

Life Cycle Inventories of Waste Treatment Services

Part II " Landfills - Underground deposits - Landfarming"

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Summary

In this study life cycle inventories of waste disposal in landfills, underground deposits and landfarming are established. As for waste incineration in part II, these inventories are attempted to be as waste-specific as possible, i.e. the chemical composition of a waste is heeded in order to establish the emissions from a disposal process. So called transfer coefficients are used to determine how much of a chemical element is emitted from a disposal process. The functional unit of landfill disposal is kilogram of waste.

It is known that landfills emit pollutants for centuries or millennia. **The future emissions of landfills have to be modelled, as no measurements of them are available.** This report develops landfill models that generate life cycle inventories for the disposal of specific single waste fractions, i.e. not just average waste. The models are based on Swiss landfills and are based on the concept of using waste composition data and transfer coefficients for different chemical elements to calculate emissions. A new approach to estimate long-term emissions was developed. **The landfill models in this study cover a time span of 60'000 years.** Clearly, modelling such long time spans inherently introduces large uncertainties, of which one should be cautious. However, the analysis also showed that uncertainties in *landfill development* are not necessarily the largest contributions to the total uncertainty in the end result. Additional large uncertainties are introduced by the *variation in the composition* of the landfilled waste, especially for trace metals. **For long-term emissions the uncertainty in waste composition is often more important than the uncertainty in landfill development** as expressed by transfer coefficients. Emissions were separated into short-term emissions occurring over the first 100 years, and long-term emissions occurring 100 to 60'000 years after waste placement. This allows for sensitivity analysis for determining the influence of the long-term emissions on end results, which is in most cases considerable.

In the sanitary landfill model the degradability of waste fractions is used to calculate the short-term behaviour of the waste in a waste-specific manner. New data was used to derive degradability of wastes. **New in the sanitary landfill model is the introduction of a so called release factor, which heeds the re-precipitation of degraded material within the landfill.** Different release factors are calculated for each chemical element and are calibrated according to field measurements of actually occurring landfill emissions. Also heeded in the model is **preferential flow of leachate transport** through the landfill body.

The waste composition vector is extended to include additional chemical elements (beryllium, scandium, strontium, titanium, thallium, tungsten) compared to earlier studies from the ETH domain. The new landfill models heed emissions of 41 different chemical elements, as compared to 14 in (Zimmermann et al. 1996, Hellweg 2000). **Landfill behaviour data (transfer coefficients) is given for all these chemical elements in all landfill models, in order to avoid the gaps of previous models.** This diminishes the risk of overlooking emissions, that might become relevant for certain wastes containing those previously neglected elements. Examples show that this assessment gap can be very relevant, e.g. for hard coal ashes.

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1 Introduction

In this part inventories of waste disposal in landfills, underground deposits and landfarming are established. As for waste incineration in part II, these inventories are attempted to be as *waste-specific* as possible, i.e. the chemical composition of a waste is heeded in order to establish the emissions from a disposal process. So called transfer coefficients are used to determine how much of a chemical element is emitted from a disposal process, i.e. in general an emission is derived from the multiplication of waste composition with the transfer coefficients. The transfer coefficients can be thought of a descriptor of the behaviour of chemical elements in a disposal process.

In case of sanitary landfills (municipal landfills for untreated solid waste) these transfer coefficients are further modified according to the degradability of the waste. For hardly decomposable wastes emissions are delayed into the future.

Also for sanitary landfills *further downstream processes* occur: the landfill leachate of the first 100 years after deposition is assumed to be collected and purified in a municipal wastewater treatment plant. The latter process produces in turn a treatment sludge which is incinerated in a municipal waste incinerator. This last step produces incineration residues which are landfilled in slag compartments and residual material landfills. All these additional downstream processes are heeded *waste-specifically* in the inventory of sanitary landfill disposal, based on the inventory models of wastewater treatment (part III), municipal waste incineration (part II) and landfills (this part).

The current model heeds only chemical elements as direct emissions. For some elements a speciation profile is established (e.g. ammonia, nitrate, nitrite etc. for emissions of nitrogen to water). The fate of individual chemical compounds (e.g. hexachlorobenzene) is *not* modelled in these inventories. All waste compositions are only heeded as a vector of 41 chemical elements plus water.

Underground deposits for hazardous waste are located in old salt mines. The risk of flooding of underground deposits was investigated. It was decided however not to heed any direct emissions from underground deposits in the inventory. Differences in burdens result from different waste packaging for storage.

2 Landfill Capacities

2.1 Sanitary landfill capacities

In Switzerland 49 sanitary landfills were in operation in 2001 (BUWAL 2001c). The remaining free landfill capacity is 5.8 million cubic meters. An additional 2.7 million cubic meters are planned (BUWAL 2002a). These numbers exclude landfill volume for residual material or slags. In 2000 the mass of waste to landfills (excluding residual material or slags) was 1.3 million tons or 800'000 m³ per year (BUWAL 2001e). The static reserve time of sanitary landfill volume is 11 years. However, roughly a third of the currently landfilled waste is burnable waste, which will be incinerated in the short-term future.

2.2 Slag compartment capacities

In Switzerland 31 slag landfills or sanitary landfills with a slag compartment were in operation in 2001 (BUWAL 2002a). The remaining free slag compartment capacity is 5.7 million cubic meters. An additional 8 million cubic meters are planned (BUWAL 2002a). With an approximate landfilled volume of 500'000 m³ of slag per year, the static reserve time is 27 years.

2.3 Residual material landfill capacities

In Switzerland 13 residual material landfills or sanitary landfills with a residual material compartment were in operation in 2001 (BUWAL 2001d). The remaining free landfill capacity is 790'000 cubic meters. An additional 2 million cubic meters are planned (BUWAL 2002a). With an approximate landfilled volume of 50'000 m³ of residual materials per year, the static reserve time is 56 years.

2.4 Inert material landfill capacities

In Switzerland 194 inert material landfills are listed to be in operation in December 2001 (BUWAL 2001a). This is an incomplete list and there are more inert material landfills in operation¹. The free inert material landfill capacity of 148 reporting landfills² (i.e. less than 76% of all landfills) was at least 25.2 million cubic meters (BUWAL 2002b). Most waste landfilled in inert material landfills is excavation material and construction waste. Approximately 4.1 million tons of inert waste per year are landfilled (BUWAL 2002c). Assuming an average density of 1.8 ton/m³, the volume landfilled per year is 2.2 million cubic meters. The static reserve time is therefore about at least 11 years.

¹ Cf. note regarding (BUWAL 2001a) on http://www.umwelt-schweiz.ch/buwal/de/fachgebiete/fg_abfall/anlagen/deponierung.

² This number includes 4 inert compartments in municipal landfills and 1 compartment in a residual material landfill.

3 Waste Compositions

Typical compositions of municipal solid waste in Switzerland and other countries is outlined in part II on incineration of this report.

3.1 Waste in sanitary landfills

In the year 2000 the 49 Swiss sanitary landfills received 2 Million tons of waste (BUWAL 2001e). This figure includes 624'600 tons of MSWI slags. 291'700 tons were untreated municipal waste. Landfilling of burnable waste was legally phased out in 2000. Nevertheless, 390'700 tons (19.6%) of the waste landfilled, was burnable waste (municipal waste, burnable construction waste, wastewater treatment sludge and others). An overview of the waste types delivered to the landfills is shown in Tab. 3.1.

Tab. 3.1 Types of waste landfilled in Swiss sanitary landfills in 2000 (BUWAL 2001e)

Waste to Swiss sanitary landfills in 2000	Tons per year
Municipal waste	291'681
Burnable construction waste	51'567
Wastewater treatment sludge	9'270
Excavation material	271'243
Polluted excavation material	69'821
Inert material	93'047
Inorganic construction waste	14'957
Other sorted construction waste	125'395
Unsorted construction waste	8'564
Slag from MSWI	624'582
Hazardous waste	92'149
Residual material	63'040
Other	276'216
total	1'991'532

3.2 Waste in residual material landfills

Different waste materials are deposited in residual material landfills. Statistical figures on deposited amounts are difficult to obtain. From MSWI alone, 44'100 tons of fly ashes and 3160 tons of scrubber residues were deposited in residual landfills (BUWAL 2001m, BUWAL 2001n).

Also industrial wastes are deposited in residual material landfills. No literature references for this latter waste stream, nor compositions, could be found. The total amount of *landfilled hazardous wastes* was 234'407 tons in 1999 (BUWAL 2001o). Approximately 162'000 tons of hazardous wastes³ were landfilled in sanitary landfills in 2000 (BUWAL 2001e). The remainder of approximately 72'000 tons per year is landfilled in residual material landfills. A total amount of approx. 120'000 tons per year (40% from MSWI, 60% from industry) is deposited in residual material landfills.

³ The categories 'hazardous waste' (Sonderabfall) and 'polluted excavation material' (verschmutzter & tolerierbarer Aushub) from (BUWAL 2001d) were added for this figure. Excluded is an amount of 270'000 tons of unpolluted excavation material.

3.3 Waste in inert material landfills

The waste deposited in inert material landfills has generally a low pollutant content and is chemically inert to a large extent. Swiss regulation uses the expression 'similar to natural rocks' ('gesteinsähnlich') (TVA 2000). Waste fractions landfilled in inert material landfills are shown in Tab. 3.2.

Tab. 3.2 Types of waste landfilled in Swiss inert material landfills in 2000. Incomplete data (BUWAL 2002c)

Waste to Swiss inert material landfills in 2000	Tons per year
Clean excavation material	2'550'532
Polluted but tolerable excavation material	477'409
Inert materials	642'182
Construction waste	408'913
Other	25'836
Total	4'104'872

4 Waste Generation and Treatment

Generation and treatment of hazardous and municipal solid waste in Switzerland and other countries is outlined in part II on incineration of this report.

5 Systems Characterisation

5.1 Introduction

Until the middle of the 20th century European urban solid wastes were collected and usually discharged to uncontrolled and unordered waste dumps. Concerns regarding water pollution, odorous air emissions, occurrence of fire and smoke, landscape protection, occurrence of vermin lead to laws prescribing ordered landfills in many countries.

Space restrictions but also pollution concerns and lacking public acceptance of new landfills made it difficult in the last twenty years to put up new landfill sites. Some countries, e.g. the Netherlands and Switzerland, have therefore mandated that no burnable wastes shall be landfilled directly, but only by way of incineration for waste volume reduction⁴.

Switzerland knows three types of landfills. These landfills are regulated by the technical ordinance on waste (TVA 2000).

- Sanitary landfills (municipal waste landfills, organic landfill, Swiss German: 'Reaktordeponie')
- Residual material landfills (Swiss German: 'Reststoffdeponie')
- Inert material landfills ('cleanfill', Swiss German: 'Inertstoffdeponie')

Residual material landfills can be stand-alone sites or can be realised as compartments of sanitary landfills. Similarly there exists a *fourth kind* of 'landfill' that is only realised as a compartment of a sanitary landfill. These are the so called 'slag compartments' that only contain slag (bottom ash) from municipal solid waste incineration (CH German Schlackekompartiment).

While sanitary landfills can receive all kinds of biologically reactive wastes, the other types of landfills receive inorganic wastes with a varying extent of harmful contaminants. This material must be deposited in a physically stable form. Sites for all landfill types must avoid sites of high risk of landslide or flooding. Actual or potential groundwater utilisation has priority against landfill projects. All landfilled waste must be inventoried by the landfill operator. Controlled landfills in Switzerland usually feature a base and boundary sealing⁵. They feature a slanted base so gravitational water removal is secured. Sealed landfills also need to have a water collection system to secure pollutant monitoring and possible water treatment. After closure the landfilled must be monitored for several years and the surface must be restored either as agricultural or natural area (TVA 2000).

⁴ Germany plans to introduce such legislation for 2005.

⁵ For Swiss inert material landfills a seal and leachate collection system can be excluded only if they are located outside of ground water zones for drinking water.

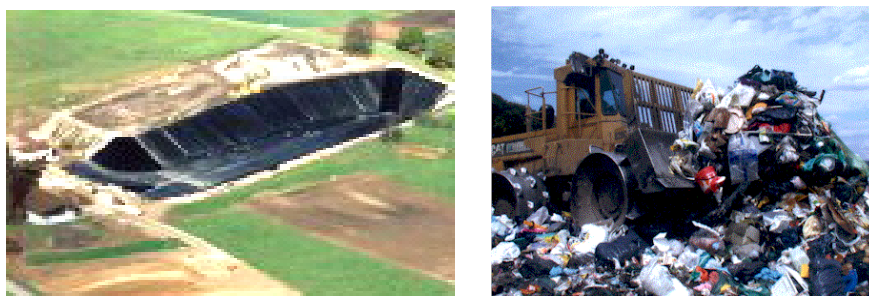


Fig. 5.1 A new and empty landfill site with visible bottom seal sheet (left) and waste placement and compaction in a sanitary landfill with special loader (right)

5.2 Sanitary landfills

Sanitary landfills are the most demanding type of landfill. They are designed to receive all kinds of waste: untreated municipal or household waste, building wastes, wastewater treatment sludge. If local capacities lack, they can also receive inert wastes or, pending special approval, hazardous wastes. Not allowed are liquid, infectious or radioactive wastes or explosives. No limiting requirements for the landfilled waste regarding composition or extraction tests results are defined. The landfill design must include base and boundary sealing, a water collection system and also a gas collection system due to the biologically reactive nature of the waste. Landfill gas is usually incinerated or converted to electricity. Restoration of the landfill site and a post-closure monitoring period of at least 15 years is mandatory. After a five-year transition period, landfilling of burnable wastes was prohibited in Switzerland in 2000⁶.

For sanitary landfills several stages of development can be discerned.

- 1. Initial phase:** The waste settles and is moistened up. Hydrolysis of hydrocarbons starts.
- 2. Oxygen and Nitrogen reducing phase:** The remaining oxygen is used up in oxidising reactions. Nitrate and ammonia act as oxidising agents and are reduced to elemental nitrogen (N_2). Redox potential E_h is falling rapidly. Duration: first two weeks after waste placement.
- 3. Acidic anaerobic phase:** Free oxygen is depleted (anaerobic conditions). Decomposition of hydrocarbons generates organic acids like acetate and propionate causing the pH to drop temporary to 4 or 5. Metals are mobilised. Duration: First two months after waste placement.
- 4. Methane generating anaerobic phase.** Further decomposition of hydrocarbons to CO_2 and methane (CH_4) as landfill gas. Carbonic acid (H_2CO_3) buffers the pH at 6 to 7. Continuous low redox potential. Temperature rises up to $120^\circ C$. The methane phase has an onset time of about 2 years and can last for 30 years.
- 5. Continuous leaching.** While the landfill pH is buffered by acid neutralising materials, the landfill chemistry is in a quasi-steady state. Constant but comparatively low concentrations in leachate lead to continuous emission of pollutants from the landfill. The duration of this phase depends on the acid neutralising capacity of the landfill material and can last thousands of years.
- 6. Possible pH decrease.** If the acid neutralising capacities of the acid buffer systems are used up by externally imported, or internally produced acids or buffer wash-out the pH of the landfill can fall sharply and formerly stable phases can be remobilised. Pollutant concentrations can rise significantly again.

⁶ However, in 2000 about 390'000 tons or 20 w-% of the waste landfilled in sanitary landfills was still burnable waste, but with a decreasing trend (BUWAL 2001e).

7. Humification and Reoxidation If the landfill has become unreactive, humification processes transform the landfill to a podzol (soil type). Intruding rain water oxidises the environment and redox potential E_h rises again. Precipitated metal sulfides can be oxidised, which lowers pH and increases metal mobility.

5.3 Inorganic landfills (slag, residual and inert material)

5.3.1 General description

'Inorganic landfills' is an ad-hoc term for landfills with mostly inorganic, low-carbon wastes, like incineration residues or inorganic building materials. Slag compartments, residual material landfills and inert material landfills can be regarded as inorganic landfills. In Switzerland, the total organic carbon concentration in the landfilled waste is legally regulated to be below 5 w-% in such landfills.

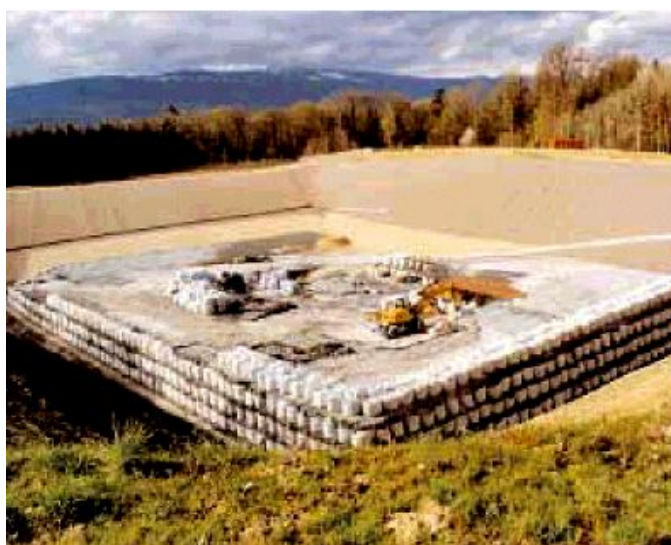


Fig. 5.2 The Swiss residual material landfill Oulens VS. The blocks of solidified incinerator filter ashes are visible

Compared to the intruding rain water, landfill bodies represent a high concentration deposit of several pollutants. The resulting concentration gradient between water and waste is a driving force for the continuous dissipation of pollutants into the leachate. The speed of dissipation depends on the dissolution and desorption characteristics of the involved materials and the geochemical stability of the mineral phases. Some incineration residues are thermodynamically unstable due to high temperature during incineration. This too, is a driving force of continuous change in the landfill. In time, landfill bodies undergo several mineralogical transformations. Over long timespans of hundreds or thousands of years, weathering processes can fundamentally change the mineralogical makeup of the landfilled material.

Following phases in an inorganic landfill can be discerned after waste placement:

1. Initial phase: The waste settles and is moistened up. Readily soluble salts like chlorides, sulfates, sodium and potassium are dissolved in the pore water and are mobile to be washed out with the leachate flow. The waste undergoes fast transformations especially in the humid environment of a landfill. Transformations consist of e.g. the corrosion and oxidation of metallic phases in fresh slags⁷,

⁷ Zeltner (1992) finds that over 80% of the metallic iron in landfilled MSWI slag is oxidised to several types of iron oxides within 5 to 10 years.

hydration of metal oxides like CaO or MgO, and carbonation of hydroxides with CO₂ from infiltration water to carbonates⁸. Also glassy phases are immediately starting to hydrolize (Frühwirth et al. 2000:149). All these reactions are exothermic and can lead to significant temperature increases in the landfill⁹; see Fig. 5.4. Even in 10 year old slag landfills temperatures of 80°C could be observed (EKESA 1992). Reactions between amorphous glass phases and alkaline leachate creates secondary mineral phases (CASH phases = calcium aluminium silica hydroxides). Dissolution of calcium hydroxide creates a high pH (Frühwirth et al. 2000:149). Over time the pH, that can initially be above 12, drops continuously. Below a pH of 10, microbial activity can decompose part of the remaining organic carbon traces (Lechner 2001). Until the pH of 8.3 of the next phase is reached, up to 50 years can go by (Kraxner et al. 2001:18).

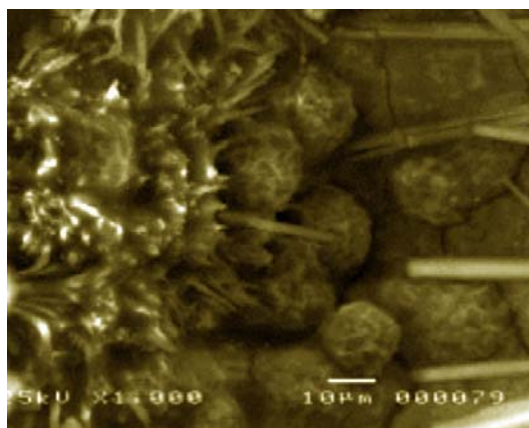


Fig. 5.3 Example of weathering products in MSW bottom ash. Calcium silicate hydrate CSH on the left and calcium aluminium silica hydroxides CASH on the right (Kraxner et al. 2001:72).

2. Carbonate buffer phase: The pH of the landfill pore water is buffered by the presence of carbonates to a value of around 8.3. As the pH is a master variable for landfill development, the landfill is in a quasi-steady-state during this phase. Depending on the acid neutralising capacity ANC, the calcium content, the quantities and pH of the intruding rain water, this phase will last for some 1000 to some 100'000 years. During that phase, natural weathering processes can transform minerals into thermodynamically more stable forms. A sequence of secondary mineral neoformations and transformations oxide→hydroxide→carbonate→silicate→... takes place (Sabbas et al. 2001). Carbonation of hydroxides, mentioned in phase 1, continues in this phase. The vitrified phases in slag can be transformed to more accessible clay phases by long-term weathering (Zevenbergen et al. 1995, Zevenbergen 1998). The liberated heavy metals from such phases might be bound again in secondary calcium aluminium silicates and -hydroxides (CASH phases) (Huber et al. 1996:5).

