

3.9 Coal spoil disposal model

Data on coal spoil compositions is scarce in the literature survey. For the purposes of this report, composition and leachate concentrations of coal spoil are adopted by proxy from lignite spoil. Since lignite is a geological precursor to coal, the characteristics of coal overburden and lignite overburden are on average likely to coincide. This approximation is further supported by the observation that the characteristics of coal tailings and lignite spoil collected in the literature survey are – within the bounds of geological variability – comparable. On average mineral phases in or near lignite veins are quite similar to the mineral phases in or near coal veins. This means that coal spoil can well be approximated with lignite spoil.

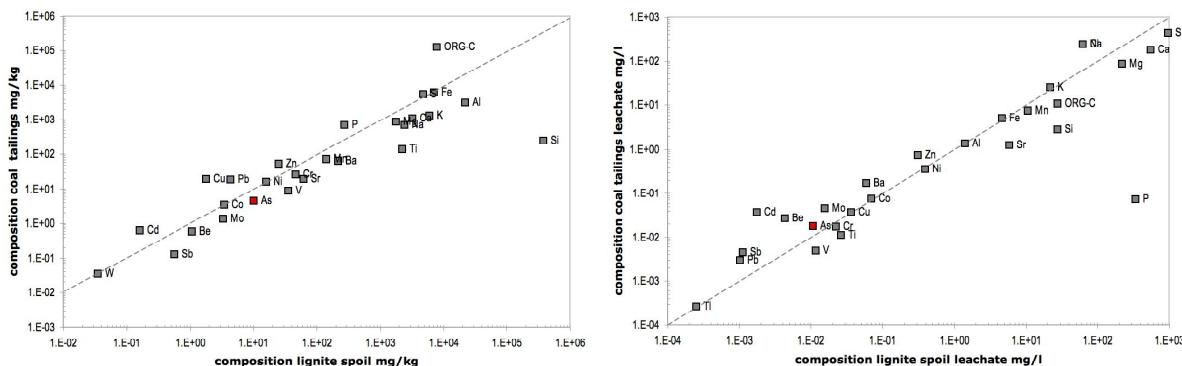


Fig. 3.2 Correlation plots of lignite spoil (x) vs. coal tailings (y) for solids composition (left) and leachate data (right), showing the similarities within the bounds of geological variability of the two materials. The point in red is for arsenic, as the environmentally most relevant constituent.

This approximation only relates to the composition of spoil and leachate concentrations in spoil. For the coal spoil model the annual leachate volume based on the world's major coal mining sites is applied, not the one based on lignite sites.

Leachate volume

For calculation of the specific leachate volume V_{eff} , a spoil density of 2200 kg/m³ was assumed. The average height of deposited coal spoil bodies was established using a worldwide weighted mean of open cast vs. underground mines, with a height of 200 m for open cast mines and 30 m for underground mines, estimated from photographs of various mines in the world. The generic height for coal spoil heaps, weighted according to annual coal spoil generation, is 130 meters. The rainwater infiltration is calculated as a weighted mean of the infiltrations at major coal mining sites, weighted according to coal spoil production. A value of 371 mm/m²·a infiltration results for generic coal spoil. Other parameters for the landfill model were adopted from residual material landfills. The resulting effective leachate volume V_{eff} for coal spoil landfills is 0.00101 $\lambda/(a \cdot kg)$.

Tab. 3.9 Calculation of the generic water infiltration rate for the most important coal spoil sites, weighted according to annual spoil production.

	Average annual coal production Mio. t / a 1)	overburden-coal ratio t / t coal	Spoil production Mio. t / a	Annual water infiltration mm/m ² ·a
China	1881	1.64	3084	400
USA	948.5	6.11	5796	550
India	369.2	4.9	1809	550
Australia	283.1	17.15	4855	250
South Africa	239.3	7.7	1842	0
Russia	204.3	7.4	1512	250
Indonesia	142.9	5	714.7	450
Poland	98.57	2.15	211.9	150
Germany	9.867	1.68	16.58	150
Czech Republic	13.87	1.915	26.55	550
World	4690	4.74	22'240	371

1: Data for 2000, 2005 and 2006 from BGR 2008

2: Ritthoff et al. 2007. Tab. 4 on "abiotic material input", accompanying text describes this data as spoil. Value for Germany of 1.47 in approximate accordance to value from (Reichelt & Sitte 2004) of 1.68.