3. Possible pH decrease. If the acid neutralising capacity of the carbonate buffer system is depleted, the pH of the landfill can fall sharply below 7. Silicates, aluminium- and iron (hydr)oxides could buffer the system and affect the mobility of heavy metals (Sabbas et al. 2001). Formerly stable phases – e.g. CASH phases – can be remobilised. Pollutant concentrations can rise significantly again. A pH drop to a value of 4 must be considered as a worst case scenario (Huber et al. 1996:5, Johnson et al. 1995).

⁸ Similar to the hydration and hardening in cement: $\text{CaO} + \text{H}_2\text{O} = \text{Ca}(\text{OH})_2$; $+ \text{CO}_2 = \text{CaCO}_3 \cdot \text{H}_2\text{O}$.

⁹ The high temperature can damage landfill seals or lead to incrustations in leachate collection pipes by increased evaporation. These reactions can be enforced before landfilling, e.g. by slag quenching with water after incineration or by extended surface storage prior to landfilling. Even after water quenching, the temperature can reach 60°-70°C in surface stored slag heaps (Huber et al. 1996:4)

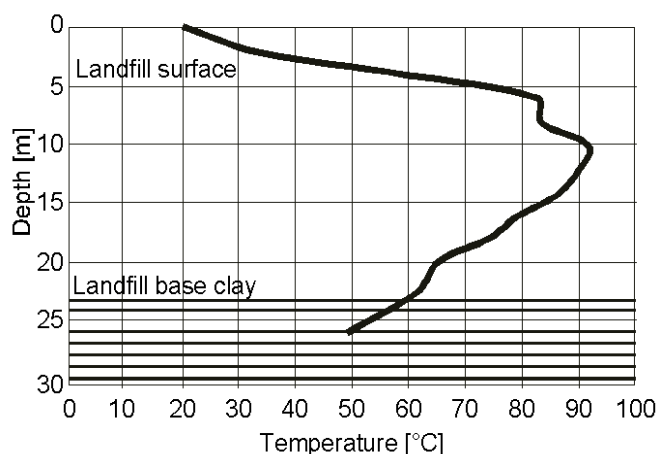


Fig. 5.4 Temperature measurement along a drilled hole in a slag landfill (Turk 1995)

The duration of carbonation ($\text{MeO} + \text{CO}_2 \rightarrow \text{MeCO}_3$) depends on the metal oxides present, the available carbon dioxide and the landfill depth. Frühwirth et al. (2000:182) calculates for a 15 m deep slag landfill a carbonation duration of 30 years, while the carbonate buffer phase lasts 5000 years.

Huber et al. (1996:91ff.) show that for MSWI slag the dominant loss of acid neutralising capacity occurs not through in-flow of protons in acid rain or acid producing internal reactions, but from the wash-out of buffering carbonate minerals.

Hellweg (2000:79) makes thermodynamic calculations of the reactions in a slag landfill body and finds the carbonate buffer is washed out after 56'000 years, using a atmospheric CO_2 partial pressure of $10^{-3.5}$ atm (316 ppm). In natural soil the CO_2 partial pressure is usually higher (10^{-2} atm, 10'000 ppm) due to decomposition in the detritus layer. Rain water intruding in landfills can be exposed to higher CO_2 partial pressures, when flowing through the vegetated landfill cover. Sensitivity calculations with a CO_2 partial pressure of 10^{-2} atm induced a threefold increase of carbonate washout and the carbonate buffer phase ends after 17'000 years. Also heavy metals like Cd, Zn, Pb, Cu were washed out 3 to 5 times faster during the carbonate buffer phase due to the lower pH.

5.3.2 Slag compartments

As explained above, slag compartments are not stand-alone sites but are separated parts of sanitary landfills. A separation that prohibits contact with other municipal waste is mandatory. Compartments are therefore not separate from sanitary landfills in a logistic sense, but they are separate in a physical and chemical sense. Landfill design requirements are the same as for sanitary landfills. Slag from MSWIs shall not have more than 3 w-% organic carbon (TOC). It is legally possible to recycle slag in road or dam constructions (TVA 2000, Art. 13). However, in the current Swiss practice all MSWI slag is landfilled (BUWAL 2001f).

5.3.3 Residual material landfills

Residual material landfills can receive wastes that are low in organic carbon (< 5 w-%) and are not biologically or chemically reactive in water. Less than 5 w-% of the pulverised waste shall be soluble when mixed with 10 parts water. Additionally, the waste must pass two extraction tests regarding

several parameters¹⁰. Residual materials landfilled are mainly filter ashes from municipal waste incineration and complying industrial wastes. The MSWI filter ashes are solidified with cement to comply with TVA extraction tests for residual material. A post-closure monitoring period of at least 10 years is mandatory.

5.3.4 Inert material landfills

Inert material landfills can receive unproblematic inorganic wastes. These wastes need to consist to at least 95 w-% of rock-like or mineral material like silicates, carbonates or aluminates. The waste composition must not exceed limits set for lead, cadmium, copper, nickel, mercury, and zinc. Less than 0.5 w-% of the pulverised waste shall be soluble when mixed with 10 parts water. The waste must pass two extraction tests regarding essentially the same parameters as for residual material landfills, but the limits are generally set 10 times lower. Inert wastes landfilled are clean or tolerable excavation material, road foundations, road tarmac (excluding old, tar-containing asphalt) and building wastes like concrete, tiles, bricks, glass, gypsum, asbestos-reinforced cement (BUWAL 2000). A post-closure monitoring period of at least 5 years is mandatory.

5.4 Underground deposits

Underground deposits are ordered waste storage spaces that are usually created in old potash or rock salt mines. No underground deposits exist in Switzerland, but some Swiss waste is exported to the German salt mines Herfa-Neurode and Heilbronn. Underground deposits are monitored and separated from the conventional salt mine part. Wastes are stored in steel drums, large steel containers or big bags.

¹⁰ The limit values to respect are set for aluminium, arsenic, barium, lead, cadmium, chromium III and VI, cobalt, copper, nickel, mercury, zinc, tin, ammonia, cyanides, fluorides, nitrites, sulfites, sulfide, phosphates, dissolvable organic carbon DOC, biochemical oxygen demand BOD₅, hydrocarbons, lipophilic non-volatile chlorinated hydrocarbons, chlorinated solvents and pH value.



Fig. 5.5 Separation of one completed compartment by a brick wall in the underground deposit Herfa-Neurode. At the ceiling the rock salt vein is visible¹¹

In Germany four underground deposits for non-radioactive wastes exist¹²; all in old potash or rock salt mines. In 1999 these deposits received 226'000 tons of hazardous wastes like incineration ashes from coal power plants and MSWIs, wastes from metal-producing and chemical industry, contaminated soils and dismantling wastes, electronic components like transformers and condensers (Borsch 2001). In Herfa-Neurode approximately 80'000 tons of hazardous waste are deposited annually in a depth of 700–800 meters and over an underground area of 20 km² (Brendel 2000).

Since 1999, a new deposit also exists in Wittelsheim (France-Alsace) operated by StocaMine with a licensed capacity of 320'000 tons of waste. A new salt mine deposit by Minosus Ltd. in Winsford, Cheshire, United Kingdom, seeks official licensing since 1999, but meets continued opposition by adjacent residents.

In Germany there are also 20 coal mines where the exploited veins are filled up with material to mechanically stabilise the shafts (German 'Berg(e)versatz', also called 'mine-valorisation'). The material used is excavation material but since 1991 also hazardous wastes like filter ashes is mixed in. No monitoring or ordered storage of the deposited materials is required by law. Since the material is used to stabilise the mine, this type of disposal can be declared as 'recycling' (Jansen 2000). This controversial practice is not under consideration here¹³.

¹¹ Picture from the operator of Herfa-Neurode 'Kali + Salz GmbH' <http://www.k-plus-s.com/medien/images/herfa-neurode.jpg>. More pictures of Herfa-Neurode can be seen at <http://www.ks-entsorgung.com> or in the preview section of <http://www.photoweb.de>.

¹² 1. Herfa-Neurode in Hessen (operating since 1972), 2. Heilbronn in Baden-Württemberg (since 1987), 3. Zielitz in Sachsen-Anhalt (since 1995), 4. Borth in Nordrhein-Westfalen (since 1997). A fifth deposit Riedel, near Hannover in Niedersachsen in the process of licensing (Borsch 2001).

¹³ Germany has accepted an ordinance that demands the same management standards for coal mine stabilisation with hazardous waste as for salt mine underground deposits. The ordinance is effective as of October 2002 (BMU 2002).

Only orderly disposal in underground salt mines – where some Swiss hazardous wastes are deposited – is inventoried here. Orderly disposal includes the record-keeping of waste samples, the full documentation of waste composition and location, which allows retrieval. Indeed, some wastes like disodium cyanamide, calcium formiate and arsenic trioxide have already been retrieved from Herfa-Neurode and used as secondary raw materials in the chemical industry (Brendel 2000). 7000 transformers containing PCB are stored separately, as it is expected that they can be recycled for their copper content in the future (WEKA 1993).

5.5 Emissions from landfills

Emissions to air and water from landfills can be separated into indirect emissions and direct emissions. Indirect emissions occur from the production of used materials and fuels. Direct emissions can be divided into emissions from landfill operations like transports, waste spreading and landfill shaping, and emissions that originate from the waste itself.

Emissions from the waste are usually the most important burden within the LCIA result of a landfill. Depending on the waste composition, waste reactivity and degradability the waste material is transferred to the landfill gas and the water outflow from the landfill (leachate). For reactive wastes in a sanitary landfill gas production can occur up to several decades. Leachate is produced continuously in landfills with permeable top-cover. In landfills with impermeable top-cover ('dry tombs') leachate is produced (again) after the top seal has been rendered unfit through ageing¹⁴. In any case leachate is usually produced continuously over an essentially unlimited timeframe.

Leachate can contain significant amounts of pollutants washed from the waste material in landfills. The temporal dynamics of these emissions are difficult to predict over such long timespans, but are governed such parameters as pH-value¹⁵ and redox potential (E_h) in the landfill body as well as the chemical composition, chemical speciation and mineralogy of the waste. Usually landfills develop a quasi-stable emission level after an initial settling phase. During that settling phase emissions can be high and need to be monitored.

The pH value development of a landfill depends on the acid neutralising capacity ANC of the landfilled waste, which is diminished by

- rain water inflow containing acidifying pollutants (NO_x , SO_2 , HF etc.) or carbonic acid (HCO_3^-) from atmospheric CO_2 ,
- the acid produced during the degrading or weathering of the waste and
- the washing out of buffering materials.

Once the acid neutralising capacity is used up, which can be after thousands of years after landfill closure, the pH drops sharply and the landfill turns acidic. This change in acidity leads to an increase in solubility of formerly stable phases containing heavy metals and consequently to large emissions in the leachate. This scenario is likely to occur sooner in sanitary landfills containing biologically

¹⁴ Landfills that receive a top seal after closure ('dry tombs') are initially closed-systems, but will become open systems after the failure of these technical barriers and penetration of rain water (Lee & Lee 1999). Technical barriers are bound to become faulty after decades or a few hundreds of years (Lee & Lee 1999, Covelli & Baumann 2000:6). So technical barriers only achieve a delay of emissions, in the case of dry tombs with a high risk of sudden and grave deterioration after centuries of eventlessness, when the dry environment is activated by water intrusion. Dry tombs are not common practice in Switzerland.

¹⁵ Lysimeter experiments with MSWI slags showed that pH is the most important influence on the leachate concentration of Pb, Cd, Zn, Ni, V, Co, Mn, Al, Ca, Mg and can cause changes of a factor of over 100, while sulfate, B, Si and easily soluble salts show no pH dependency. Other tested influences were liquid/solid-ratio, grain size, ageing, chemistry changes through drying (Kraxner et al. 2001:100).

reactive wastes, and later in the inorganic landfills (slag compartments, residual material landfills, and inert material landfills).

In a less dramatic scenario the acid neutralising capacity can remain positive and no dramatic increases in emissions occur over time. Nevertheless, normal weathering and gradual changes in the redox chemistry of the landfill (aerobic regime) can turn stable mineral phases into more soluble phases and subsequent wash-out (Huber et al. 1996:90). Results from leaching or availability test are used in environmental studies to gain information on the leaching behaviour of waste materials. Such tests typically do not last longer than hours or days. In such a short time it is not possible to simulate the weathering of the constituting minerals that occurs over long timespans (see e.g. section 'Heterogeneity in material' on page 33). Current leaching tests can therefore merely give an indication of the leachability of the *current* waste material or for a short term phase, but not of the *complete* long-term leachability of a material altered in time¹⁶. Also legislative leaching tests for landfill in Switzerland and the EU are performed on fresh, not weathered samples and are therefore are merely a test for the short-term behaviour of those wastes (Kraxner et al. 2001:17)¹⁷.

The continuous throughput of leachate water secures the open-system character of a landfill after closure. In that manner, possibly small concentrations of pollutants can be gradually washed out from the landfill during very long timespans, usually following an initial settling phase. In the following section, I shall discuss the problems and solutions of integrating such long-term effects and emissions in life cycle inventories.

5.5.1 Future emissions in LCA

Previous studies on landfills indicated that the pollutant potential remaining in a landfill after one hundred years after waste placement is significant (e.g. Zimmermann et al. 1996, Sundqvist et al. 1997, Hellweg 2000, Doka et al. 2002). Landfill experts generally agree that regarding long-term behaviour landfills cannot be regarded as stable systems, even if catastrophic events like landslides or floods are excluded (Leuenberger 1999, Sabbas et al. 1998, Huber et al. 1996, Lechner 2001). The designated barrier systems – 'inertisation', solidification, sealing sheets etc. – deteriorate in time and have a limited functional lifetime. There is a relevant and plausible potential that the remaining pollutant load in a landfill will be *completely* released, *if long enough timespans are considered*.

In other words, only a minor part of the harmful substances contained in waste have been released to the environment after one hundred years. From a LCA point of view, landfills postpone emissions from today's wastes into the future. This remaining future pollution potential must not be ignored, since life cycle assessment is concerned with creating a complete synopsis of environmental burdens attributable to a process, *wherever* or *whenever* they occur.

Inclusion of long time horizons in life cycle inventories and in LCA is problematic and needs special understanding. First, a choice of the time frame of the assessment has to be made. Second, the future emissions must be predicted using a landfill model, because no measurements can be made in the far future. The choice of the temporal system boundary is a relevant key question, because it influences to

¹⁶ Such tests were used to estimate long-term emissions in former waste LCI studies (Zimmermann et al. 1996, Hellweg 2000). This method is not continued in this study. An alternative idea was to use sequential extraction data which allows to discern different phases in waste materials, e.g. easily dissolvable, carbonate phases., metals bound to amorphous FeO_x/MnO_x, metals bound to crystalline FeO_x, organic fraction and metal sulfides, metals in crystal lattices/silicates (see Huber et al. 1996, Frühwirth et al. 2000). This idea was abandoned for the same reason that sequential extraction is unable to simulate future material changes due to weathering, which turns all mineral phases into leachable phases (Personal communication with A.C. Johnson, EAWAG Dübendorf, September 12, 2002).

¹⁷ Kraxner et al (2001) validate several leaching methods for waste materials, which also include methods with *enforced artificial weathering*.

a large extent the type of landfill model. In the next section I will look at the problem of temporal system boundaries.

The default procedure for future emissions in LCA

Within the framework of LCA it has been common practice to neglect space or time issues. For example, ISO 14042 states in chapter 8: "*LCIA typically excludes spatial, temporal, threshold and dose-response information, and combines emissions or activities over space and/or time.*" (ISO 2000). This implies that the mentioned types of information are also disregarded in the LCI stage of LCA, i.e. emissions are inventoried *regardless of their time of release*. All emissions are treated as if they would be released at the reference point of time¹⁸ and within one reference geographic region¹⁹. LCIA practice currently lacks the tools to model damages to *future* environments from future emissions. Hence, as the default procedure in LCA, future emissions are included in the assessment without any kind of weighting and are treated just like short-term emissions²⁰.

Alternative procedures: temporal discounting of future emissions

Besides this default procedure of treating future emissions just like present emissions, alternative procedures of dealing with future emissions are imaginable. It is possible to differentiate and weigh future emissions differently in the assessment than present emissions. Weighting future emissions *lower* than present emissions is called positive temporal discounting. Weighting future emissions *higher* than present emissions is called negative temporal discounting. For both types of discounting and the default procedure (no discounting) several arguments can be put forward. These arguments are discussed at large in (Hellweg 2000, Hellweg et al. 2002, Doka et al. 2002). Some of the most memorable arguments for either procedure are given in the following paragraphs.

Some arguments for positive temporal discounting: In economics, discounting future monetary benefits is done, e.g., because an individual might have a pure time preference. He/she rather has the profits now, than later, possibly by mere impatience. Also economists usually presume a growing economy with continuously increasing productivity of capital. This makes current benefits more valuable than the same nominal amount of benefits in the future. Another reason for positive temporal discounting can be that the further we go into the future the more the individual might be unable to take advantage of these profits because there is an increasing probability that the individual has died²¹ (Hofstetter 1998). By symmetry, future *monetary costs* can also be subject to discounting. For example, costs that arise in the future are less serious than the same costs today. By this reasoning it can be argued, that future *environmental burdens* are less bothersome than present burdens.

¹⁸ The reference time point is usually the present with the current environmental state.

¹⁹ Currently, the reference geographic region depends on the LCIA valuation method and can be e.g., Switzerland, Netherlands, or Western Europe.

²⁰ However, also encountered are hard *temporal cut-offs*, i.e. all emissions after a certain time span, e.g., 150 years, are completely neglected, which is a form of temporal discounting. But, within the considered period usually no temporal discounting is performed.

²¹ This argument can also be stated in the forms that every generation has its own problems, and our generation should only be bothered with its own problems.

Some arguments for negative temporal discounting: One could argue that future generations might be compensated financially for the environmental impact. This compensation should be high enough to satisfy those damaged. The necessary money could be invested on the capital market. However, considering that environmental goods and resources might become very scarce while monetary wealth might increase in the future, this compensation cost could be very high, even approaching infinity. In other words, future generations who, according to the economical framework, will probably not lack money, might demand a very high compensation for an additional risk to their health.

The annual pollutant loads in a future environment are bound to be higher than currently. This leads to higher burdens per kilogram pollutant than today, i.e. increased LCIA classification factors for future emissions (Hellweg 2000:141ff.).

Another reason for positive discounting can be that future releases often represent an uncertain and/or poorly manageable risk. A *risk avoiding* personality (e.g. the Egalitarian archetype) will find a future and uncertain release less desirable than a present and certain release, and therefore weigh present emissions higher than future emissions.

Some arguments for no temporal discounting: Within the framework of sustainability, we should not consider damages to future generations to be less important than damages to the present generation. Doing so would undermine the notion of sustainability. Also applicable is the 'polluter pays principle' as decreed by Swiss environmental law (BUWAL 1986): We have the use of the utility, we cause the damage, we should be held responsible for the *full* extent of the caused pollution. Obviously, this notion is also a core concept of life cycle assessment.

Discussion of temporal discounting

Whether discounting of long-term emissions is appropriate, depends on the goal of an LCA as well as the subjective preferences expressed in the impact assessment LCIA. In general, LCA is a tool to develop less environmentally burdening products and services or optimise ecological efficiency. The motivation for that is the insight of producers or consumers that present consumption levels in developed countries are unsustainable: Consumption at present level with current impacts could not be sustained for an indefinite time. The notion of sustainability includes the concept that the present generation should not fulfil its needs while jeopardising the means for the future generation to fulfil their needs. LCA is a tool for the environmental part of this sustainability discussion by pointing out less burdening options. **The mere shifting of burdens into the future shall not be regarded as a sustainable solution of environmental problems²².** All of the possible arguments put forward to argue in favour of positive temporal discounting are in obvious disregard of this understanding of LCA. A pure time preference ('impatience') leading to positive temporal discounting implies that future people do not have equal rights as present people (Finnveden 1997). Furthermore, it *"is in the egoistic interests of present persons, those responsible for creating the waste, not to bear the consequences of their actions and instead to force those consequences on others who do not deserve them. Such an egoistic position is not ethically defensible"* (Shrader-Frechette 2000). LCA aims at displaying potential damages to *create a motivation* to reduce the risk of those damages actually happening. So, LCA needs to *show* those damage potentials and not conceal them by anticipating that

²² Let's consider two alternative product options: Product A produces 90 burden points in the short-term, and an additional 10 burden points in the long-term future. Product B produces 50 burden points in the short-term, but an additional 450 burden points in the long-term. If we adopt a *short-term perspective*, product B with 50 points seems preferable over A with 90 points. Product B successfully transfers the major part of its attributable burden into the future. In a long-term perspective there is no reason to discount future emissions, if these emissions are really attributable to a product option, and product B with 500 points is found to be more burdening than product A with 100 points. For waste processes the release of waste components is clearly attributable to the waste product and there is no reason in LCA to discount those emissions from a disposal process. A short-term perspective and discounting of long-term emissions favours processes and products which shift burdens and imposes responsibilities onto future generations. This cannot be regarded as a sustainable conduct.

others will have to deal with the problems. Otherwise, incentives to decide in favour of less burdening technologies/products would disappear (Steen 1999)²³.

It can be concluded that the LCA of landfill processes should generally also include long-term emissions, since the exclusion of those emissions is usually not defensible within the ecological sustainability discussion and the overall understanding of LCA as a sustainability tool. Due to the possibly controversial perception of long-term emissions it is a good idea to inventory long-term emissions separately. This allows for sensitivity analysis of LCA results with and without long-term emissions. In early 2009 theecoinvent management has introduced changes in the suggested LCIA assignments of the ecoinvent database v2.1. For Eco-indicator'99 and EDIP the long-term groundwater emissions are heeded only and exclusively in newly created, separate LCIA categories (e.g. "stored ecotoxicity" and similar). In the ecological scarcity 97 and 06 methods, *all* groundwater emission CFs (short- and long-term) are removed altogether. Despite the unaltered name, the category "Eco-indicator'99 HA total" does not heed *all* emissions in v2.1, but excludes long-term groundwater emissions. Ecoinvent management refers to the discussion in (Frischknecht et al. 2003b, chapter 2.1.3 "Assessment for long-term emissions") for user guidance. There the conclusion was, in accordance to the discussion above, to heed long-term emissions with the same factors as for short-term emissions. Accordingly, the "stored..." categories need to be added to LCIA results.

Timeframes in this study

Within the framework of this inventory study, long-term emissions are modelled and included in the inventory of landfilled wastes. To have a rudimentary distinction of short-term versus long-term emissions, emissions that occur *after 100 years* after deposition are inventoried in a special emission subcategory ('*water emissions, long-term*'), while emissions that occur before 100 years are inventoried in the same subcategory as present emissions.

Emissions from landfills are split up into emissions that occur in the first 100 years after waste deposition and emissions that occur afterwards.

The point of 100 years is an arbitrary choice. It includes the complete or larger part of the surveyed periods²⁴ of landfills. It coincides with the choice made in studies of the Department of Systems Ecology at Stockholm University (e.g. Finnveden 1998, Sundqvist et al. 1997). The *uniform* separating time mark of 100 years is a deviation of the choice made in former LCI studies of ETH Zürich (Zimmermann et al. 1996, Hellweg 2000, Doka et al. 2002) where the *variable* surveyed periods of the different landfills were used to separate short-term emissions from long-term emissions (i.e. 40, 75 and 150 years, respectively. See footnote 24).

²³ It can be imagined that a human society in the far future will have the means to clean up the environment at virtually no cost because of large advances in technology, e.g. with nano-robots. Today's long-term burdens are then of no or little consequence, because they can be easily avoided by intervention and remediation. This can be regarded as science-fiction, because such technology does not exist today and the capabilities of future societies cannot be guaranteed. On the other hand long-term leaching is the intended, planned-for and normal development of landfills, which is not doubted by any landfill expert, and was observed e.g. in 2000 year old roman landfills (Lee & Lee 2000). There is little justification in disregarding a highly likely process on the grounds of a pure speculation.

²⁴ Surveyed periods are the lengths of time a landfill's emissions are actively monitored by the local government. Surveyed periods are *at least* 10 or 15 years (TVA 2000) and are estimated to be 40, 75 and 150 years for residual material landfills, slag compartments and sanitary landfills, respectively (Zimmermann et al. 1996:B.13).

Cut-off at background concentrations?

It could be argued that pollutant concentrations in leachate become very small over time and have little ecological relevance. Following this argument, leachate emissions which, for example, fall below some natural background concentration value should not be inventoried, since they are assumed to have no ecological relevance. Following objections to this view can be put forward.

- The notion that concentrations observable in nature are benign and can be used as a cut-off criteria of relevance should be challenged as a whole. If, in a hypothetical situation, humans would pollute all rain water only *exactly up to* the background concentration, the pollutant load in water would double, since the anthropogenic pollutants would *add* to the actual natural background. It is not guaranteed that this doubling of load would still be benign, but the understanding presented above would perceive no harm and no signal in an according LCA would be generated.
- In LCA, generally no concentration or dose-response information is heeded, i.e. flows are inventoried in the form "0.3 kg Cd per functional unit" and not as "0.3 kg Cd (in a concentration of 0.5 mg/l) per functional unit" (ISO 2000, cf. also section 'The default procedure for future emissions in LCA' on page 15). This makes sense in LCA, as the *sum total* of *all* the contributions of a pollutant flux (within a region and during a certain time) determines ecosystem and human health states²⁵. This annual total of emissions determines the ecological impact, *irrespective if this total is made up from a few large or many infinitesimally small sources*²⁶. Critical loads can be reached by many diffuse, low-concentration sources, too. In LCIA the total contributions to a burdened region are valued (actual load or working point). Hence, there is no justification to exclude emissions just because they are small.
- The notion that small enough concentrations are not damaging (Paracelsus principle) is factually true on an individual organism level. But including it in an LCA would signify that diluting emissions with a large enough amount of media (air, water, soil) would solve environmental problems. This would send the entirely wrong – even illegal – message to decision makers. As mentioned above, LCA has a larger ecosystem view and not only an individual organism or local view.
- It has been pointed out by (Sundqvist 1998) that it may be irrelevant to use today's acceptable levels as a cut-off, since the environment may be more or less sensitive in the future due to a changing background contamination.
- Even if the argument would be accepted, and concentrations below an acceptable or natural background level would be ignored in the inventory, little would change in the outcome. An exponential decline model could be used to predict the concentration development of the landfill leachate (see section 'Short-term emissions' on page 21). The levels in unpolluted surface waters can be used as the natural background concentrations²⁷. The time t when a certain leachate concentration c_x is reached, can be calculated according to Eq. 5.1 below (definition of variables see section 'Short-term emissions' on page 21). Time spans until geogenic background concentrations are reached in slag compartment or residual material landfill leachate, are in the

²⁵ This is the consequence of the **I=PAT** formula by (Holdren & Ehrlich 1974), i.e. 'Impact on the environment' [pollutant/year in a region] equals the multiplication of 'Population' [persons/region] times 'Affluence' [consumed products/person*year] times 'Technology' [pollutants/product].

²⁶ On a *local* scale and for assessment of *acute* damages, high density of large polluters can be relevant. Current LCIA methods presently look at chronic effects occurring at current background levels (dose-response working point) and are calibrated on national or continental levels. Also LCA is not a precision tool to accurately model local effects, but a coarse tool to create a *synoptic life-cycle view* establishing the *relative importance* of many different environmental effects or effect potentials.

²⁷ 52 mg Ca/l, 14.75 mg S/l, 7.2 mg Cl/l, 1.8 mg C/l, 0.03 mg Zn/l, 0.01 mg Cu/l, 0.0016 mg Cr/l, 0.0015 mg Pb/l, 0.0001 mg Hg (unreferenced geogenic background concentrations in Zimmermann et al. B.17).

order of thousands of years for highly soluble elements like C, S, N, but hundred thousands to *tens of millions* of years for heavy metals like Zn, Pb, Cd, Cu. If a timeframe were chosen, that would end, when *all* elements have reached their natural background concentration, *more than 94%* of the theoretically mobilisable fraction of an element would have been washed out already (smallest share is for Pb). In other words, heeding the 'natural background cut-off' argument – and ignoring its incompatibility with LCA methodology – would decrease the long-term emission potential *at best* by only 6%. With uncertainty factors in the heavy metal content of landfilled waste which can easily be above 300% (GSD²), these 6% would influence the overall result only negligibly.

$$t = \ln\left(\frac{c_x}{c_o}\right) / -k$$

Eq. 5.1

$$-k = \ln\left(1 - \frac{STTK}{TK_\infty}\right) / 100a = (V \cdot c_o / m \cdot TK_\infty)$$

For all these reasons (mainly B) a limited timeframe with a 'natural background cut-off' is not used in this study. The long-term phase is in principle an open-end timeline.

5.5.2 Modelling of landfill emissions

Introduction

As concluded above the inventory of landfill processes includes emissions occurring during a potentially unlimited time span after the deposition of the waste. As pointed out before, these future emissions must be modelled based on assumptions, since no measurements are available for the far future emissions of landfills.

Similar to the direct emissions of MSWI (see part II of this report), the emissions from landfilled waste are understood as *waste-specific* emissions, i.e. the emissions are derived from the chemical composition of a specific waste and transfer coefficients for chemical elements. These transfer coefficients determine for each landfill which percentage of a certain chemical element is released to the environment. Landfill transfer coefficients essentially make up the landfill model used here. The need to inventorise *waste-specific* emissions is a major difference to other existing landfill models, that usually model the *average* behaviour of the landfill, i.e. the emissions of the average mixture of landfilled waste.

$$\text{Eq. 5.2} \quad \text{Emission}_{\text{media,phase},i} [\text{kg } i / \text{kg waste}] = TK_{\text{media,phase},i} [-] \cdot \text{waste composition}_i [\text{kg } i / \text{kg waste}]$$

In the landfill model, emissions of a certain chemical element *i* (e.g. zinc, copper, cadmium) to a certain environmental media (usually pore water, but also to air for sanitary landfills) during a certain landfill phase (here only the short-term phase < 100 years and long-term phase > 100 years are distinguished) are determined. The emissions are calculated from the transfer coefficient for that element *i*, for the appropriate media (water or gas) and landfill phase (short-term or long-term²⁸). The transfer coefficient is multiplied by the content of the element *i* in the assessed waste. In that way the various transfer coefficients represent the characteristics of the assessed landfill and the waste composition guarantees that the result is waste-specific. It is currently not possible to have waste-

²⁸ By convention, the term 'long-term transfer coefficients' refers to the *total* release of elements during the short-term *and* the long-term period (total availability or total long-term emissions). The term 'short-term transfer coefficients' refers only to the emissions during 0–100a (short-term emissions). The emissions >100a need to be calculated by difference (total minus short-term).

specific transfer coefficients, except for sanitary landfills the short-term and long-term transfer coefficients are adapted to the degradability of the assessed waste.

To summarise, the general method in this report to create waste-specific landfill models is to first calculate *average* transfer coefficients from *average* operation and *average* waste compositions. In the case of inorganic landfills these average transfer coefficients are multiplied directly with the specific waste composition under consideration to calculate waste-specific emissions, cf. Fig. 5.6 top half. In the case of sanitary landfills, *waste-specific* (not average) transfer coefficients are calculated from *average* landfill data and waste-specific degradability parameter D , cf. Fig. 5.6 bottom half. All calculations apply to several chemical elements, represented as matrix vectors in Fig. 5.6.

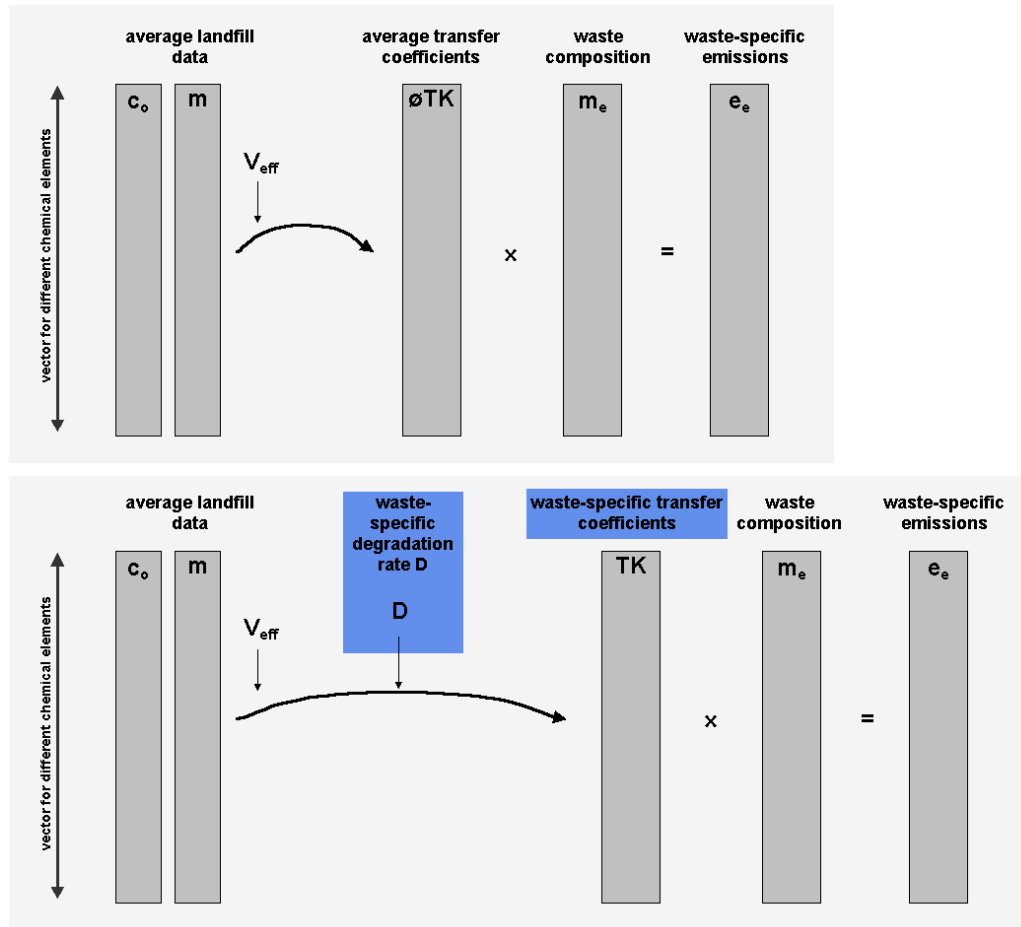


Fig. 5.6 General calculation scheme for landfill emission models in this report, for inorganic landfills (top half) and sanitary landfills (bottom half)

Lifetime of barriers and groundwater threat

The landfill is designed to collect leachate and discharge it to a sewer or to a surface water body. It can be assumed that for the short-term leachate <100 a these systems remain intact – at least for the majority of the time. But base seals and collection systems of landfills have a limited lifetime. This includes the geological barrier (clay layer). Considering long timespans, base seals will leak, collection systems will fail and leachate will enter the ground below the landfill. Baccini et al. (1992) consider a lifetime of 50 years for base seals. Frühwirth et al. (2000:128) discuss the special

circumstances and ageing processes of geological barriers in a landfill and consider a lifetime of a clay barrier²⁹ of 100 years to be realistic.

Do emissions into the ground³⁰ pose a risk? A threat to human health exists, if the pollutants reach the groundwater. But even if the pollutants remain in the ground below the landfill site, an ecosystem damage risk of plant take-up, e.g. by trees, remains. Stefanie Hellweg (2000:98ff.) calculates the retention time of pollution fronts until they reach the groundwater table heeding several soil parameters like infiltration rate, preferential flow paths, cation exchange capacity, acidity, clay content, iron oxide content, organic content and distance to groundwater table. About 20% to 40% of the metals reach the groundwater directly through flow in continuous macropores. Macropore flow occurs relatively quickly, it interacts hardly with the soil matrix and the transfer to groundwater levels takes only approximately 3 to 25 years. The remaining 60% to 80% of the leached metals are slowed through sorption and reach the groundwater table later (so called 'matrix flow'). Retention times for metals transported by matrix flow are around 400 to 3000 years for Cd^{2+} , Ni^{2+} , Zn^{2+} , Cu^{2+} , Cr^{3+} , and 5000 to 33'000 years for Pb^{2+} and Hg^{2+} . These values are calculated with default values for Switzerland proposed by (Hellweg 2000:107). For macropore and matrix flow combined, average retention times of 1000 years result (12'000 years for Pb^{2+} and Hg^{2+}). Over the time span considered for the landfill models (60'000a) the majority of the emitted pollutant load can be expected to reach the groundwater.

Erosion and climate change

In the long timespans considered, erosion is a probable influence on landfill development. For Swiss landscape and climate erosion rates of 5 to 100 cm in 1000 years are realistic (Baccini et al. 1992). However, the geotextiles usually applied on the landfill surface during closure have high resistance against erosion. The decomposed or remaining waste can have very variable erosion characteristics. With the erosion rates cited above, a landfill of 15 m thickness plus 4 m recultivation layer would be eroded within 19'000 to 380'000 years. In such a long timespan the climate and accordingly erosion strength can change.

Catastrophic events like flooding or landslides can short-cut erosion in the long run, although Swiss landfills must be built outside (today's) flooding zones. Changing climate can increase precipitation³¹, water and wind erosion regimes. Cooler periods can promote glacier formation, which increases the likelihood of landscape remodelling and distribution of landfill contents. On the other hand, new permafrost soils can postpone landfill leaching.

These effects sustain the notion of a *worst case* that represents a complete mobilisation of all pollutants from the landfill. This worst case will be heeded in the uncertainty data of the long-term transfer coefficients.

Short-term emissions

For the time after the placement of the waste, information is available in the form of leachate and landfill gas measurements of present landfills and closed landfills. Also the gas production during the methane phase of a sanitary landfill is well understood and can be used to calculate emissions to air.

²⁹ Clay layer of 50 cm thickness and water permeability (percolation coefficient) k_f of 1^{-10} to 1^{-11} m/s.

³⁰ 'Ground' signifies a horizon *15 to 25 meters deep* below the landfill, and not a (vegetated) *surface* soil, that is sometimes considered in LCA as emission media, e.g. for agricultural emissions.

³¹ Like predicted from global warming model calculations.

The exponential model

Short-term emissions transfer coefficients can be derived from measurements of the leachate in existing landfills, by assuming that the measured trends will continue into the future. In 1989 Hasan Belevi and Peter Baccini introduced a popular formula to predict long-term emissions from sanitary landfills after the initial active phase (Belevi & Baccini 1989). The behaviour of sanitary landfills was assessed by extrapolating the observations made in existing landfills. The time after the initial active phase (methane phase) is observed. The landfill body is assumed to be a simplified chemical reactor without concentrations gradients. The leachate concentrations can then be predicted based on the following parameters³².

- c_o Mean concentration of the observed element in the leachate during the *first year* after the active landfill phase [g/l].
- V Mean annual leachate output from the landfill per kg waste. Assumed to be constant [l/a per kg waste].
- m Total amount of the observed element in the landfill at the end of the initial active phase [g/kg waste].
- TK_∞ Fraction of the observed element that can be mobilised at the end of the initial active phase [kg/kg].

All these parameters are constant or assumed to be constant after the initial active phase. From these constants the time-dependent concentration of the observed element in the leachate $c(t)$ is calculated according to the following formula:

$$\text{Eq. 5.3} \quad c(t) = c_o \cdot e^{-(V \cdot c_o / m \cdot TK_\infty) t}$$

where t is the time variable in years.

This model features an exponential decline of concentrations from the starting level c_o . An example from the original publication (Belevi & Baccini 1989) is shown in Fig. 5.7. Similar exponential decline formulas have been formulated e.g. by Rowe (1991) or Kruse (1994).

³² The parameters are adapted here to relate to 1 kg of waste and to the variables used in this report.

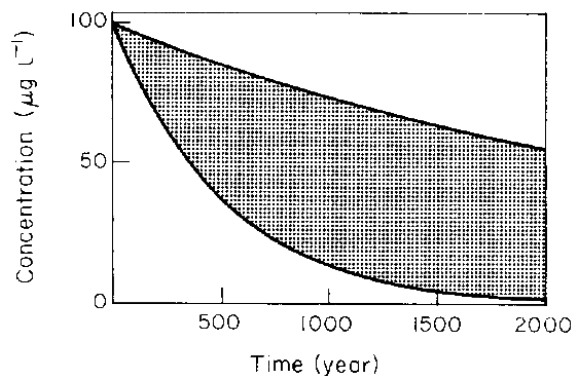


Fig. 5.7 Concentrations dynamics in leachate according to simplified exponential decline. The range is defined by variance in the content in the waste m . Original chart from (Belevi & Baccini 1989)

The term $m \cdot TK_{\infty}$ in the exponent of Eq. 5.3 signifies the total mobilisable amount of the element in the landfill per kg average waste at the end of the initial active phase³³. This amount is the total initial emission potential and is represented by P_o .

$$\text{Eq. 5.4} \quad P_o = m \cdot TK_{\infty} \quad \text{Total initial emission potential per kg average waste}$$

If the fraction c_o/P_o is large, then a relatively large amount of the total initial emission potential has been washed out in the first year, and the element can be considered to be washed out fast. The concentration $c(t)$ shows a rapid decline for 'fast' elements in the sense above and for large volumes of leachate water flow V . Belevi and Baccini determined the mobilisable fraction of an element by performing a leaching experiment on pulverised waste samples with four consecutive leaching steps with distilled water for 0.5, 5, 50, and 288 hours respectively ($L/S = 10$). After that time all mobilisable matter was assumed to be extracted, which is an underestimation, "since the experiment does not guarantee that further extractions give negligible contributions", as the authors note themselves (Belevi & Baccini 1989).