3: Own estimate

4: Reichelt & Sitte 2004

5: Average of Germany and Poland

Transfer coefficients

Tab. 3.10 Transfer coefficients of the coal spoil landfill model.

Element	Short-term transfer coefficients	Long-term transfer coefficients
	0 - 100 years kg emitted / kg deposited	0 - 60000 years kg emitted / kg deposited
C org	0.0355%	21.3%
S	2.02%	100%
N	1.47%	100%
P	12.5%	100%
B	-	-
Cl	3.3%	100%
Br	8.01%	100%
F	8.01%	100%
I	8.01%	100%
Ag	14.1%	100%
As	0.0107%	6.22%
Ba	0.00276%	1.66%
Cd	0.11%	66%
Co	0.203%	100%
Cr	0.00489%	2.89%
Cu	0.205%	100%
Hg	0.247%	100%
Mn	0.754%	100%
Mo	0.0471%	24.6%
Ni	0.247%	100%
Pb	0.0024%	1.44%
Sb	0.0197%	11.1%
Se	0.229%	74.7%
Sn	0.00152%	0.911%
V	0.00331%	1.97%
Zn	0.127%	76%
Be	0.0405%	24.3%
Sc	0.0132%	7.61%
Sr	0.959%	100%
Ti	0.000117%	0.0705%
Tl	0.00761%	4.57%
W	0.153%	60%
Si	0.000744%	0.446%
Fe	0.00665%	3.99%
Ca	1.75%	100%
Al	0.000639%	0.383%
K	0.0373%	20%
Mg	1.26%	100%
Na	0.26%	79%

3.10 Inventory tables

The inventoried emissions for each landfill are obtained by multiplying the solids composition with the transfer coefficients for each element. In the inventory the long-term emissions are the

complement to the short-term emissions, i.e. representing the time frame 100 to 60'000 years. For the inventory S, N, P are converted to the appropriate weights of sulfate, nitrate, phosphate. Chromium is inventoried as the soluble species (Cr^{VI}). All emissions are inventoried as emissions to groundwater.

Tab. 3.11 Unit process raw data for the disposal of lignite spoil, coal tailings, and coal spoil (corrected in March 2021)