Mobilised and remaining mass in the exponential model

Since the model follows a simple first-order decay dynamic, the *current* leachate concentration $c(t)$ is always proportional to the *remaining* mass of mobilisable matter. This remaining mass of mobilisable matter can be regarded as the remaining emission potential $P(t)$, i.e. the amount of potentially mobilisable pollutant that has not yet been emitted. Similarly, the initial leachate concentration c_o (i.e. $t=0$) is equally proportional to the total initial emission potential P_o . From this proportionality follows:

$$\text{Eq. 5.5} \quad \frac{c_o}{P_o} = \frac{c(t)}{P(t)} \quad \text{and from that} \quad P(t) = P_o \cdot \frac{c(t)}{c_o}$$

The removed and emitted amount of mobilisable matter $E(t)$ can be calculated as difference of P_o and $P(t)$:

³³ In the original form, Belevi and Bacini (1989a) formulated the equation for $c(t)$ for the whole waste mass of the landfill. Here the formula is normalised to refer to 1 kg of average waste. In the original form, the total initial emission potential $m \cdot TK_{\infty}$ is represented by the total mass of the landfilled waste M times the initial concentration of the element in the waste that can be mobilised m_o , which is the equivalent expression, but for the whole waste mass of the landfill and not just 1 kilogram.

$$\text{Eq. 5.6} \quad E(t) = P_o - P(t) = P_o \cdot \left(1 - \frac{c(t)}{c_o}\right) = m \cdot TK_\infty \cdot \left(1 - e^{-(V \cdot c_o / m \cdot TK_\infty) t}\right)$$

The fraction of mobilised element $E(t)$ in relation to the total initial amount m at any time t is equal to the transfer coefficient for that time t :

$$\text{Eq. 5.7} \quad TK(t) = \frac{E(t)}{m} = TK_\infty \cdot \left(1 - e^{-(V \cdot c_o / m \cdot TK_\infty) t}\right) \quad \text{Exponential transfer coefficient}$$

With this formula the transfer coefficient for any point in time can be calculated. The transfer coefficient $TK(t)$ approaches TK_∞ for large times t . The specific annual leachate volume V will later be replaced by the effective leachate volume V_{eff} , heeding preferential flow in leachate (see section 'Heterogeneity in landfill hydrology' on page 30).

Exponential versus linear approach

It is interesting to note the relation of Eq. 5.7 for exponential decline with a linear approach, i.e. assuming that the observed leachate concentrations are *constant* over time and are always equal to c_o . The transfer coefficient $TK_{lin}(t)$ in such a linear model would be calculated from Eq. 5.8.

$$\text{Eq. 5.8} \quad TK_{lin}(t) = \frac{V \cdot c_o \cdot t}{m} \quad , \text{with } TK_{lin}(t) \leq 100\% \quad \text{Linear transfer coefficient}$$

In general, a term $(1 - e^{-x})$ equals approximately x , if x is much smaller than unity. This approximation can be applied to Eq. 5.7, with x being the term $V \cdot c_o \cdot t / (m \cdot TK_\infty)$ in the exponent. If this term is much smaller than unity, it means that the element under consideration is not easily leached out (c_o is small compared to $m \cdot TK_\infty$) and that after the time t still a majority of the leachable mass remains in the landfill. Hence, if $V \cdot c_o \cdot t / (m \cdot TK_\infty)$ is much smaller than unity, Eq. 5.7 can be approximated with $TK_{exp}(t) = TK_\infty \cdot V \cdot c_o \cdot t / (m \cdot TK_\infty)$, which equals $V \cdot c_o \cdot t / m$, i.e. the equation for the linear model (Eq. 5.8). It can be concluded that for elements which are washed out over time spans considerably longer than 100 years, there is *no perceptible difference between a linear and an exponential approach* to determine the *short-term* TK. Elements which do not fulfil this condition are the easily soluble ones, like sodium, potassium or chlorine.

Supplementary formulas

If TK_∞ and $TK(T)$ for a specific time T are known, any $TK(t)$ can be calculated:

$$\text{Eq. 5.9} \quad \begin{aligned} TK(t) &= TK_\infty \cdot (1 - e^{-kt}) \\ -k &= \ln\left(1 - \frac{TK(T)}{TK_\infty}\right) / T \end{aligned}$$

The time t until a certain leachate concentration c_x is reached is calculated with following formulas.

$$\text{Eq. 5.10} \quad \begin{aligned} t &= \ln\left(\frac{c_x}{c_o}\right) / -k \\ -k &= \ln\left(1 - \frac{TK(T)}{TK_\infty}\right) / T \end{aligned}$$

Application of the model for short-term behaviour

Although the Belevi-Baccini-model (Eq. 5.3) was derived based on observations from sanitary landfills after the reactive methane phase, it can be applied to any landfill, assuming that it has reached a quasi-steady-state and leachate concentrations are ruled by first-order kinetics. Although inorganic landfills have initial settling reactions, this formula will be used *for the residual landfill and the slag*

compartment to calculate transfer coefficients of soluble elements. For the sanitary landfill another approach will be used that considers the different waste-specific degradation rates to derive waste-specific transfer coefficients (see chapter 6.1 'Waste-specific emissions from sanitary landfills' on page 43). For the inert material landfill no direct leachate emissions will be used, cf. chapter 9.1 'Emissions from inert material landfills' on page 74.

For the residual landfill and the slag compartment the following procedure to calculate average short-term transfer coefficients is used:

For easy soluble elements an exponential decline of the leachate concentration is assumed. Easy soluble elements are the monovalent ions (Na^+ , K^+ , Cl^- , F^- , Br^- , I^-), oxianion-forming elements (WO_4^{2-} , HVO_4^{2-} , $\text{Cr}^{(\text{VI})}\text{O}_4^{2-}$, MoO_4^{2-} , HBO_3^{2-} , HSeO_4^- , SbO_4^{3-} , HASO_4^{2-}) and also nitrogen (as nitrate NO_3^-).

For all other elements (mostly cationic metals) a constant leachate concentration is assumed. This corresponds well with results from thermodynamic calculations of the leaching behaviour in long-term landfills models (Hellweg 2000). The concentrations of these elements in the leachate are governed by the conditions and processes in the landfill. If the conditions do not change, the concentrations will remain constant. Heterogeneity in landfill hydrology and waste distribution can lead to variations in leachate, but the average leachate concentration is established during the flow of the leachate through the landfill, with the leachate having average residence times of years.

The transfer coefficients are calculated from Eq. 5.7 and Eq. 5.8 on page 24. The necessary information for c_o and m is derived from literature values (see Tab. A.2, Tab. A.3, and Tab. A.4 on page 112ff. in the appendix). Data from various landfill sites is collected, implying that a collection of values from various sites gives mean values representative of a average generic and typical landfill site. The reason for this is that no complete data set for all necessary elements from one single landfill site is available.

A total mobilisable fraction TK_∞ of 100% is used in Eq. 5.7, with the exception of chromium. In slag, 75% of chromium is present as chromite (Cr^{III} in FeCr_2O_4), which is very stable and not available even under most aggressive conditions (Huber et al. 1996:47). For the TK_∞ of chromium a value of 25% is used. In Eq. 5.8 the values are corrected to $\leq 100\%$. For most elements a complete mobilisation is assumed. In reality, this mobilisation potential is initially not necessarily 100%, but will be achieved over time through weathering. The exact weathering dynamics are not modelled here, so in the model the TK_∞ is constant over time³⁴. Where no information on c_o and m is available, transfer coefficients are estimated from other sources, or as proxies from other elements based on chemical similarities.

Long-term emissions

This section relates only to the residual material landfill and slag compartments, i.e. inorganic landfills. The calculation of sanitary landfills (organic landfills) are specified in chapter 6.1 'Waste-specific emissions from sanitary landfills' on page 43.

As mentioned in footnote 28 on page 19 the long-term transfer coefficient refers to the total fraction of an element in waste, that can be mobilised and emitted in the long-term perspective. The master variables that ultimately control the pollutant potential are the pH of the pore water (leachate) and the mineralogy of the landfilled material. The pH determines which chemical species are mobile. The pH

³⁴ Exemplary, numerical simulations with a mobilisation potential which *changes in time* were performed. For demonstration purposes it was assumed that the mobilisable fraction is in the beginning only 20% of the total amount (without any measured evidence for that figure) and increases over time with a square-root dynamic, i.e. proportional to $\sqrt{\text{time}}$. This dynamic is likely to occur for progressive weathering. Compared to the used model with *constant and complete mobilisation potential throughout* ($TK_{\text{inf.}} = 100\%$) the resulting TK were 0% to 20% smaller, depending on point in time (i.e. between identical and 80% of the values from the used model). This incompleteness of the used model must be accepted due to lack of field data and indeed seems acceptable, compared to the large uncertainties already observed in landfill measurements of current composition and content, which form the basis of TK calculations in this study.

and redox potential are usually closely related (Huber et al. 1996:52). Mineralogy of the waste determines if phases are stable under the encountered conditions. Both variables are intertwined and can change in time. Ongoing geochemical weathering processes change the mineralogy of the waste. The development of the pH value depends on the acid neutralising capacity of the landfilled material, which in turn is defined by the minerals present.

Meaning of uncertainty ranges for long-term transfer coefficients

The prediction of landfill development over very long time spans is vague at best and depends on many uncertain parameters like development of geochemical weathering, climate conditions, vegetation. Accordingly the transfer coefficients which describe the landfill development can cover a large range representing the differences between a best case and a worst case.

The goal of this study is to estimate *waste-specific* emissions. Theoretically, a specific waste can influence the behaviour of its constituting elements in the landfill. For example lead in a glass matrix that is transferred to MSWI bottom ash will likely have a different range for its long-term transfer coefficient than finely dispersed lead from the incineration residue of a plastic. MSWI bottom ash is a heterogeneous material and the term 'slag' is somewhat misleading as only about 40 w-% consists of molten phases (Lichtensteiger 1995). About 50 w-% of the 'slag' contains material, which pass through the incineration grate without changes in mineralogy (ceramics, the core of larger metal parts, rocks from minerals like quartz, feldspar, calcite) (Lichtensteiger 2002). However, for inorganic landfills it is not yet possible to derive waste-specific transfer coefficients, as the mineralogical fate of constituents of all specific waste fractions is not known. Within this study all inorganic landfills have therefore *average generic* and not waste-specific transfer coefficients. In this model, all elements in all types of wastes feature an *average behaviour* as conceived for the elements in the average landfilled waste.

For the inorganic landfill model it is necessary to obtain a range for the long-term transfer coefficients. This range shall express the likeliness that an element is leached out from the landfill. The lower 2.5% confidence value or minimal value shall represent the fraction of the element that is emitted *at least*, i.e. even under most optimistic circumstances this fraction is most certainly emitted. The upper 97.5% confidence value or maximal value shall represent the fraction of the element that is emitted only under the most unfavourable conditions and thus represents a worst case. The mean value shall represent an intermediate case, ideally representing the conditions that are to be expected in an average or likely case. As long-term landfill development can currently not be observed or measured, the definition of the intermediate and best-case development case must of course remain speculative. For the worst-case, however, there is an upper limit: 100% discharge; i.e. the landfill cannot emit more pollutants than were present in the landfilled waste.

Application of the model for long-term behaviour

The following procedure is applied for the long-term development of residual material landfills and slag compartments. The model choices for the calculation of the *short-term* transfer coefficients of different elements are maintained. The leachate dynamics (exponential decline or constant) are *continued into the long-term future*. At some time, the acid neutralising capacity ANC of the landfilled material will be used up and the carbonate phase ends. The loss of ANC is mainly caused by washout of calcium and magnesium. If the carbonate phase ends, the pH drops and metal cation concentrations in leachate increase, cf. arrow 1 in Fig. 5.8. The concentrations of metal oxianions *decreases* as they are less soluble at low pH, cf. arrow 3 in Fig. 5.8. The washout of other easy soluble monovalent ions (Na^+ , K^+ , Cl^- , F^- , Br^- , I^-) and nitrate NO_3^- is unaffected by pH³⁵. The factor of concentration increase for metal cations at higher pH values is measured for some elements in (Johnson et al. 1996). The factor x_e is 158 for Cd, 100 for Cu, 251 for Pb, and 158 for Zn. For other elements an average value of

³⁵ See also footnote 15 on page 13.

167 is used³⁶. Oxianions are less soluble at low pH, i.e. the factor x_e is <1 and assumed to be 0.01. For silicon x_e is also 0.01 as it is only soluble as hydroxide at high pH, cf. arrow 5 in Fig. 5.8.

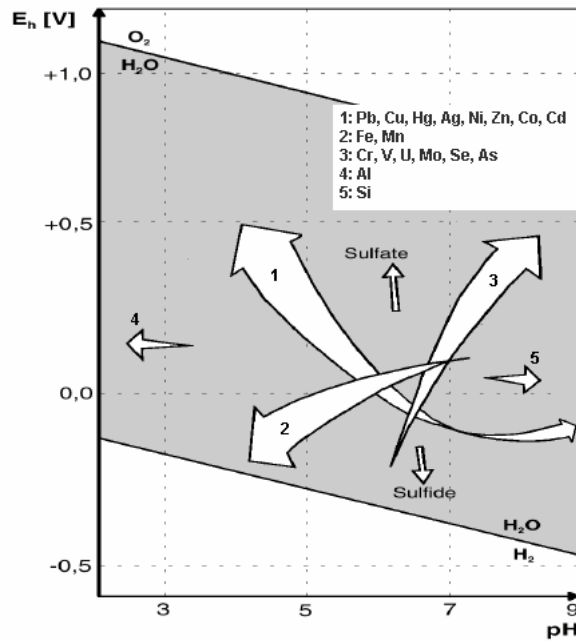


Fig. 5.8 Diagram of pH versus redox potential E_h and relative metal mobility. Arrows indicate directions of *increasing relative mobility* for the corresponding metals. Adapted from (Förstner et al. 1989)

The washout at low pH will fundamentally continue unlimited. There is fundamentally no end to washout other than the complete removal of material from the landfill site. All known effects can only delay complete washout (Sabbas et al. 1998). This is the reason to set the *worst case* or maximal value for long-term transfer coefficients to TK_{∞} . The worst case includes also all possible mechanisms leading to that result at a more accelerated pace: stronger surface erosion, landslides, faulting, intensified leaching from larger precipitation rates or other mechanical disruption of the landfill.

To calculate a *mean* transfer coefficient value, only the emissions from an intact landfill up until the next expected plateau-covering glacial period are heeded (see Fig. 5.9). This is assumed to represent a generic 'ecological planning horizon'. Glaciers reshape and remodel the landscape, break up the biosphere and thus create new boundary conditions for environmental goals. The emissions over 60'000 years therefore represent an expectation value of the burden inflicted on the ecosphere as we know it now and are concerned about. When glaciers cover the Swiss plateau (Mittelland), landfills will be eroded too and their contents redistributed in the landscape³⁷. The question when the next glacial period in Switzerland can be anticipated is discussed in the section 'The next glacial period in Switzerland' on page 30 and is answered there with 'approximately 60'000 years from now'. In case the

³⁶ These increases in concentration of two orders of magnitude are probably conservative. Kraxner et al. (2001:161) find increases of *three orders of magnitude* when moving from pH 8 to pH 4 for lead and zinc, and a bit lower for copper, manganese and iron. For some bivalent metals Me^{2+} solubility is controlled by hydroxide phases ($Me^{2+} + H_2O = 2 H^+ + Me(OH)_2$). Based on data from (Chandler et al 1997) Kraxner et al. (2001:162) derive a simplified rule for such metals by which each decrease in pH per unit increases metal concentration (as mg/l) by two orders of magnitude. An increase of 3 in pH therefore increase metal concentrations by *six orders of magnitude*. This latter rule however disregards formation of complexes and ion pairs.

³⁷ The landfill then ceases to exist, but its redistributed contents represent an environmental impact after the retreat of the glaciers.

carbonate phase lasts until the next glacial period occurs, the mean value will be calculated from the emissions that occur up to the next glacial period.

A possible *best case* or minimal value for the transfer coefficients can be derived from the following arguments. It is established that landfills emit pollutants and will continue to do so. During the carbonate phase a certain pollutant fraction is leached out, but after the carbonate phase a substantial pollutant potential still remains. After the carbonate phase, the remaining pollutants are washed out with increased rates due to a decrease in pH. The best case assumes that this pH drop does not occur, and total long-term emissions are *at best* as large as the emissions during the carbonate phase alone (see Fig. 5.9). In case the carbonate phase continues until the next glacial period occurs, the minimal value can be set equal to the mean value³⁸. The three values for best, average and worst case represent a triangular distribution.

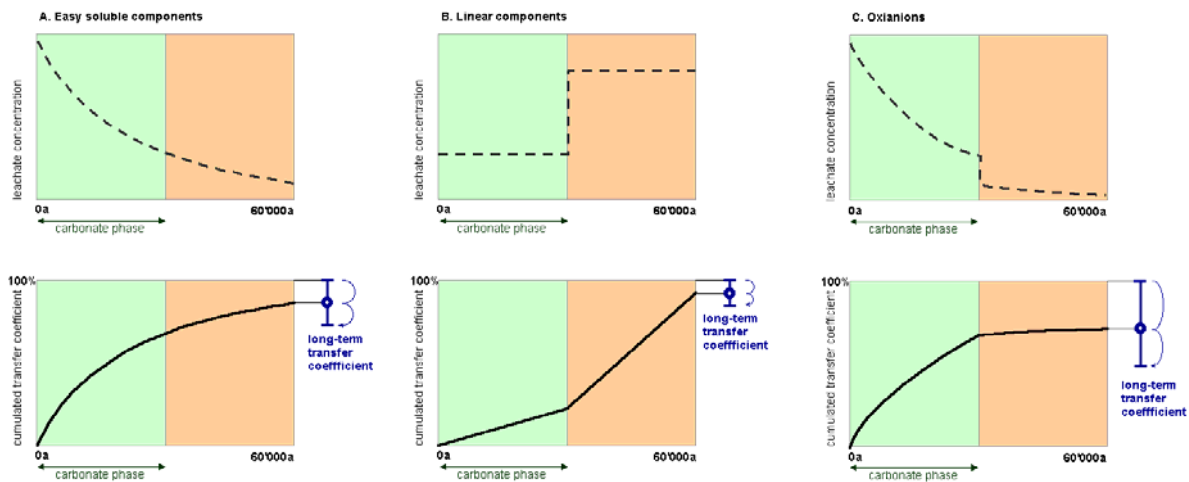


Fig. 5.9 Calculations scheme for long-term transfer coefficients for different elements: A.) Exponential decline for soluble elements with no change after the carbonate phase (left), B.) Constant concentrations for metal cations with an increase after the carbonate phase (middle), C.) Exponential decline of concentrations for metal oxianions with a decrease after the carbonate phase (right). The range indicated at the right side of each panel (blue) represents the derived lognormal uncertainty range for transfer coefficients.

With the choices made above, the three values for best, average and worst case actually represent a timeline in the modelled landfill development. The best case marks the end of the carbonate phase, the mean value marks the time of the next anticipated plateau-covering glacial period in Switzerland (the end of the landfill), and the worst case represents an infinite time³⁹. These marks in time complement the information of the short-term emissions which occur during the first 100 years after deposition. However it would be wrong to understand for example a best case value for long-term *emissions* as a value which merely discounts emissions after the carbonate phase. The best case emission value is not solely determined by the minimal long-term transfer coefficient alone, but also by the *best case waste*

³⁸ Due to late changes described in section 'Uncertainty of transfer coefficients and emissions' on page 36 this best case or minimal value will not be used in the model. Instead a lognormal distribution will be estimated from the medium and worst case value, as indicated in Fig. 5.9.

³⁹ Such long timeframes are already invoked in LCA for example in the LCIA stage. Modelled environments for calculating characterisation factors of the LCIA method CML'01 have in principle infinite timeframes (Guinée et al. 2001). In actuality they cover very long timeframes (e.g. millions of years for soluble pollutants like fluoride in marine water).

composition (i.e. lower boundary pollutant concentration in the waste). The best case is a value derived for optimistic scenarios for *all* determining factors.

For some elements the values of c_o or m are not known from literature. But the values for the long-term transfer coefficients can be calculated from the short-term transfer coefficients (STTK) derived above. For exponentially decreasing leachate concentrations the formula is:

$$\text{Eq. 5.11} \quad \begin{aligned} TK(t) &= TK_{\infty} \cdot (1 - e^{-k \cdot t}) \\ -k &= \ln\left(1 - \frac{STTK}{TK_{\infty}}\right) / 100a \end{aligned}$$

where STTK is the short-term transfer coefficient = $TK(t=100a)$

For constant leachate concentrations the formula is:

$$\text{Eq. 5.12} \quad TK(t) = \frac{STTK}{100a} \cdot t \quad \text{corrected to } TK(t) \leq TK_{\infty}$$

where STTK is the short-term transfer coefficient = $TK(t=100a)$

These formulas can be used to calculate the best case (minimal) transfer coefficient values, when t is set as the time the carbonate phase ends (= t_e). In case the carbonate phase would end after the next glacial period ($t_e > t_g$), these formulas are also used to calculate the mean value.

If the carbonate phase ends before the next glacial period ($t_e < t_g$), the leachate concentrations are altered by a factor x_e . The mean value of the transfer coefficients is then calculated heeding these post-carbonate concentrations. The time when the carbonate phase ends is t_e ; the time when the next glacial period occurs is t_g .

$$\text{Eq. 5.13} \quad \begin{aligned} TK(t_g) &= TK(t_e) + (TK_{\infty} - TK(t_e)) \cdot (1 - e^{-k \cdot x_e \cdot (t_g - t_e)}) \\ -k &= \ln\left(1 - \frac{STTK}{TK_{\infty}}\right) / 100a \end{aligned}$$

For linear washout the formula is:

$$\text{Eq. 5.14} \quad TK(t_g) = TK(t_e) + \left(\frac{TK(t_e)}{t_e}\right) \cdot x_e \cdot (t_g - t_e) \quad \text{corrected to } TK(t_g) \leq TK_{\infty}$$

The end of the carbonate phase

To calculate the minimal values of the long-term transfer coefficients, the time t_e when the carbonate phase ends needs to be known. Carbonate producing metals are calcium, magnesium, strontium, barium and others, but usually calcium and magnesium carbonates contribute over 99.5% to the carbonate buffer. To determine the time when the acid neutralising capacity is used up, the complete washout of calcium and magnesium is modelled. With a constant washout during the carbonate phase, the time when an element is completely leached out can be determined by Eq. 5.15.

$$\text{Eq. 5.15} \quad t_e = \frac{m_{Ca, Mg}}{c_{o, Ca, Mg}} \cdot V$$

When calcium and magnesium are washed out, the carbonate phase ends and the landfill enters a new stage with lower pH. Usually calcium is the limiting element, i.e. calcium takes more time than magnesium to be washed out.

The next glacial period in Switzerland

To calculate the mean values of the long-term transfer coefficients, the time t_g , when the next Swiss plateau-covering glacial period begins, needs to be known. In the last 2 Million years, 15 glacier advances were recorded in the Swiss alps (Schlüchter 2002). Approximately every 110'000 to 130'000a a glacial period occurs. Not all glacier advances did necessarily cover the Swiss plateau (Mittelland) with ice. These ice covers erode soil and ground and reshape the landscape. Essentially the ecosphere is reshaped. But also in an altered ecosphere and the resulting biosphere, release of pollutants – e.g. heavy metals from glacier rubble – will have an ecological impact. This supports an upper, worst case long-term transfer coefficient of 100%. Following timeline was compiled from (Schlüchter 2002).

- 180'000 to 120'000a BP (before present): the Riss glacial period: The Swiss plateau were largely ice-free.
- 115'000 to 10'000a BP: the Würm glacial period, where
- 100'000a BP the first Würm glacier advance occurred.
- 60'000 to 28'000a BP: the Swiss plateau were ice-free, with a steppe/tundra climate and a forest limit at 1800 meters above sea level.
- 32'000 to 10'000a BP: the second Würm glacier advance occurred. Forest limit dropped to 800 meters above sea level.
- After 10'000a BP the Swiss plateau became ice-free again until today.

Hence, in the last 200'000 years the Swiss plateau were iced over only twice, and on average, every 67'000 years. In the future this period could presumably be extended by present anthropogenic global warming. A period of 70'000 years between plateau-covering glacial periods is estimated here. 10'000 years since the last glacial period (second Würm advancement) have already passed. For the present study, a duration t_g of 60'000 years from now until the next plateau-covering glacial period is estimated.

Heterogeneity in landfill hydrology

Leachate does not flow homogeneously through a landfill. Regions with low water permeability can form which hinders leachate flow. On the other hand, preferential flow paths lets leachate flow relatively quickly through the landfill. This can lead to leachate which has hardly had exchange with the landfill body and is therefore not effective as a reactand media (i.e. media for washout of material). This slows down the landfill development as compared to a homogenous model with homogenous average infiltration. This postpones the end of the carbonate buffer phase further into the future. In the following paragraphs calculations with sample figures are performed.

Preferential flow in the leachate has a share of 10% to 50% in the total water discharge in a slag landfill (Hellweg 2000:77, based on a personal communication with Annette Johnson, EAWAG). The geometric mean is 22%.

If I is the rain water infiltration rate (in mm/a per m^2 area), then this volume must be (in the long-term average) released as leachate for reasons of mass balance⁴⁰. The water flows through the water pores of the landfill. The water pore volume $V\%$ of landfills is estimated to be 31%⁴¹. Assuming all water pores are open and connected the average cross section area of water pathways is 0.31 m^2 per m^2 landfill area. Since the cross-section area of the flow-paths is smaller than the landfill area receiving

⁴⁰ Neglecting any uptake or liberation of waste in the waste and lateral flow to or from the landfill body through seal leaks.

⁴¹ Calculated from a water content of 20% (Johnsson et al. 1998:370) and an average density of 1500 kg/m^3 .

the water, the leachate flow within the landfill must be *faster* than the infiltration speed at the surface suggests. The average leachate speed within the landfill is $I/V\%$. From this speed, the average residence time of the water in the landfill can be calculated. If the landfill has the height h in meters, then the *average* residence time of leachate water is T_a .

$$\text{Eq. 5.16} \quad T_a = \frac{h[m] \cdot 1000 \cdot V\%}{I[\text{mm} / \text{m}^2 \cdot \text{a}]}$$

T_a is approximately 0.6 years *per meter of landfill height*, and approximately 9.3 years for a landfill of 15 m height and infiltration of 500 mm/m²a. Again, in the long-term average, after the landfill has been saturated, the average residence time *must* be achieved, otherwise the landfill would absorb or generate water. Therefore, if a part of the leachate is preferential with a residence time much *shorter* than average, then the *rest* of the water must flow more *slowly* to compensate the effect of the preferential flow and to maintain the overall average residence time of leachate, respectively mass balance (cf. Fig. 5.10). If $w\%$ is the preferential part of the collected leachate and T_p is the residence time of preferential leachate, then the residence time T_s of the non-preferentially flowing water is expressed with:

$$\text{Eq. 5.17} \quad T_s = \frac{T_a - w\% \cdot T_p}{1 - w\%}$$

With a residence time T_p of 9 weeks (0.17a) and a share of preferential leachate $w\%$ of 22%, the residence time of non-preferentially flowing water is approximately 11.7 years or 1.3 times larger than the average residence time of 9.3 years.

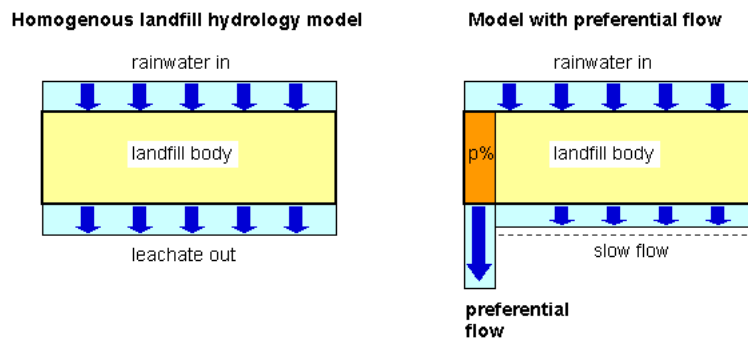


Fig. 5.10 Scheme of landfill hydrology with homogenous flow (left) and with preferential flow included (right). Not to scale

If 22% of the collected leachate (output) are obtained through preferential flow, it does not necessarily mean that 22% of the *landfilled contents* are exposed to preferential flow. A lot of 'fast', preferential water can pass through the landfill and still be in contact with just a marginal part of it (cf. Fig. 5.10). Residence times can in general be calculated by the formula 'stock [kg] divided by rate [kg/a]'. If we know the 'stock' of preferential water, i.e. the percentage of the total water *within* a landfill that is flowing preferentially, we can estimate which part of the landfill is actually in contact with preferentially flowing water. Let $p\%$ be the (unknown) part of the landfilled material that is exposed to preferential flow. Per 1 m² of landfill area there is a total mass of $1\text{m}^2 \cdot h \cdot \delta$ of landfilled material with density δ , with h being the landfill height. The average water content $v\%$ ⁴² in this waste column is

⁴² Not to be confounded with the water pore volume $V\%$ defined above.

20%. So, per 1 m² of landfill area there is a total water mass of $v\% \cdot 1m^2 \cdot h \cdot \delta$, which we will call M_w ⁴³. Accordingly, the 'stock of preferential water', i.e. the amount of water in the part of the landfill that is affected by preferential flow, is $M_w \cdot p\%$. The output rate of preferential water is part of the leachate output, that flowed preferentially, i.e. $I \cdot w\%$. The residence time for preferentially flowing water can therefore be written as:

$$\text{Eq. 5.18} \quad T_p = \frac{\text{stock}}{\text{rate}} = \frac{M_w \cdot p\%}{I \cdot w\%}$$

From that, $p\%$ can be isolated:

$$\text{Eq. 5.19} \quad p\% = \frac{T_p \cdot w\% \cdot I}{M_w} = \frac{T_p \cdot w\% \cdot I}{v\% \cdot 1m^2 \cdot h \cdot \delta}$$

Using the values derived above, a share of <1% of the landfilled material is actually in contact with preferentially flowing water⁴⁴. In other words, most of the landfill is in contact with non-preferentially flowing water.

So the effect of including preferential flow in the landfill model landfill is twofold. First, *less* water than in the homogenous model is in contact with the majority of the landfill body (>99%), since 22% are preferential water with little contact to the landfill. Second, the remaining, non-preferential water body is *slower* than suggested by a homogenous model. Almost all of the landfill body is in contact with this 'slow' water. Both effects delay the landfill development.

The equations to calculate the transfer coefficients (Eq. 5.7 and Eq. 5.8 on page 24) are defined for the annual leachate volume V [l/a*kg waste]. This annual leachate volume needs to be replaced by the annual *effective* leachate volume V_{eff} . The effective leachate volume is the water that interacts with the landfill and promotes weathering reactions, i.e. the non-preferential 'slow' flow. The annual mass of leachate water flowing non-preferentially is $I \cdot (1-w\%)$. The part of the landfill exposed to 'slow' flow is $(1-p\%)$. Per m² of landfill area, the mass of waste exposed to slow flow is $(1-p\%) \cdot 1m^2 \cdot h \cdot \delta$. The effective annual leachate volume is:

$$\text{Eq. 5.20} \quad V_{eff} = \frac{I \cdot (1-w\%)}{(1-p\%) \cdot h \cdot \delta} = \frac{1-w\%}{\left(\frac{h \cdot \delta}{I} - \frac{T_p \cdot w\%}{v\%} \right)}$$

with

I	Rain infiltration rate (~500 mm/m ² a)
w%	share of preferential flow in leachate output (22%)
p%	share of the landfill exposed to preferential flow (<1%), cf. Eq. 5.15
h	landfill height (15 m slag, 10 m residual)
δ	waste density (kg/m ³)
v%	water content in waste (20 w-%)
T _p	Residence time of preferentially flown water (0.17a)

With the above parameters, the effective leachate volume V_{eff} is 0.017 l/kg waste per year. This is volume is 20% smaller than in the homogenous modelling, and the carbonate phase therefore lasts 1.3 times longer than in the homogenous case. For each landfill type different V_{eff} are calculated in the corresponding chapters.

⁴³ It is assumed that water distribution within the landfill is homogenous.

⁴⁴ If an inhomogeneous water distribution is assumed and $v\%$ is increased, $p\%$ decreases.

It is assumed that the complete landfill body is exposed to 'slow' flow, i.e. the accelerated weathering in the small share of landfill body exposed to preferential flow ($p\%$) is neglected.

During the long-term time frame of 60'000 years climate changes could occur, which change *precipitation rates* and therefore speed of landfill weathering. Such changes in precipitation rates were not considered for the landfill model, i.e. the parameter I was kept constant over 60'000 years. The changes in precipitation until 2100 due to anthropogenic greenhouse gasses are estimated to be below 5% in most Swiss regions, based on the IPCC scenario A2, which has the largest amount of GHG emissions of all IPCC scenarios (EU 2008). This scenario however covers only 1/600-th of the timeframe of the landfill model. Due to reasons of time and effort no precipitation modelling over 60'000 years was attempted here.

During the long-term chemical weathering of the landfill body mass is removed from the landfill and changes in *landfill height* can be expected, i.e. the percolating water is affecting a *decreasing* mass of landfill contents. Such changes were not incorporated into the landfill model. The *overall* average mass transfer from landfills in 60'000 years is 31%, 68%, and 33% of the initially deposited waste mass (based on dry mass) for municipal waste landfill, slag compartment, and residual material landfill, respectively. The changes in height of the landfill body are therefore noteworthy. A decreasing landfill height would mean a larger amount of *effective leachate volume* V_{eff} (see Eq. 5.20 above) and thus accelerated emissions. However this effect can be counterbalanced by a decrease in *porosity*, due to landfill aging, which in turn increases the residence time of leachate and thus the annual leachate volume. These changes are neglected in the model and all parameters of landfill height, and the porosity-related share of preferential flow, waste density and water content are kept constant.

Heterogeneity in material

In the homogenous landfill model, it is assumed, that the landfill body is a homogenous mixture of water and solid material. In reality, the material has varying grain sizes, varying porosity and varying mineralogical composition. For large particles, the corrosion or weathering speed might become a rate determining step of washout, if these surface processes are slow enough. Corrosion of metals was observed to be quite fast (10 years for metallic iron). For glass particles, corrosion is thought to be slow⁴⁵. Accordingly, heavy metals embedded in glass matrices will only be released slowly. If the dimensions of solid, unfractured domains of glass are large enough, heavy metals could be bound in such material for long times, even if favourable conditions of their washout (low pH after carbonate buffer phase) would establish in the landfill. In MSWI bottom ash (slag), approximately 40 w-% is molten, glass-like material (Lichtensteiger 1995). The size distribution from sieving in Fig. 5.11 suggests that there are over 30 w-% in slag that have a size over 1 cm. Some of this coarse material is bulk metal, not glass. Lichtensteiger (1995:2) points out that the molten phases in slag, while having particle sizes of millimetres up to centimetres, usually feature small scale associations of different phases at or below the millimetre scale. Speiser (2001:3) describes MSWI slags as "porous matrix with mixed-in phases on a nanometer to micrometer scale". Speiser (2001:5) gives a typical pore volume of 50% for molten, glassy phases in MSWI slag. There is however no information on the degree of fracture of the coarse glass material.

⁴⁵ Personal communication with A.C. Johnson, EAWAG Dübendorf, September 12, 2002.

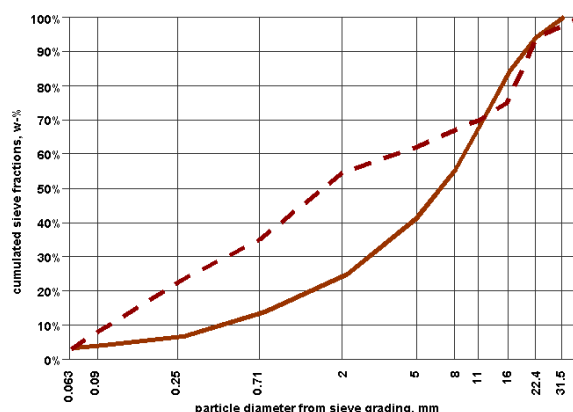


Fig. 5.11 Grading curve of MSWI bottom ash from size analysis with sieves. Adapted from Frühwirth et al. 2000:148 (hatched line) and Reichelt & Pfrang 1998:3(solid line)

It is important to note the difference between particle sizes derived from sieves and the unfractured bulk material dimensions, that are effective in corrosion processes. Fractures and open pores in the material can decrease the domain size, which is effective and applicable in corrosion estimates. Compared to homogenous, unfractured, unporous material, corrosion can be completed much earlier, if the material is fractured or porous. Since MSWI slag is usually quenched in water after the incinerator grate, i.e. transferred from 1000°C to <100°C, fractures are likely to occur.

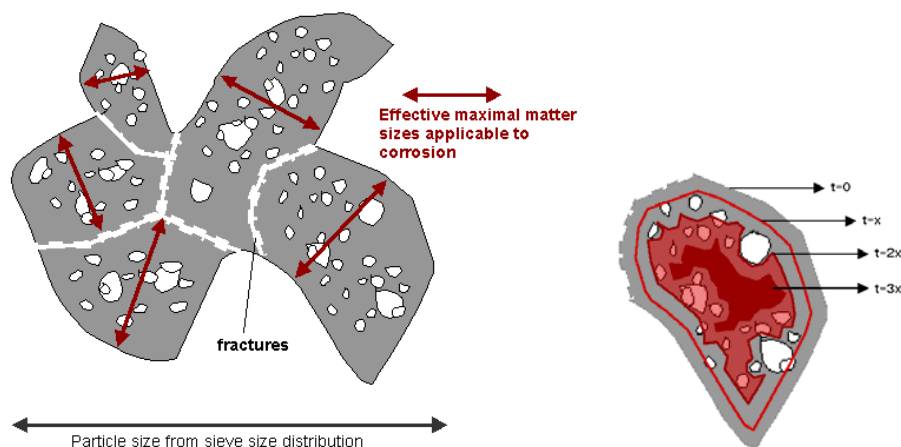


Fig. 5.12 Schematic of a glass particle showing the difference between the particle size from sieved size distribution and domain sizes applicable to corrosion (left). On the right the qualitative temporal dynamics of particle corrosion and the effects of closed pores are shown

Speiser (2001:12) points out that the corrosion of glass phases in MSWI slag landfills is much faster, than in natural glassy minerals, such as volcanic basalt. Conversions of glass phases via hydration to gels and subsequently conversion to clay minerals has been observed in slags only 12 years old. Heavy metals, formerly enclosed in the glass matrix, were converted to fine suspensions of calcium salts and carbonates (Speiser 2001:12, Huber et al. 1996:90). Similar processes in natural glasses take thousands of years. MSWI slag, which was stored open-air for 3 months, already shows signs of glass corrosion and formation of new calcium-silicate-hydrates visible on a micrometer scale (Speiser 2001:143).

Gross et al. (1999) measured corrosion rates of commercial $\text{CaO}-(\text{Mg,Fe})\text{O}-\text{Al}_2\text{O}_3-\text{SiO}_2$ glass samples. Disks of glass were exposed to 120 days of distilled water with 0.154 mol NaCl at 37°C and buffered to varying pH values. These conditions are similar to those in a landfill. Landfill leachate has elevated

temperatures of approximately 25°C and contains 0.16 mol NaCl. The corrosion rates found⁴⁶ were between 0.19 mg/cm² (at pH 7.4) and 310 mg/cm² (at pH 3). At pH 9 a rate of 0.33 mg/cm² was found. During the carbonate phase of an inorganic landfill the leachate pH is approximately 8.3. Under these conditions, a glass corrosion rate in the range of 0.19 to 0.33 mg/cm² could be expected. With a glass density of 2200 kg/m³, a corrosion speed of 2.6 to 4.5 micrometers per year can be calculated. With corrosion speeds of this magnitude, a *large, unfractured, homogenous glass domain* with a diameter of 1 cm would be corroded within 1000-2000 years. Gross et al. (1999) report increasing corrosion speed with decreasing pH. This would signify that during landfill development the corrosion of glasses would increase as the pH drops.

Theoretical, thermodynamic calculations on corrosion speed of glass phases in MSWI bottom ash performed by Yan (1998:45) shows ranges between 12.3 and 0.08 micrometers per year, compatible with the ranges above. These are reported as initial rates that slow down over time.

These reported corrosion speeds are also compatible with observations about alterations in the surface domains of glassy blast furnace slags (Olbrich & Frischat 2001). The surface altered to a depth of 150 nanometers *within only 7 days* in an alkaline solution of 0.1 M KOH. This solution is considered representative for early pore water solutions in this material. Even the concentration of network elements (Al, Si), which constitute the glass matrix, is altered. This extrapolates to an alteration rate of 7 micrometers per year. This figure is not a corrosion or dissolution rate, and it is also daunting to extrapolate a 7-day result to years, because alteration rates might slow down over time. Presuming that a matrix alteration precedes corrosion, this rate is however compatible with annual corrosion rates in the micrometer range reported by Gross et al. (1999) cited above. These results also underline the dynamic and non-inert properties of glasses in the surface domain. Olbrich & Frischat (2001) observe surface transition from mellilite glass to pyroxene and further to forsterite glass *within one hour* in a solution of 0.1 M KOH. Alterations in the surface roughness are also visible on a micrometer scale. All these observations occur at a high pH of approx. 13. As noted before, the corrosion speed increases with decreasing pH (Gross et al. 1999).

An uncertain point is, which percentage of pollutants are contained in the glassy phases of MSWI slag. Corrosion of glass might be slow, but if heavy metals are predominantly contained in other more accessible phases or near the surface of glass particles, the delaying effect of glass corrosion won't be noticeable.

It can be concluded that the occurrence of glass phases in waste pose *some* barrier function that delays emissions into the future. Ultimately, also glasses will be corroded and their constituting materials will be further weathered. With the limited available information, the delay can not be quantified but seems to be in the range of tens to hundreds of years, depending on the size of unfractured, homogenous glass domains. This range of time seems small compared to the total long-term time period of 60'000 years modelled in this study⁴⁷.