Name	Category	SubCategory	Unit	<i>disposal, spoil from lignite mining, in surface landfill</i>	<i>disposal, tailings from hard coal milling, in impoundment</i>	<i>disposal, spoil from coal mining, in surface landfill</i>
				<i>GLO 0 kg</i>	<i>GLO 0 kg</i>	<i>GLO 0 kg</i>
Location InfrastructureProcess Unit						
Sulfate	water	ground-	kg	0.000243295	0.000643164	0.000292169
Nitrate	water	ground-	kg	2.30764E-06	1.31737E-05	2.7678E-06
Phosphate	water	ground-	kg	8.79167E-05	1.12085E-07	0.000105578
Boron	water	ground-	kg	0	0	0
Chloride	water	ground-	kg	1.22565E-05	0.000103141	1.46776E-05
Bromine	water	ground-	kg	0	0	0
Fluoride	water	ground-	kg	1.24851E-07	6.23946E-07	1.48902E-07
Iodide	water	ground-	kg	0	0	0
Silver, ion	water	ground-	kg	4.15876E-10	2.4459E-09	4.99419E-10
Arsenic, ion	water	ground-	kg	8.95643E-10	8.99868E-09	1.07555E-09
Barium	water	ground-	kg	5.02693E-09	8.41109E-08	6.03675E-09
Cadmium, ion	water	ground-	kg	1.48271E-10	1.85684E-08	1.78057E-10
Cobalt	water	ground-	kg	5.86851E-09	3.78982E-08	7.0474E-09
Chromium VI	water	ground-	kg	1.89661E-09	8.75497E-09	2.2776E-09
Copper, ion	water	ground-	kg	3.08416E-09	1.86935E-08	3.70372E-09
Mercury	water	ground-	kg	9.99241E-11	5.87686E-10	1.19997E-10
Manganese	water	ground-	kg	8.91284E-07	3.71578E-06	1.07033E-06
Molybdenum	water	ground-	kg	1.31817E-09	2.26883E-08	1.58291E-09
Nickel, ion	water	ground-	kg	3.30287E-08	1.78327E-07	3.96637E-08
Lead	water	ground-	kg	8.71955E-11	1.49561E-09	1.04712E-10
Antimony	water	ground-	kg	9.37863E-11	2.22819E-09	1.12625E-10
Selenium	water	ground-	kg	1.42291E-09	8.32977E-09	1.70842E-09
Tin, ion	water	ground-	kg	2.85351E-11	1.67824E-10	3.42673E-11
Vanadium, ion	water	ground-	kg	9.82029E-10	2.4205E-09	1.1793E-09
Zinc, ion	water	ground-	kg	2.64035E-08	3.71776E-07	3.17075E-08
Beryllium	water	ground-	kg	3.64599E-10	1.33995E-08	4.37841E-10
Scandium	water	ground-	kg	4.88602E-10	2.87286E-09	5.86748E-10
Strontium	water	ground-	kg	4.94653E-07	6.20866E-07	5.94021E-07
Titanium, ion	water	ground-	kg	2.20971E-09	5.5224E-09	2.65361E-09
Thallium	water	ground-	kg	2.10832E-11	1.25655E-10	2.53185E-11
Tungsten	water	ground-	kg	4.45259E-11	2.61059E-10	5.34636E-11
Silicon	water	ground-	kg	2.35521E-06	1.43504E-06	2.82834E-06
Iron, ion	water	ground-	kg	3.93303E-07	2.5207E-06	4.72311E-07
Calcium, ion	water	ground-	kg	4.69473E-05	8.70838E-05	5.63783E-05
Aluminum	water	ground-	kg	1.18278E-07	6.71613E-07	1.42038E-07
Potassium, ion	water	ground-	kg	1.85722E-06	1.24222E-05	2.23023E-06
Magnesium	water	ground-	kg	1.87197E-05	4.22081E-05	2.24802E-05
Sodium, ion	water	ground-	kg	5.28133E-06	0.000108755	6.34089E-06
Sulfate	water	ground-, long-term	kg	0.014252049	0.016042766	0.014203175
Nitrate	water	ground-, long-term	kg	0.000186497	0.000175749	0.000186128
Phosphate	water	ground-, long-term	kg	0.000755237	6.71387E-05	0.000737576
Boron	water	ground-, long-term	kg	0	0	0
Chloride	water	ground-, long-term	kg	0.000432748	0.000341863	0.000430327
Bromine	water	ground-, long-term	kg	0	0	0
Fluoride	water	ground-, long-term	kg	1.73515E-06	1.23605E-06	1.7111E-06
Iodide	water	ground-, long-term	kg	0	0	0
Silver, ion	water	ground-, long-term	kg	3.12412E-09	1.0941E-09	3.04058E-09
Arsenic, ion	water	ground-, long-term	kg	5.22409E-07	3.12027E-06	6.24021E-07
Barium	water	ground-, long-term	kg	3.01113E-06	5.03824E-05	3.61602E-06
Cadmium, ion	water	ground-, long-term	kg	8.88145E-08	6.21106E-07	1.06656E-07
Cobalt	water	ground-, long-term	kg	3.47E-06	3.39115E-06	3.46882E-06
Chromium VI	water	ground-, long-term	kg	1.12231E-06	4.7496E-06	1.34447E-06
Copper, ion	water	ground-, long-term	kg	1.79926E-06	1.11974E-05	1.79864E-06
Mercury	water	ground-, long-term	kg	4.8499E-08	4.80112E-08	4.84789E-08
Manganese	water	ground-, long-term	kg	0.000141126	6.88242E-05	0.000140947
Molybdenum	water	ground-, long-term	kg	7.03477E-07	1.31206E-06	8.25849E-07
Nickel, ion	water	ground-, long-term	kg	1.60456E-05	1.61151E-05	1.60389E-05
Lead	water	ground-, long-term	kg	5.22301E-08	8.95872E-07	6.27223E-08
Antimony	water	ground-, long-term	kg	5.35076E-08	1.25855E-07	6.36359E-08
Selenium	water	ground-, long-term	kg	5.07524E-07	7.37466E-07	5.56069E-07
Tin, ion	water	ground-, long-term	kg	1.70925E-08	1.69532E-08	2.05261E-08
Vanadium, ion	water	ground-, long-term	kg	5.834E-07	1.3351E-06	6.99435E-07
Zinc, ion	water	ground-, long-term	kg	1.58157E-05	5.38945E-05	1.89928E-05
Beryllium	water	ground-, long-term	kg	2.18395E-07	5.79436E-07	2.62267E-07

4 Result discussion

The resulting emission inventories are shown in Tab. 3.11 on page 22. Emissions of arsenic (As), which contributes dominantly to toxicity burdens, are a factor 5 to 6 larger for coal tailings than for the spoil materials. This is the result of a larger mobility of arsenic in coal tailings compared to spoil, due to a combination of larger initial leachate concentrations (0.0181 vs. 0.0106 mg/λ) and a larger effective leachate volume (0.00496 vs. 0.00101 λ/(a·kg)).