The calculations of transfer coefficients in the present landfill model are based on the relation between total content of elements in the waste (*m*) and the *measured* concentrations of these elements in the leachate (*c_o*). Cf. Eq. 5.7 and Eq. 5.8 on page 24. Glassy phases, that would initially prevent elements from leaching out, are heeded in this *measured* leachate data. It was decided above not to correct for the effects of glassy phases. This negligence actually leads to *smaller* inventoried emissions, because any initial abating effect of glasses occurring in the first hundred years or so, is extrapolated *without*

⁴⁶ The probe contained SiO₂ 42.4%, Al₂O₃ 19.9%, TiO₂ 2.08%, FeO 5.92%, CaO 18.7%, MgO 6.5%, Na₂O 0.61%, K₂O 2.91%, P₂O₅ 0.5%, MnO 0.92%.

⁴⁷ Assuming a homogenous distribution of elements in the slag material, assuming that 40% of slag is glassy, and 30% of the glassy phase has (overestimated) domain sizes of over 1 cm with corrosion times of over 1000 years, the mistake in emissions over the course of 60'000 years from neglecting these glassy phases is less than 1%.

correction into the long-term future. In this respect, the whole landfill inventory profits from any abating effect of initially present glasses.

Categorisation of landfill emissions

In chapter 'Lifetime of barriers' on page 20 it was concluded that it is just a question of time until pollutants, that leaked into the ground below the landfill, reach the groundwater. In this study all long-term emissions >100 a are integrated over time (see section 'The default procedure for future emissions in LCA' on page 15). Consequently, retention times are not detailed in this study. It is assumed that after 100 years, the collection system fails, the base seal leaks and leachate is entering the ground below the landfill. Hence, the *long-term emissions are inventoried as emissions to groundwater* (exchange subcategory 'Water, ground-, long-term'). The exchange subcategory 'soil' would not be appropriate, as this normally refers to emissions to the soil *surface* with e.g. wind erosion which does not occur in deep soil.

The short-term emissions of inorganic landfills are expected to be collected and discharged into a surface water body. The exchange subcategory 'Water, river' is chosen for short-term emissions. The short-term emissions of sanitary landfills are collected, discharged into a sewer and treated in a municipal wastewater treatment plant.

Uncertainty of transfer coefficients and emissions

Transfer coefficients of landfills bear significant uncertainties, mainly because of the long time spans involved. Transfer coefficients represent the fate of a chemical element in a landfill. It is important to distinguish this *uncertainty in fate* from the *uncertainty in the waste composition*⁴⁸. The uncertainty of the waste composition is already heeded in the waste input. In defining an uncertainty value for transfer coefficients, one has to ask, how uncertain the landfill behaviour is even for a perfectly defined waste.

Similar to the transfer coefficients for incineration (see part II), it can be argued that a major pathway (i.e. a large transfer coefficient) has a low uncertainty, as this pathway seems to be popular and 'well travelled'. For example chlorine is well soluble in leachate and is likely to be washed out rapidly. The transfer coefficient is large and the heterogeneity in the landfill can have but a small influence on the overall chlorine emissions. On the other hand the outcome of a minor pathway can be influenced by many stochastic events of which there are many in the heterogeneous milieu of the landfill body.

The uncertainty for short-term transfer coefficients (ST) is assumed to follow a lognormal distribution. It is assumed that for minor pathways (i.e. for small values of the mean m_{ST}) the GSD_{ST}^2 is approximately 10; i.e. the confidence range for a ST value is between $m_{ST}/10$ and $m_{ST} \cdot 10$. The GSD for large m_{ST} decreases and approaches 1 (100%) for $m_{ST} = 1$. To derive GSD_{ST} values from m_{ST} values the following formula is used (cf. uncertainty calculations for TK in part II on incineration):

$$\text{Eq. 5.21} \quad GSD_{ST} = N \cdot \ln(m_{ST}) + c \quad \text{with } N = -0.18, m_{ST} \text{ as [kg/kg]} \text{ and } c = 1$$

The value of N is chosen so that the GSD^2 value for the smallest m_{ST} values are approximately 10. The smallest m_{STTK} are in the range 0.0006% to 0.001%. The lognormally distributed short-term transfer coefficients are multiplied with the lognormally distributed waste composition to obtain the *short-term emissions*.

⁴⁸ The uncertainty range shown in leachate development in Fig. 5.7 from (Belevi & Baccini 1989) on page 23 is entirely derived from the variability in the waste composition.

The uncertainty of the long-term transfer coefficient (LT) are given as a triplet of minimal, average, and maximal values (lower = l_{LT} , mean = m_{LT} , upper = u_{LT}). Those values were derived with the procedure described in section 'Long-term emissions' on page 25. These values represent a triangular distribution. It was first intended to use this distribution directly for transfer coefficients and accordingly to enter long-term emissions as a triangular distribution. Discrepancies in the implementation of triangular distributions in the ecoinvent 2000 database made it impossible to use this approach without violating the mass balance for the medium case⁴⁹. Instead a lognormal approach is used also for long-term transfer coefficients. It is assumed that the medium and maximum case value represent the mean and upper value of a lognormal distribution. The lognormal distribution has then a GSD² of (maximal/mean)⁵⁰.

The long-term coefficients expressed in Tab. A.7 on page 119f. express the *total* transfer for the short-term and the long-term emissions *together*. To avoid double counting the short-term emissions must be subtracted from the total long-term emissions. This subtraction is performed already on the level of transfer coefficients, i.e. transfer coefficients for the emissions >100a alone are calculated. These transfer coefficients are labelled ΔLT , i.e. the difference between the total long-term and the short-term transfer coefficients. Following procedure is applied.

The mean ΔLT is derived by subtracting the mean short-term transfer coefficient from the mean long-term transfer coefficient.

$$\text{Eq. 5.22} \quad m_{\Delta LT} = m_{LT} - m_{ST}$$

The maximal ΔLT is derived by subtracting the lower short-term transfer coefficient value from the maximal long-term transfer coefficient value.

$$\text{Eq. 5.23} \quad u_{\Delta LT} = u_{LT} - l_{ST} = u_{LT} - \left(\frac{m_{ST}}{GSD_{ST}^2} \right)$$

As explained above, the distribution of the ΔLT TK is assumed to be lognormal. The GSD of this lognormal distribution is derived from the mean and upper value:

⁴⁹ In the landfill model the average or medium case value of a triangular distribution is actually a Maximum Likelihood value MLV. It was originally intended to use this Maximum Likelihood Value as the mean (average) value for emissions in the inventory. However, for the ecoinvent 2000 database it is mandatory to enter the *arithmetic mean* of the triangular distribution (Frischknecht et al. 2003a). The arithmetic mean is expressed by (Min+MLV+Max)/3, i.e. the unweighted mean of the minimal, the maximal and the MLV (geometrically, the mean expresses the *centre of gravity* of the triangle, while the MLV is the *top corner*). In the originally intended model calculation, the MLV would have fulfilled mass balances, i.e. the use of MLV as the average value would assure that the emitted mass is not larger than the mean input mass. The *arithmetic mean* however can become larger than the MLV, if the distribution is left-skewed. Since for the maximum emission value the upper limit waste composition is used (and not the average) the *arithmetic mean* does not fulfil mass balances. With the *arithmetic mean* as the average case for emissions (as prescribed for triangular distributions in ecoinvent 2000) the mass balances cannot be fulfilled. This shortcoming has led to abandoning triangular distributions in this report altogether and some late changes.

⁵⁰ The transfer coefficients will in reality not follow a lognormal distribution. A transfer coefficient cannot go beyond the range of 0–100%. The lognormal distribution is always left-skewed, has a long tail to the right and has a non-zero probability for values larger than 100%. For transfer coefficients distribution that is right-skewed for mean values close to 100% would be more suitable (e.g. the beta distribution. Not available in ecoinvent 2000). Also suitable is a discrete range between 0 and 100%, i.e. zero probability outside $0\% \leq x \leq 100\%$. In the model, however, distributions will be narrow for most elements (i.e. the mean is close to the upper boundary value) and the probability for values larger than 100% is very small.

$$\text{Eq. 5.24} \quad GSD_{\Delta LT} = \sqrt{\frac{u_{\Delta LT}}{m_{\Delta LT}}} = \sqrt{\frac{u_{LT} - \left(\frac{m_{ST}}{GSD_{ST}^2}\right)}{m_{LT} - m_{ST}}}$$

The long-term emissions after 100a are then calculated by multiplication with the landfilled waste composition. Both factors are lognormally distributed and the GSD of the resulting emission is calculated by using the uncertainty propagation formula for multiplication.

This procedure neglects the information provided by the minimal TK value in Tab. A.7 on page 119f. The minimal TK value applied in the database is determined by the mean and the GSD value. I.e. the used minimal LTTK does *not* refer to the emissions, that occur up to the end of the carbonate phase, as discussed in 'Long-term emissions' on page 25. The deviations of this minimal value (ΔLT) to the minimal data expressed in Tab. A.7 on page 119f. (given from ST and LT) are of minor importance.

As explained in sections 'Short-term emissions' and 'Long-term emissions', the transfer coefficients are largely calculated from literature data on the average waste composition m and the average initial leachate concentrations c_o from slag compartments and residual material landfills. The literature data is shown in the appendix (Tab. A.2, Tab. A.3, and Tab. A.4 on page 112ff. in the appendix). For the 'common' heavy metals like Cd, Cr, Cu, Hg, Ni, Pb, and Zn there is usually a lot of data. For some more 'exotic' elements there is little data available. Given the large possible variations in waste composition and leachate concentrations, the uncertainty is high for poorly referenced elements, i.e. there is not enough data to allow averaging over several landfill sites. Still, this data is used – even if derived from few data points – as this represents the measured state of knowledge. The resulting uncertainty from inferior averaging is assumed to be of minor importance compared to the uncertainty ranges derived above.

5.6 System boundaries

The goal of this study is to inventory disposal of different waste materials in landfills and subsurface deposits, so that this data can be used in LCAs of waste-producing activities. The slag compartment and residual material landfill are also used for the inventory of waste incineration (see part II).

Fig. 5.13 shows the process chain for landfilling of municipal waste. First the waste needs to be transported from the location of the waste-producing activity to the landfill. *Within the scope of ecoinvent 2000 this transport must be inventoried by the waste-producing activity and not the waste disposal process.* To avoid double counting the waste inventories devised in this report, do not contain the transport of the waste⁵¹. In the sanitary landfill, leachate is collected the first 100 years and treated in a municipal wastewater treatment plant. The resulting wastewater treatment sludge is incinerated in a municipal incineration plant. The incineration residues are landfilled in slag compartments and residual material landfills.

⁵¹ This is in contrast to the waste inventories performed before. In (Frischknecht et al. 1996, Hellweg 2000, Doka 2002) the waste inventories already contained the transport to the landfill.

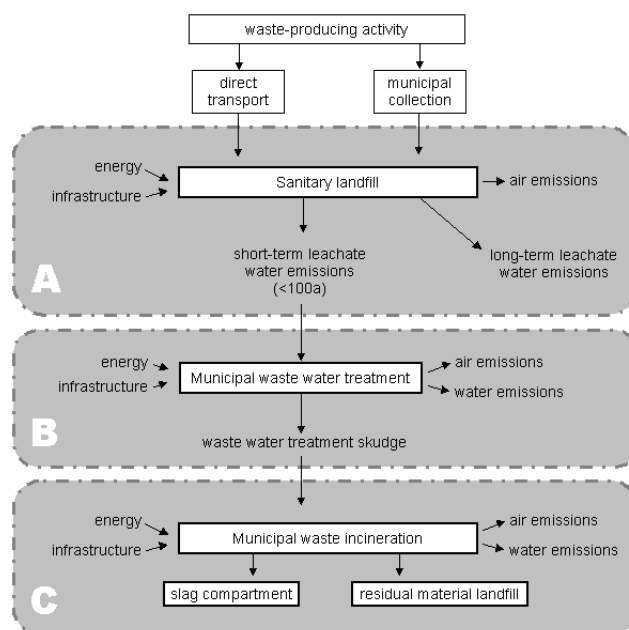


Fig. 5.13 Process chain for landfilling of municipal waste

The landfilling of waste leads to direct air and water emissions as well as land use burdens. Indirect burdens originate from the consumption of energy and infrastructure materials. The data modules devised in this study contain all the burdens from the landfill (A), and from the wastewater treatment (B) and from incineration and slag/residual landfills (C). The burdens associated with wastewater treatment are described in part IV of this report. The waste-specific emission data module also calls up two data modules for landfill infrastructure and for process-specific burdens, see Fig. 5.14. Both these modules are independent of waste composition.

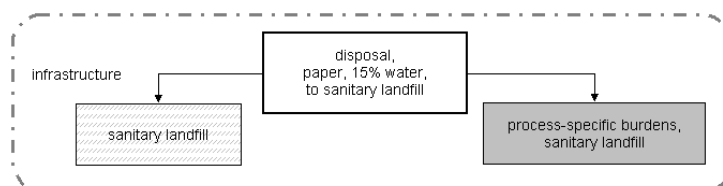


Fig. 5.14 System boundary and data modules for the inventory of landfilling of municipal waste

5.7 Necessary waste-specific data

To inventorise waste-specific burdens, the most important information within the landfill models is the elemental composition of the waste. An extensive vector of elemental concentrations in the waste can be specified by the user in the Excel calculation workbook on incineration '13_MSWIv2.xls' located on the on the CD-ROM. This data is given in kilogram element per kilogram wet waste. All composition data – also upper and lower heating values – refers to the *wet composition* of the waste. The corresponding water content can be specified for information purposes, although it has little consequence later in the calculations, but might be of significance if the heating values are extrapolated using the formulas of Michel (1938). For energy balance calculations, the upper and lower heating value must be specified in megajoules per kilogram wet waste. The lower heating value should be consistent with the given water content.

For disposal in sanitary landfills, the user must additionally specify the degradation rate during the first 100 years (See chapter 6.1.1 'Waste-specific degradability in sanitary landfills' on page 43).

Within the ecoinvent 2000 project, CO₂ and CH₄ air emissions are separated into emissions originating from biomass (paper, cardboard, vegetable waste, wastewater sludges) and from fossil material (plastics, mineral oil, solvents). The user must specify, what percentage of the given carbon content is biogenic.

For disposal in residual material landfills, the user can specify, if the waste is solidified with cement before deposition. The same solidification procedure as for MSWI residual material is assumed (50% residue, 30% cement, 20% water. See part II on incineration).

5.8 Waste heat from landfills

Decomposition of municipal waste produces a lot of heat. The decomposed material liberates waste heat from the microbial breaking up of chemical bonds. Some of the waste's energy is still contained as chemical energy in the landfill gas as methane. Some of the landfill gas is incinerated and the methane energy is liberated. In sanitary landfills the full upper heating value of the waste *without the chemical energy in non-incinerated methane* is inventoried as waste heat to air and water.

Also in inorganic landfills temperature gradients are observable due to e.g. carbonation of metal oxides and -hydroxides in ashes (cf. Fig. 5.4 on page 10). These energies dissipate and represent real waste heat emissions of such landfills. However, the energy needed for the formation of such energy containing ash minerals originates from the incineration of waste in the waste incinerator. The full waste heat amount of the waste is already inventoried in the incineration process, i.e. storage of chemical energy in incineration residues was not heeded in the incineration in accordance with the ecoinvent 2000 methodology (Frischknecht et al. 2003a). To avoid double counting of waste heat, no waste heat emissions from incineration slag and residue landfills are inventoried. The case of heat from corrosion processes of metals in waste is discussed in part I on incineration for steel and aluminium metal waste to municipal waste incineration.

5.9 Landfill infrastructure

In landfills some construction materials are used for ground work, base seals, flank seals and leachate collection systems that can be regarded as the 'infrastructure' of a landfill, while the landfilled waste is the 'product' of the landfill. In most industrial processes the infrastructure or production means are used to perform a desired task on the product. The product enters, proceeds through and leaves the infrastructure. The infrastructure is removed or replaced when aged. By contrast, the landfill infrastructure remains on site after landfill closure and is *not removed*. In a manner of speaking the infrastructure becomes part of the landfill. Also the landfill 'product' – the waste – remains in the 'factory' – the landfill – and is not removed. The landfill infrastructure is more like a permanent 'packaging' or 'property control device' for the waste, and much more closely combined with the 'product' – the waste – than production infrastructure is with its product.

In the ecoinvent 2000 project, infrastructure must be inventoried in separate data modules. This allows for LCI calculations with or without infrastructure. The landfill infrastructure is inventoried accordingly in a separate data module. Users who wish to use LCI data without infrastructure should check the compatibility of their rationale of doing so with the fact that the 'stationary' infrastructure of landfills is not included in such data.

5.10 Land occupation and transformation of landfills

Land occupation is inventoried separately from land transformation in ecoinvent 2000. Usually in industrial processes, land occupation is associated with infrastructure, e.g. areas occupied by factories. In landfills, the land occupation occurs not by mere infrastructure, but by the 'product', i.e. the waste itself during deposition. Land occupation and transformations for landfills are therefore not inventoried within the landfill *infrastructure* module (see chapter 5.9 'Landfill infrastructure'), but along with the *process expenditures*⁵². In this respect, land use of landfills resembles agriculture or open pit mining of coal, where the product itself leads to land use and not the production means.

5.10.1 Ecoscarcity eco-factors for occupied landfill volume

For land transformations and occupations associated with landfills the surface type with the CORINE code 132 ('dump site') is suitable. In ecoinvent 2000 this code is differentiated into several types for several near-surface landfill types (codes 132b-132e in Tab. 5.1). This is not to suggest, that the ecological quality of these landfill types are significantly different⁵³. The reason for this differentiation is that in the LCIA method of ecological scarcity⁵⁴ a midpoint eco-factor per kilogram landfilled waste exists (BUWAL 1998). Ecoscarcity does not differentiate the different surface landfill types, so the *same* eco-factor is applied to each kilogram of landfilled waste⁵⁵. This eco-factor is 500 eco-points/kg. In the LCIA calculations of the ecoinvent database, direct valuation of technosphere processes is not possible. It is therefore not possible, nor pragmatic the valuate each waste module 'Waste XY in landfill type Z' with this eco-factor directly⁵⁶. To be able to fully assign the Ecoscarcity method in ecoinvent 2000, an approach via the landfill land area was chosen. The landfill land area is inventoried with every landfill waste module. The necessary land area for the landfilling of one kilogram waste can be calculated from the landfill depth and the waste density. This area is inventoried in the database as a transformation to and from a landfill area (in m²) and as an occupation of landfill area for the duration of the landfill operation (in m²a) for each kilogram of waste. Since the average depth and waste density is different for each landfill type, different areas per kilogram waste result. Since the concerned area is inventoried directly as a land transformation, it is possible to attach an adapted waste eco-factor in 'eco-points per m² landfill area' to the inventoried landfill area transformation⁵⁷. The adapted eco-factor must be differentiated for the different landfill types (see Tab. 5.1). Using these modified eco-factors, each kilogram landfilled waste will be attributed a constant burden of 500 eco-points.

⁵² One exception is the land use for the access road, which is considered to be an infrastructure part.

⁵³ Though a sanitary landfill with vermin and food wastes will have a different internal biodiversity and also a different impact on the biodiversity of the surrounding land than a inert material landfill. These effects are not quantified in this report.

⁵⁴ Synonyms: Ecoscarcity, BUWAL method, Swiss Ecopoints, German: Umwelt-Belastungs-Punkte, UBP'97.

⁵⁵ There are however different ecofactors for wastes to underground deposits (salt mines) and for radioactive wastes.

⁵⁶ This would be not pragmatic because each time a *new* landfill waste module were created, the LCIA calculation matrix for Eco-scarcity would have to be expanded to include that module.

⁵⁷ The area ecofactor is applied only to the 'transformation *to* dump site type Z' and not to 'transformation *from* dump site type Z'. Applying it to *both* would be double counting.

Tab. 5.1 Differentiated CORINE land types for landfills and ecofactors for landfill areas

CORINE code	Landfill type	Waste density kg/m ³	Landfill depth m	Kilogram waste per m ² landfill area kg/m ²	Ecofactor per m ² landfill area (from 500 eco-points/kg) eco-points/m ²
132	dump site (general)	–	–	20'000 ¹	10'000'000
132a	dump site, benthos ²	–	–	0.92 – 3.85	462 – 1925
132b	dump site, sanitary landfill	1000	20	20'000	10'000'000
132c	dump site, slag compartment	1500	15	22'500	11'250'000
132d	dump site, residual material landfill	1600	10	16'000	8'000'000
132e	dump site, inert material landfill	1500	15	22'500	11'250'000

1 The landfill type 'dump site (general)' is not used in this report, but elsewhere in ecoinvent 2000 (e.g. uranium milling tailings, hard coal mining, silicone plant infrastructure). The value from sanitary landfill was adapted.