As an intermediate result of the modelling, transfer coefficients for chemical elements were calculated. These can be compared with the transfer coefficients of a residual material landfill⁸ from (Doka 2009). The transfer coefficients express the relation of emitted mass over the initially deposited mass and can be understood as a parameter for the relative mobility of an element within a landfill body. The comparison therefore highlights *differences in mobility behaviour of elements* in different landfill settings, see Fig. 4.1, Fig. 4.2 and Fig. 4.3 below. Some elements appear to have a lesser mobility within all three types of landfills compared to a residual material landfill. Several oxyanion-forming elements are distinctly clustered together (red areas). Oxyanions are anions, which are commonly formed by certain elements together with oxygen, e.g. arsenic in the oxyanion arsenate HAsO_4^{2-} . Oxyanions are generally more mobile at high pH values. Since residual landfills have high pH values, but the tailings and spoil landfills tend to have lower pH values due to Acid Rock Drainage, the lesser mobility established in the models is realistic.

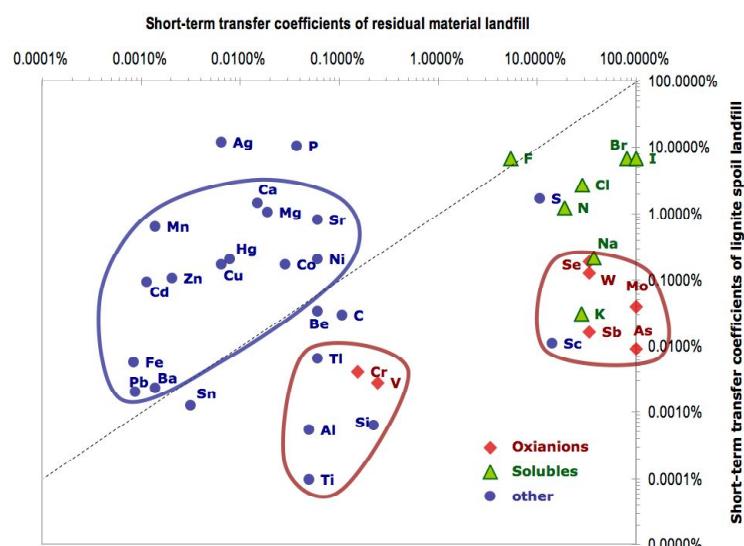


Fig. 4.1 Comparison of short-term transfer coefficients (100a) for residual material landfill (x axis) and a lignite spoil landfill (y axis).

⁸ The Swiss residual material landfill is for inert, inorganic waste materials sometimes solidified with cement.

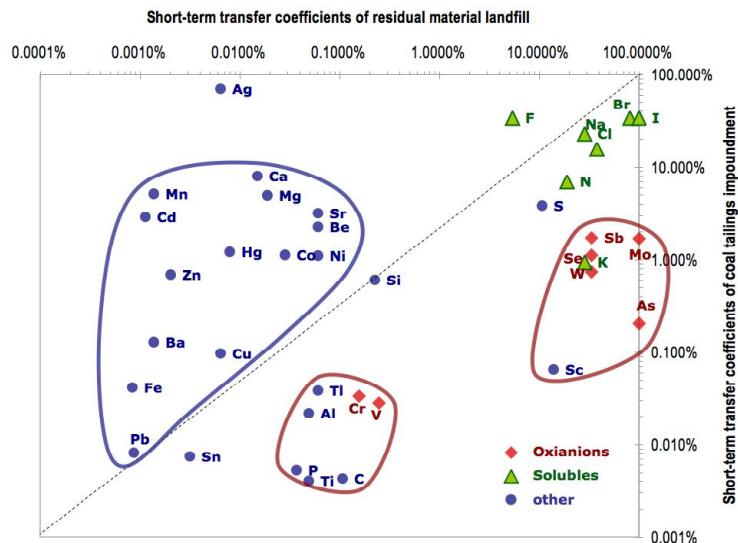


Fig. 4.2 Comparison of short-term transfer coefficients (100a) for residual material landfill (x axis) and a coal tailings impoundment (y axis).