2 Used for offshore drilling wastes spread on the seafloor. With 260 m² covered area per drilling meter and 240 – 1000 kg waste per drilling meter.

5.11 Underground deposit volume occupation

In ecoinvent 2000, the volume occupied in underground deposits is inventoried as a consumption of a natural resource (Volume occupied, underground deposit). This exchange signifies the volume actually occupied by the waste and not the complete volume of the emptied salt vein⁵⁸. This exchange can be used to apply the eco-factor for underground deposits from the LCIA method of ecological scarcity (BUWAL 1998). In this method each kilogram of waste in a underground deposit is valued with 24'000 eco-points (irrespective of its composition). With an average density of 1600 kg/m³ (estimate), an adapted eco-factor of *38.4 Million eco-points per m³ occupied underground deposit volume* is calculated. This eco-factor is applied to the exchange 'Volume occupied, underground deposit, m³'.

No information on the complete volume of the inventoried underground deposit Herfa-Neurode is available. The complete deposit area is given as 20 km² (Brendel 2000). With an average ceiling height of 2.7 m a total volume of 54 million m³ can be calculated.

⁵⁸ There is usually some empty space between the deposited waste and the cavern wall, cf. Fig. 5.5 on page 12.

6 Life Cycle Inventory for sanitary landfills

6.1 Waste-specific emissions from sanitary landfills

When specifying a waste for sanitary landfills in this report, information is given about the degradability of the waste within the first 100 years. This information can be used to derive waste-specific transfer coefficients for short-term and long-term emissions.

- A waste that has zero degradability shows no decomposition in the first 100 years after the waste placement. There are no short-term emissions (from zero to 100 years) for such a waste. This waste foregoes active participation in the methane phase of the landfill and accordingly has no gas emissions either. All waste-specific pollutants are emitted as long-term emissions only.
- A waste that has 100% degradability is completely decomposed in the first 100 years after the waste placement. Decomposition means that the waste matrix is destroyed. However, destruction of the matrix does not equate to emission from the landfill body. Re-precipitation within the landfill body can lead to storage deposits and delayed emissions, for example metals that are liberated from a decomposed waste matrix can be precipitated in secondary solid phases. Only a fraction of the decomposed waste is actually released as emissions either to landfill gas or to leachate.
- A waste with intermediate degradability can be calculated as a linear mixture of the two extreme cases above. After 100 years, the pollutants of such a waste had one of the four fates (cf. left side of Fig. 6.1): a) not-decomposed, b) decomposed but re-precipitated and not released, c) decomposed and released in landfill leachate, or d) decomposed and released in landfill gas.

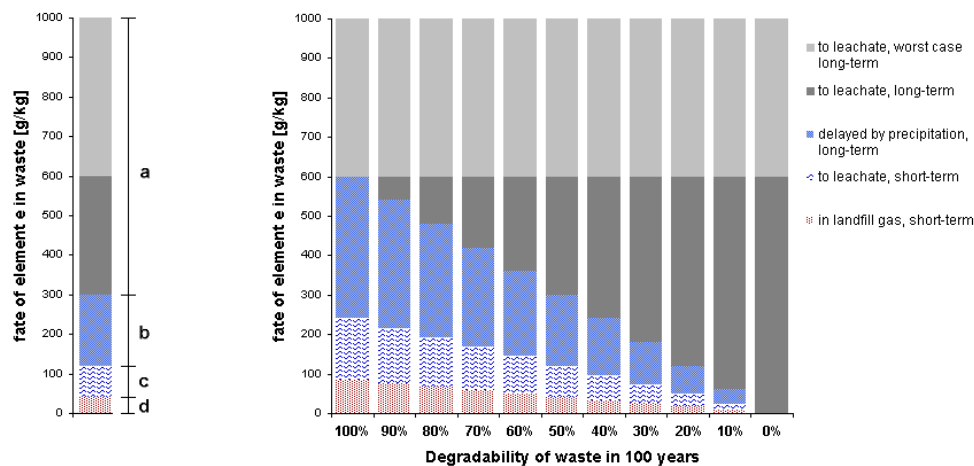


Fig. 6.1 Scheme of the fate of an element in waste depending on waste degradability. A long-term leachability of 60% was assumed

In the next chapters, I look at the decomposition rates of different waste fractions and subsequently an estimate of the re-precipitation of elements from decomposed waste.

6.1.1 Waste-specific degradability in sanitary landfills

To discern short-term emissions (<100 a) from long-term emissions (> 100 a), information on the degradability of a specific waste within the first 100 years after waste placement is needed. For example a landfilled plastic product will be much less degradable in 100 years than a biomass product

like paper or cardboard. Degradability refers to the decomposition and mineralisation of the materials in a waste matrix, which is usually a prerequisite for an emission of this material. In the former ETH LCI studies coarse assumptions on the degradability were made (Frischknecht et al. 1996:F.21, Zimmermann et al. 1996:B.69). It was assumed that biomass products like garden and kitchen waste, paper, cardboard, and wood could be *completely* decomposed and emitted within 150 years. This led to calculations that 97.1% of the carbon in these wastes would be transferred to landfill gas as CO₂ and CH₄, while 2.9% would be emitted via the leachate.

A closer look reveals that in 100 years a biomass product might disintegrate and lose its shape, but its components are not necessarily decomposed and mineralised. I.e. the decomposition rate should *not* refer to merely the disintegration of the shape of the waste. Carbon in biomass products consists of cellulose, hemi-cellulose and lignin. These materials have very different microbial degradability rates in the anaerobic environment of a sanitary landfill. Cellulose and hemi-cellulose are generally considered to be well degradable, but lignin is hardly decomposed. Under optimised laboratory conditions 71% of cellulose and 77% of hemi-cellulose in average municipal solid waste are degraded, but lignin decomposition is negligible (Micales & Skog 1997).

The degradability rates of cellulose, hemi-cellulose and lignin in a waste product are very dependent on their aggregation on a micro-scale. No general degradability rate for cellulose, hemi-cellulose and or lignin can be given. For example cellulose in newspaper, printer paper and filter paper will be decomposed by 40%, 90% and 97%, respectively. High contents in lignin can obstruct the decomposition of cellulose within the same waste. Lignin content in newspaper is 20% to 27% (Micales & Skog 1997). Also coatings of printed papers can delay decomposition.

For different types of waste, data is available on the transfer coefficient of carbon in the waste to carbon in landfill gas during the methane phase of a landfill⁵⁹. These figures can be taken as the waste specific transfer coefficient of carbon to landfill gas during the methane phase.

After the methane phase, degradation of the waste continues. Decomposed substances can be washed out with the leachate, but only negligible gas production occurs. During the first 100 years (including the methane phase) 2.9% of the decomposed carbon is emitted as dissolved matter in the leachate and 97.1% (= %gas_C) as gaseous carbon in the landfill gas⁶⁰. With this information it is possible to convert the carbon transfer coefficients to landfill gas into overall degradability rates of the waste through division by 97.1%.

⁵⁹ This figure is also called carbon (or microbial) conversion rate.

⁶⁰ There is no re-precipitation for carbon and 100% of the carbon from decomposed waste is emitted (see next 6.1.2 'Re-precipitation and the release factor').

Tab. 6.1 Transfer coefficient of carbon to landfill gas for several waste types (Micales & Skog 1997:151) and derived waste degradability in 100 years

Material	Carbon conversion rate	Waste degradability rate D in 100 years
Office Paper, tissue paper, towels	38.2%	39.34%
Cardboard boxes	31.5%	32.44%
Paper average	26.2%	26.99%
Coated paper	17.5%	18.02%
Newsprint/newspaper	15.7%	16.17%
Wood	0% to 3.2%	0% to 3.30%
Plastics, Paints	–	1%¹
Compostable materials	–	27%¹
Textiles	–	12%¹
Bulk metals	–	50%²
Gypsum	–	100%³
Wood ash	–	5%¹
Sludges	–	60%¹

1 Estimate

2 Estimate base on high corrosion rates. Re-precipitation of metals diminishes the actual emissions. Cf. chapter 6.1.2 'Re-precipitation and the release factor' on page 45.

3 Estimate based on gypsum solubility. Re-precipitation of sulfur diminishes the actual emission. Cf. comment on gypsum disposal as fine fraction in part V on building materials.

The degradability rates above are assumed to be homogenous, i.e. all elements are liberated from the waste matrix with the same rate. This is a simplification, as it is feasible for water-soluble elements to be washed off by leachate water even from an intact waste matrix. This effect is actually observed and introduced to the model in the next chapter 6.1.2 'Re-precipitation and the release factor' on page 45.

The degradability rates in Tab. 6.1 are significantly below 100%, as was assumed in former ETH LCI studies (Frischknecht et al. 1996:F.21, Zimmermann et al. 1996:B.69). This agrees well with observations about landfilled newspaper which were still legible after 20 years, or intact wood from 151 year old landfills in New York (Belevi & Baccini 1989a:47, Micales & Skog 1997:153).

A negative effect of the small degradability rate is that landfilled waste produces only a fraction of the energy recovered, if the same waste were incinerated. Less than 40% of the carbon in waste is transferred to the landfill gas. In the biogas already half of the carbon is present as energetically unusable CO₂. Also not all of the landfill gas can be collected and converted to energy for technical and logistic reasons. On the other hand, over 60% of the landfilled carbon end up in soil and water, which represent a partial sink, while in incineration almost all carbon is converted to CO₂, adding to the greenhouse effect.

For non-biomass wastes, generic degradabilities from Tab. 6.1 are applied. For plastic wastes a degradability of 1% is assumed based on estimates in (Zimmermann et al. 1996:B.69). For natural products (textiles, compostable material) a degradability of 27% (like paper average) was assumed.

The uncertainty of the degradability rate (GSD) is set to 100%, i.e. no uncertainty variation. It is assumed that the uncertainty of the short-term transfer coefficients already cover the possible variations in the decomposition process (cf. section 'Uncertainty of transfer coefficients and emissions' on page 36.)

6.1.2 Re-precipitation and the release factor

As pointed out above, *decomposition* of a waste does not necessarily mean that the decomposed part is actually *released* as an emission from the landfill. Re-precipitation occurs within the landfill, which

binds pollutants from decomposed waste matrices in secondary solid phases and postpones emissions into the future. This effect is important as it affects the relative magnitude of short-term vs. long-term emissions and also the ratio of air vs. water emissions.

Precipitation causes a discrepancy between the amount of waste that is decomposed (destroyed waste matrix) and the actual emission from the landfill (waste components emitted in gas and leachate, cf. Fig. 6.1). In other words, only a fraction of the decomposed waste is actually released as an emission. Apart from the waste-specific degradation rate, a *release factor* r_e is defined here to calculate emissions from the landfill. This release factor r_e is different for every chemical element.

It is possible to estimate average release factors r_e from the compositions of waste fraction in average municipal solid waste and the observed emissions from municipal solid waste landfills. On one hand the elements present in the *decomposed parts of waste fractions* can be calculated. These elements are liberated from the waste matrix and can potentially be emitted. In other words, it is assumed that an element needs first to be liberated from the waste matrix in order to be emitted. This theoretical emission potential P_e of the element e from 1kg of average waste is calculated as follows.

$$\text{Eq. 6.1} \quad P_e = \sum_{\text{fraction } i} D_i \cdot m_{e,i} \cdot \%i$$

where,

- P_e Theoretical emission potential of the element e from 1kg of average MSW, [kg /100a]
 D_i Decomposition rate of waste fraction i , [% / 100a]
 $m_{e,i}$ Concentration of element e in waste fraction i , [kg/kg]
 $\%i$ Share of fraction i in average municipal solid waste, [weight-%]

Waste-specific degradation rates D_i for different waste fractions in average municipal solid waste were derived in chapter 6.1.1 'Waste-specific degradability in sanitary landfills' on page 43. Weight shares and compositions of waste fractions in average municipal solid waste are listed in the appendix of part II on incineration.

The theoretical emission potential P_e of the element e can be expected to differ from the *actually occurring* leachate emission E_e . The *actually occurring* emission E_e is the potential emission P_e taking into account the release factor r_e for re-precipitation and also the fraction that is emitted via landfill gas as expressed by the parameter $\%gas_e$.

$$\text{Eq. 6.2} \quad E_e = P_e \cdot r_e \cdot (1 - \%gas_e)$$

where

- E_e Leachate emissions of the element e from average MSW, [kg /100a]
 P_e Theoretical emission potential of the element e from average MSW, [kg /100a]
 r_e Average release factor for element e , [kg/kg]
 $\%gas_e$ Fraction of the released amount of element e emitted in landfill gas [weight-%]

Information on $\%gas_e$ for different elements is obtained from (Belevi & Baccini 1989a), see Tab. A.5 on page 117 in the appendix. The remaining unknown variable is the sought after release factor r_e . Since the decomposition rate D_i is defined for 100 years, the emissions E_e are occurring over 100 years.

These *calculated* emissions can be compared with *measured* emissions from sanitary landfills. From several literature sources measured concentrations of pollutants in sanitary landfill leachate compiled (cf. Tab. A.1ff. on page 111). From this data the actual leachate emissions occurring over 100 years (= E_e) are estimated with following simplifications. The leachate concentrations c_o in the landfill are

assumed to be constant in the first 100 years. The geometric mean of the reported leachate concentrations is used for the concentration c_o . Of course, in reality landfill leachate emissions have large variations and are constant, show exponential decline or bell-shaped dynamics (cf. Fig. 6.2). The 'noise' in measured leachate concentrations, i.e. temporal variation in concentrations, is very large. Also landfill sites are very heterogeneous regarding the composition of the landfilled waste and the development of the landfill. With a literature survey of landfill leachate concentrations the right order of magnitude for a generic concentration can be estimated. With such a coarse estimate, neglecting the exact temporal dynamics seems justifiable.

Using these simplifications the leachate emissions during the first 100 years can be calculated.

$$\text{Eq. 6.3} \quad E_e = c_o \cdot V \cdot 100a$$

where

E_e	Leachate emissions of the element e from average MSW, [kg /100a]
c_o	Average leachate concentration of element e from literature, [kg/l]
V	Mean annual leachate output from the landfill per kg waste. Assumed to be constant [l/a per kg waste].

The mean annual leachate output V , is calculated from generic landfill data. The landfill height h is 20 m, and the waste density δ is 1000 kg/m³. Other data is adopted from slag landfills: the rainwater infiltration rate I of 500 l/m²a is assumed. The water content of saturated waste $v\%$ is assumed to be 25%. The share of preferential flow in leachate output $w\%$ is assumed to be 22%. The residence time for preferentially flowing water T_p is 0.17 years (9 weeks). With Eq. 5.20 on page 32, an effective annual leachate output V_{eff} of 0.0196 litre per kilogram waste is calculated, which is used for V in Eq. 6.3.

The actual landfill emissions calculated from measurements in Eq. 6.3, can now be compared with the synthetic, bottom-up calculation from Eq. 6.2. From that comparison, r_e can be isolated, cf. Eq. 6.4. The calculated values for r_e for different elements are shown in Tab. A.6 on page 118 in the appendix.

$$\text{Eq. 6.4} \quad E_e = P_e \cdot r_e \cdot (1 - \%gas_e) = c_o \cdot V \cdot 100a$$

from that

$$r_e = \frac{c_o \cdot V \cdot 100a}{P_e \cdot (1 - \%gas_e)} = \frac{c_o \cdot V \cdot 100a}{\left(\sum_{\text{fraction } i} D_i \cdot m_{e,i} \cdot \%i \right) \cdot (1 - \%gas_e)}$$

Discussion of the calculated release factors

For some soluble elements like chloride or sodium the calculated release factor r_e is larger than 100%. I.e. for these elements the actual estimated emission E_e is larger than the theoretical emission potential P_e . A likely explanation is that for those elements wash-off is possible *even with an intact, undegraded waste matrix*. In other words, the prerequisite noted above that in order to be emitted the waste must be degraded does not hold for such elements. For example salts can be washed off from a paper matrix with the paper matrix still being intact. For these elements emissions occur faster than the decomposition of the element-bearing waste fractions suggests. This effect represents a plausible characteristic of certain chemical elements and the found values of r_e larger than 100% are used for those (B, Cl, Br, I, Ba, Mn, Na).

The calculated release factors r_e are close to 100% for carbon (as TOC) or nitrogen (as ammonia NH₄ or as organic bound nitrogen N_{org}) – i.e. hardly any precipitation occurs. The release factor for carbon

is set to 100% (from 109%) to have carbon releases consistent with the calculations in chapter 6.1.1 'Waste-specific degradability in sanitary landfills' on page 43⁶¹.

Release factors r_e for transition metals are usually below 10%, i.e. these metals are precipitated to a large extent in the landfill – possibly as metal sulfide precipitates – and are not fully emitted during the first 100 years.

The found release rates are specific to the conditions within the landfill body and specific to the chemical element once it is liberated from the waste matrix. The release rates are therefore the *same for every waste fraction* and are not waste-specific.

It seems remarkable that calculations using the highly variable field data for heterogenous landfill bodies from different sites, from different years, with different waste compositions, different climate (precipitation), landfill thickness etc. yields results that seem to make *chemical* sense for a lot of elements. It is likely that release factors r_e of some elements were misjudged in this first approach. However, the introduction of release factors is a valuable new component to achieve more realistic landfill models for LCI.

6.1.3 Short-term emissions

Based on the calculations in the previous chapter, the waste-specific discharges during the first 100 years after waste deposition (short-term) can be given as follows.

The decomposition rate D of the waste determines how much of the waste matrix is destroyed in the first 100 years. However only part of that is actually released as an emission, because re-precipitation within the landfill body occurs. The element-specific release rate r_e quantifies this effect. During the methane phase of the landfill, emission can occur as emissions to landfill gas or to leachate. The factor $\%gas_e$ expresses what share of the emission occur as gas emission. With this information, it is possible to calculate the short-term emissions.

$$\text{Eq. 6.5} \quad E_{gas,e} = m_e \cdot D \cdot r_e \cdot \%gas_e$$

$$\text{Eq. 6.6} \quad E_{leach,e} = m_e \cdot D \cdot r_e \cdot (1 - \%gas_e)$$

where

$E_{gas,e}$	Short-term emission of the element e to landfill gas, [kg /kg waste]
$E_{leach,e}$	Short-term emission of the element e to leachate, [kg /kg waste]
m_e	Concentration of element e in waste fraction, [kg/kg waste]
D	Decomposition rate of waste, [kg/kg in 100a]
r_e	Average release factor for element e , [kg/kg]
$\%gas_e$	Fraction of the released amount of element e emitted in landfill gas [weight-%]

The transfer coefficients can then be expressed as:

$$\text{Eq. 6.7} \quad TK_{gas,e} = D \cdot r_e \cdot \%gas_e$$

where

$TK_{gas,e}$	Short-term transfer coefficient of the element e to landfill gas, [kg /kg element]
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⁶¹ A carbon release factor r_c of >100% would suggest a larger carbon conversion rate than anticipated. The decomposition rate D of the waste is adopted to measured carbon conversion rates.

$$\text{Eq. 6.8} \quad TK_{leach,e} = D \cdot r_e \cdot (1 - gas\%_e)$$

where

$TK_{leach,e}$ Short-term transfer coefficient of the element e to leachate, [kg /kg element]

The release factors r_e can be larger than 100% for some soluble elements, i.e. for these elements emission is faster than the waste degradability suggests. In order to comply with the mass balance, the term $D \cdot r_e$ must not be larger than 100%. If for a specific waste with degradability D and an element with release factor r_e this term is larger than 100% it is corrected to 100%.

Both 'emissions' to gas and to leachate are not necessarily releases to the environment. Landfill gas is collected and incinerated (see chapter 6.1.5 'Speciation for air emissions' on page 53). The leachate emissions of the first 100 years are collected, discharged to a sewer and treated in a municipal waste water treatment plant. The leachate treatment is described in chapter 6.1.7 'Model for leachate treatment' on page 55.

Calculations for waste mixtures

The formulas Eq. 6.7 and Eq. 6.8 are derived for a *single waste fraction* with one *overall degradation rate* D , which applies to all elements e . In the Excel tool it is also possible to define mixtures of different waste fractions to disposal. If waste mixtures are calculated certain adaptations are necessary for the following reason.

Let's compare the two situations: A waste composed of a single waste fraction 1 and a mixed waste 2 composed of two fractions with different degradabilities. If the short-term emissions of the single waste fraction 1 are calculated, the overall degradation rate D applies to all elements. For example, for a paper waste with 50 ppm lead and a degradation rate of 22%, the degradation of lead in 100a is 11 ppm per kg waste (22%)⁶².

Say, the mixed waste 2 is composed of 30% paper (with 50 ppm lead and 22% degradation) and 70% plastic (with 90 ppm lead and 1% degradation). Waste 2 has then a total lead content of 78 ppm and an overall degradation of 7.3% (= 30%*22% + 70%*1%). The lead in this mixed waste is *not* degraded with the *overall* degradation rate of 7.3%, but with the D_{pb} of 5%⁶³, since more lead is contained in the less degradable fraction. For this reason, when defining waste compositions in the Excel tool waste composed of fractions with dissimilar degradation rates must be divided up into several, more homogenous fractions.