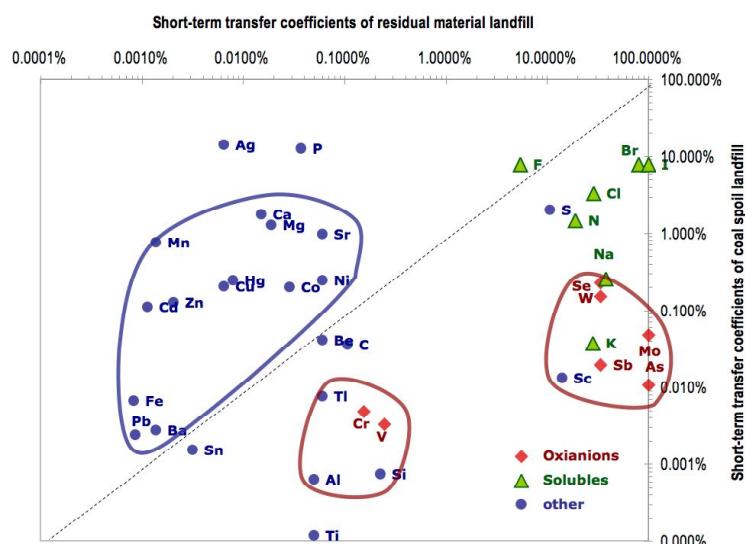


Fig. 4.3 Comparison of short-term transfer coefficients (100a) for residual material landfill (x axis) and a coal spoil landfill (y axis).

Conversely there is a cluster of elements (blue area), that contains elements that are commonly more soluble at low pH values. These are elements that usually occur as positively charged ions, i.e. cations. These cations show increased mobility which again can be explained by lower pH values due to Acid Rock Drainage.

While the uncertainties in these simplified models are high due to the inherent geochemical and climatic variability, it is an encouraging fact that they succeed in depicting chemically reasonable mobility behaviour for elements. The resulting clustering is not overly dependent on the variable nature of the employed data population. Otherwise such distinct clustering of elements could not be observed. This is a sign that the sample size of the employed data population is large enough to represent the characteristic, generic behaviour of these waste materials.

Noteworthy in these plots is the splitting of the oxyanion clusters in two separate groups, while in similar plots for other tailings landfills (non-iron, uranium) the oxyanions are more or less fairly

within one single cluster (see Classen et al. 2009, Dones et al. 2009). In the models developed here, the oxyanions selenium, arsenic, molybdenum, antimony and tungsten form a comparatively mobile or "fast" cluster, located close to easily soluble ions (green triangles). The oxyanions chromium and vanadium form a "slow" cluster with a mobility reduced by approximately two orders of magnitude. The interpretation of this cluster separation is indeterminate. Suggested by the proximity of the "slow" Cr-V cluster to carbon (C) it could be hypothesized that mobility of Cr and V are moderated by the presence of organic carbon compounds, possibly by formation of organometallic compounds. Another interesting finding is the association of scandium (Sc) with the "fast" oxyanion cluster. Scandium is usually known as cationic species in aquatic environments⁹, and therefore *expected* to appear in the cluster of cations (blue area). Thus in addition to creating an LCI for waste disposal, the investigation yields following predictions or hypotheses:

1. Tungsten content in lignite spoil, coal tailings and coal spoil is expected to be in the vicinity of 0.035 mg/kg (see explanatory text in chapter Transfer coefficients on page 13).
2. The mobility of chromium and vanadium is expected to be reduced, possibly owing to the large concentration of carbon in these waste materials, possibly owing to formation of organometallic compounds or adsorption on aromatic structures on coal solid surfaces.
3. Scandium atypically does not behave like the trivalent cation it is expected to form in aqueous solution. It has a mobility behaviour like an oxyanion. This might be explained by formation of an unknown (oxy-)anionic species, or a delaying effect by formation of organometallic compounds/adsorption on aromatic structures.

⁹ Scandium is not known to form oxyanions in aqueous solution, though the salts of "scandic acid" are known as solids, e.g. NaScO₂. The anionic scandium hydroxide Sc(OH)₄⁻ is a dominant species at alkaline pH above 10 in aqueous solution; conditions which are unlikely to occur in these waste materials.

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