If mixtures of waste fractions are defined, the variable D in Eq. 6.7 and Eq. 6.8 must be replaced by the element-specific degradation rates D_e for the specified waste. In the case of a single waste fraction all D_e are equal to D .

6.1.4 Long-term landfill development

After the methane phase, the landfill continues to emit pollutants via leachate, but usually on a lower level. Emissions to air are negligible. Waste continues to be degraded and emitted, also precipitated secondary solid phases are washed out in the long term. After the acid buffer capacity is used up, a decrease in pH can lead to increased leachate concentrations. To estimate the fate of the elements after the short-term phase a development based on average waste is regarded. The results from this model

⁶² The 11 ppm are then multiplied by the release factor for lead r_{pb} to obtain the short-term emissions to air and water.

⁶³ Calculated from $(30\% \cdot 50 \text{ ppm} \cdot 22\% + 70\% \cdot 90 \text{ ppm} \cdot 1\%) / 78 \text{ ppm} = (3.3 \text{ ppm} + 0.63 \text{ ppm}) / 78 \text{ ppm} = 5\%$.

are then applied to the remaining amounts of elements of any specific waste. The schematic development of the landfill is shown in Fig. 6.2.

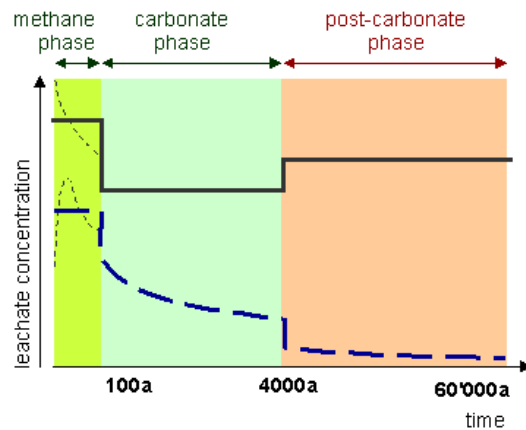


Fig. 6.2 Schematic of the modelled development of the leachate concentrations of a sanitary landfill for linear concentrations (upper line) and exponential decline (bottom line) with concentration changes after ending of landfill phases. Not to scale

The leachate concentrations up to 100 years after waste placement were modelled as being constant. After the first 100 years, a reduction of the leachate concentrations is assumed. A reduction factor x_s to express this reduction is derived from the initial leachate concentrations in slag compartments vs. initial leachate concentrations of sanitary landfills. I.e. after 100 years the leachate concentrations are assumed to decrease to the level of slag landfills. There are of course distinct differences between the chemistry in a slag landfill and a 100 year old sanitary landfill, for example content of organic carbon. Nevertheless, in absence of better data, the slag landfill concentrations can be used as being representative after the municipal waste has been stabilised to a certain extent and is at least not biologically reactive anymore.

The landfill enters a quasi-steady state. For the further development of the concentration levels following choices are made, which are based on the models for inorganic landfills (5.5.2 'Modelling of landfill emissions' on page 19). For easily soluble elements an exponential decline of the leachate concentration is assumed. Easy soluble elements are the monovalent ions (Na^+ , K^+ , Cl^- , F^- , Br^- , I^-), oxianion-forming elements (WO_4^{2-} , HVO_4^{2-} , $\text{Cr}^{(\text{VI})}\text{O}_4^{2-}$, MoO_4^{2-} , HBO_3^{2-} , HSeO_4^- , SbO_4^{3-} , HASO_4^{2-}) and also nitrogen (as nitrate NO_3^-). For all other elements (mostly cationic metals) a constant leachate concentration is assumed.

$$\text{Eq. 6.9} \quad \begin{aligned} TK(t_e) &= STTK + (TK_\infty - STTK) \cdot \left(1 - e^{-k \cdot x_s \cdot (t_e - 100a)}\right) \\ -k &= \ln\left(1 - \frac{STTK}{TK_\infty}\right) / 100a \end{aligned}$$

$$\text{Eq. 6.10} \quad TK(t_e) = STTK + \left(\frac{STTK}{100a}\right) \cdot x_s \cdot (t_e - 100a) \quad \text{corrected to } TK(t_e) \leq TK_\infty$$

To calculate the transfer coefficients up to the point where pH drops (t_e), Eq. 6.9 and Eq. 6.10 are used (derived from Eq. 5.13 and Eq. 5.14 on page 29).

After approximately 4500 years the pH of the landfill drops and leachate concentrations change again. Metal cation concentrations increase, while metal oxianions concentrations decrease as they are less soluble at low pH. The washout of other easily soluble monovalent ions (Na^+ , K^+ , Cl^- , F^- , Br^- , I^- , nitrogen) is unaffected by pH. The factor of concentration increase for metal cations at higher pH

values is measured for some elements in bottom ash landfills in Johnson et al. (1996). These factors are adopted here. The factor x_e is 158 for Cd, 100 for Cu, 251 for Pb, and 158 for Zn. For other elements an average value of 167 is used. Oxianions are less soluble at low pH, i.e. the factor x_e is <1 and assumed to be 0.01. For silicon x_e is also 0.01 as it is only soluble as hydroxide at high pH. Leaching is then continued until the next plateau-covering glacial period is assumed to be occurring ($t_g = 60'000$ years after waste placement).

$$TK(t_g) = TK(t_e) + (TK_\infty - TK(t_e)) \cdot \left(1 - e^{-k \cdot x_s \cdot x_e \cdot (t_g - t_e)}\right)$$

Eq. 6.11

$$-k = \ln\left(1 - \frac{STTK}{TK_\infty}\right) / 100a$$

Eq. 6.12

$$TK(t_g) = TK(t_e) + \left(\frac{TK(t_e)}{t_e}\right) \cdot x_s \cdot x_e \cdot (t_g - t_e) \quad \text{corrected to } TK(t_g) \leq TK_\infty$$

To calculate the transfer coefficients up to the next plateau-covering glacial period (t_g) Eq. 6.9 and Eq. 6.10 are used (derived from Eq. 5.13 and Eq. 5.14 on page 29).

Long-term transfer coefficients can be calculated for an average municipal solid waste composition using the formulas above. For a *specific* waste composition the variable short-term coefficients (depending on degradability) must be heeded, i.e. the amount of not-emitted elements after 100a is variable. The long-term emissions of the material not-emitted in the short term from a *specific* waste are assumed to be occurring in proportional amounts to the material not-emitted in the short term in *average* municipal solid waste.

Eq. 6.13

$$LTTK_e = STTK_e + (TK_\infty - STTK_e) \cdot \left(\frac{\emptyset LTTK_e - \emptyset STTK_e}{TK_\infty - \emptyset STTK_e}\right)$$

Eq. 6.14

$$(\emptyset)STTK_e = (\emptyset)STTK_{gas,e} + (\emptyset)STTK_{leach,e}$$

where

LTTK_e Long-term transfer coefficient of the element *e* for a specific waste, [kg /kg element]

STTK_e Short-term transfer coefficient of the element *e* for a specific waste, sum of gas+ leachate [kg /kg element]

TK_∞ Maximal long-term transfer coefficient of the element *e*, [kg /kg element]

∅LTTK_e Long-term transfer coefficient of the element *e* for average waste, [kg /kg element]

∅STTK_e Short-term transfer coefficient of the element *e* for average waste, sum of gas+ leachate [kg /kg element]

If STTK_e is small (low degradability of waste) then the LTTK_e much larger than the STTK_e, because after 100 years only a minor amount of the waste has been degraded, and a large pollution potential remains for >100a. If STTK_e is large (high degradability of waste) then the LTTK_e only slightly larger than the STTK_e, because after 100 years a large fraction of the waste has been degraded, and a relatively small pollution potential remains for >100a. Remember that total long-term transfer coefficients are referring to the *cumulated* emissions occurring after a long time period and therefore also include the short-term emissions.

Uncertainty in long-term transfer coefficients

In analogy to the inorganic landfills, a minimal, a mean and a maximal value for the long-term transfer coefficients are set at certain points in time:

As a mean value the transfer coefficients up to the next expected plateau-covering glacial period are heeded (after 60'000 years). There is fundamentally no end to washout other than the complete removal of material from the landfill site. All known effects can only delay complete washout (Sabbas et al. 1998). This is the reason to set the *worst case* or maximal value for long-term transfer coefficients to $TK_{\infty} = 100\%$ for all elements. For chromium a TK_{∞} of 25% is set, due to chromite stability (Huber et al. 1996:47).

The long-term coefficients described above refer to the time of zero to 60'000 years, i.e. also include the short-term emissions. To avoid double counting and to obtain the long-term emissions occurring after 100 years alone, the short-term emissions must be *subtracted* from the total emissions. The uncertainty ranges of the inventoried long-term emissions is derived from the procedure for inorganic landfills described in section 'Uncertainty of transfer coefficients and emissions' on page 36. In short, transfer coefficients for the emissions after 100a are calculated (ΔLT). This difference is obtained from the short-term and (total) long-term transfer coefficients. A GSD is estimated from a mean and a maximal case, i.e. the ΔLT transfer coefficients are assumed to be lognormally distributed. The leachate emissions after 100 years are obtained by multiplication of the ΔLT transfer coefficients with the waste composition. Uncertainty of the resulting emission (GSD) is calculated by using the uncertainty propagation formula for multiplication with the GSD values of the factors.

The minimal value of the long-term transfer coefficients as defined in section 'Uncertainty of transfer coefficients and emissions' on page 36 is not used in this approach.

Corrections for very large GSD values

In some cases this procedure can lead to very large $GSD_{\Delta LT}$ values, especially if $m_{\Delta LT}$ is very small. This is the case for easily soluble elements, which are leached in the first hundred years (e.g. sodium and chlorine); i.e. if $m_{STTK} \cong 100\%$ and $m_{\Delta LTTK} \cong 0\%$. There is however a non-zero chance that the short-term leaching of these substances is smaller than 100%, therefore there is also a non-zero chance that long-term leaching of these substances is larger than 0%. The situation arises, where the m_{LTTK} is close to 0%, but the upper value u_{LTTK} is much larger than 0%. With Eq. 5.24 on page 38 this leads to very large GSD values for $\Delta LTTK$.

The problem with these very large GSD values (i.e. in cases over 100'000%!) is, that in subsequent calculations these GSD values can dominate the uncertainty range. In the sanitary landfill process chain *two* sources for long-term emissions exist: the direct long-term emissions from the sanitary landfill and the indirect long-term emissions which originate from the landfilling of incineration residues of the incineration of wastewater treatment sludge from the treatment of short-term leachate from the sanitary landfill. These two long-term emissions are added to result in the final inventory figure. The uncertainty of this sum is calculated with the Wilkinson-Fenton approximation for sums. This approximation gives deviant values for very large GSD of the summands. I.e. a very large GSD can come to dominate the resulting GSD of the sum, irrespective of the contribution of the relative summand to the sum; i.e. even negligible summands can control the final GSD of the sum. In some examples the resulting GSD values in the inventory data suggested maximal long-term emissions of well over 1 kg of pollutant per kg of waste, which is not physically feasible. This error was only detected in the first Monte-Carlo calculation of the whole database, which was performed a few days before data publication in September 2003. In the few remaining days it was not possible to find a better approximation formula for the calculation of GSD of sums.

Since the involved exchanges are of no or minor ecological importance (sodium, chloride, nitrate) and the phenomena of bloated GSD values occurs only in a few modules to sanitary landfill (refinery

sludge, sludge from pulp and paper production, gypsum, tin sheet, aluminium, paper, packaging paper) a pragmatic ad hoc solution was used to avoid bloated GSD in the database: In cases where the mean m_{ALTK} is below 0.00001% the GSD was set to a fixed value of 400%. This correction is only required in cases where the majority of the pollutant is already emitted in the short-term, and was only implemented for the soluble species sodium, chloride, and nitrogen. **The mean value of the long-term emission is not affected by this correction.** However, the *maximal value* does not correspond to the value calculated according to the procedure described in section 'Uncertainty of transfer coefficients and emissions' on page 36. The resulting gap for the maximal emission is however below 5%. In the future a better way to operate with uncertainty of sums should be found, which was not possible in the current project due to pending deadlines.

6.1.5 Speciation for air emissions

The produced landfill gas is collected for incineration and energy production. However not all landfill gas is collected and only part of the collected landfill gas is incinerated for energy (cf. chapter 6.5 'Energy production' on page 62).

Of the carbon in raw LFG, 56% is $\text{CH}_4\text{-C}$ and 44% $\text{CO}_2\text{-C}$ (see Tab. 6.2). Nitrogen in raw LFG is elemental N_2 . Raw landfill gas is collected from an anaerobic environment. However, measured LFG compositions feature distinct contents of elemental oxygen (O_2). It is assumed here that the origin of oxygen in landfill gas is from atmospheric air, which has been drawn into the gas collection system, i.e. LFG contains not only gas from degraded waste. With an oxygen content in atmospheric air of 24 w-% it can be calculated that only 30% of the elemental nitrogen N_2 in LFG stems from waste, while 70% stems from atmospheric air. On the other hand, NO_x in combusted LFG can be expected to stem completely from the nitrogen in waste. The effect of correcting for atmospheric air in raw LFG is that the ratio NO_x/N_2 in combusted LFG is higher and more nitrogen from waste is emitted as NO_x (see Tab. 6.3).

Tab. 6.2 Raw landfill gas composition per cubic meter non-combusted gas

Raw LFG	Total ¹ Vol.-%	Total ¹ g/m ³	Part from drawn air ² g/m ³	Part from waste ³ g/m ³
CH_4	47%	335.7		335.7
CO_2	37%	726.8	(neglected)	726.8
N_2	13%	163	114	49
O_2	2.5%	36	36	

1 Mean value of five unreferenced literature sources in Zimmermann et al. 1996:B.79

2 Atmospheric air in LFG, based on an air composition of 24 w% O_2 and 76 w% N_2 and the assumption that all elemental oxygen in LFG is from atmospheric air.

3 Difference of the former two columns on the left.

In combusted LFG, carbon is converted to several carbon containing species (CO_2 , CO , CH_4 , NMVOC, PM10). A species profile is calculated from typical exhaust compositions from landfill gas burners and flares (see Tab. 6.3). CO_2 , CO , and CH_4 , are divided into fossil and biogenic emissions according to the share of biogenic carbon of the total carbon released during the short-term. Most of the carbon released during the short-term is biogenic, as fossil carbon is usually less degradable, i.e. D_i and $\%b_i$ are positively correlated. The biogenic share of the carbon released during the short-term ($\%b_{\text{waste}}$) is calculated according to the following equation.

$$\text{Eq. 6.15} \quad \%b_{\text{waste}} = \sum_{\text{fraction } i} D_i \cdot m_{c,i} \cdot \%i \cdot \%b_i$$

where,

- $\%b_{\text{waste}}$ Share of biogenic carbon of the carbon released during the short-term, [kg / kg*100a]
 D_i Decomposition rate of waste fraction i , [% / 100a]
 $m_{c,i}$ Concentration of carbon in waste fraction i , [kg/kg]
 $\%i$ Share of fraction i in average municipal solid waste, [weight-%]
 $\%b_i$ Share of biogenic carbon in waste fraction i , [weight-%]

Airborne particles are also allocated to the carbon in combusted landfill gas. Some nitrogen oxides are formed in combustion. They are fully allocated to the nitrogen in landfill gas. The vast majority of nitrogen in LFG is emitted as N₂.

Tab. 6.3 Calculation of species profile in combusted landfill gas. From unreferenced sources in Zimmermann et al. 1996:B.79

	Compounds per m ³ gas ¹ g compound per m ³ gas input to combustion	g element per m ³ gas	Species profile kg compound per kg element in combusted LFG
CO ₂	1650	450	3.66
CO	0.14	0.06	0.000311
CH ₄	0.01135	0.01	2.52E-05
NMVOG	0.00265	0.0023	5.88E-06
Particles	0.047	0.047	0.000104³
Total carbon		450.1	1
N ₂	49 ²	49	0.997⁴
NO ₂	0.419	0.128	0.00853
Total nitrogen		49.128	1

1 From unreferenced sources in Zimmermann et al. 1996:B.79.

2 Without N₂ from drawn atmospheric air.

3 Where 100% are PM_{2.5} (based on data for natural gas fuels in Lükewille et al. 2001:36)

4 Not inventoried in ecoinvent 2000.

Airborne sulfur is generated as hydrogen sulfide (H₂S) in the anaerobic environment of the landfill. It can be partly oxidised to sulfur dioxide (SO₂) in the top layer of the landfill. Emitted H₂S is quickly converted to SO₂ by atmospheric oxidation. If combusted in landfill gas, H₂S is also oxidised to SO₂ during combustion. All airborne sulfur from the landfill is inventoried as SO₂⁶⁴.

6.1.6 Speciation for leachate emissions

On categorisation of landfill emissions see section Categorisation of landfill on page 36.

Carbon emissions to water are inventoried *simultaneously* as TOC, DOC, BOD, and COD in the ecoinvent database (Frischknecht et al. 2003a). All carbon is assumed to be emitted as dissolvable organic carbon (DOC=TOC). From (Krümpelbeck 1999) a BOD/TOC ratio of 0.26 and a COD/TOC

⁶⁴ This speciation neglects the partial emission as H₂S. However, the acidification impact is depicted correctly. As H₂S is associated with landfill odour issues the odour impact is underestimated for sulfur-bearing wastes.

ratio of 1.09 are reported from measurements in 21 to 30 year old sanitary landfills. These ratios are used to calculate BOD and COD exchanges.

For sulfur and nitrogen, there are more than one emitted compound per element. 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). Average leachate concentrations from a literature survey are used to calculate a species profile for sulfur and nitrogen. Weight increases for compounds are calculated from atomic weights of the compounds vs. atomic weight of the emitted element.

Tab. 6.4 Calculation of species profile for emissions in leachate of a sanitary landfill

	Leachate concentrations from literature ¹ mg element/l	Compound profile for emitted element	Compound inventoried in ecoinvent as	Weight increase per compound kg compound / kg element	Factor per kg element kg compound / kg element
Sulfur as SO_4	40	93.14%	SO_4	3	2.794
Sulfur as H_2S	2.944	6.86%	H_2S	1.063	0.07284
Total S	42.95				
Nitrogen as $\text{NH}_4\text{-N}$	115.8	31.71%	NH_4	1.286	0.4077
Nitrogen as N_{org}	243.3	66.64%	$\text{N}_{\text{org-N}}$	1	0.6664
Nitrogen as $\text{NO}_2\text{-N}$	2.466	0.68%	NO_2	3.286	0.02219
Nitrogen as $\text{NO}_3\text{-N}$	3.536	0.97%	NO_3	4.429	0.04288
Total N	365.1				
Phosphor as PO_4		100.00%	PO_4	3.065	3.065

¹ see Tab. A.1 on page 111 in the appendix.

6.1.7 Model for leachate treatment

The leachate emissions of the first 100 years are not emitted directly to the biosphere, but are collected, discharged to a sewer and treated in a municipal wastewater treatment plant. The inventory of the treatment of leachate can be calculated with the WWTP model developed in part IV (wastewater treatment). However, provisions must be taken in order to sustain a waste-specific calculation (e.g. wastewater without cadmium shall have no cadmium emissions from leachate treatment). The following procedure is applied to calculate the waste-specific burdens from treatment⁶⁵.

Individual factors describing the burdens created from 1 kg of pollutant in 1m^3 wastewater are calculated *for each pollutant* in the leachate (C_{org} , NH_4 , N_{org} , NO_3 , PO_4 etc., see chapter 6.1.6 'Speciation for leachate emissions' on page 54). The model to calculate those burden factors is the WWTP model described in part IV of this report. This model not only contains the direct burden from the wastewater treatment process itself, but also the downstream burdens from sludge disposal. For the calculations performed here the sludge disposal is set to 100% incineration⁶⁶. For each pollutant X a list of burden factors $B_{i,X}^0$ for different exchanges i (emissions + required processes and materials) results.

⁶⁵ The procedure is similar – not equal – to the procedure applied to heed sludge incineration burdens in wastewater treatment (cf. part IV, chapter 5.1). The procedure there is more straightforward, as allocation to pollutants is easier than in the case of sludge incineration.

⁶⁶ Sludge spreading is being phased out in Switzerland 2003-2005 and landfilling is prohibited since 2000. So for the treatment of leachate over 100 years incineration of all sludge is the likeliest disposal option.

The burdens from wastewater treatment are fully additive, i.e. each pollutant can be calculated in a modular fashion⁶⁷. However, some of the burdens are not caused by the pollutants themselves, but by the 'carrier water'. I.e. even unpolluted water causes some burdens in the WWTP process, e.g. energy for pumping and heating, the sewer and plant infrastructure, transport and disposal of grit waste (plastic and biomass waste entrained in the sewer). These 'base burdens' $B_{i,W}$ are present *irrespective of the pollutant content*⁶⁸. The other burdens are linearly proportional to the pollutant content.

For the calculation of the leachate treatment burden following calculation is performed

$$\text{Eq. 6.16} \quad B_{i,X} = m_X \cdot (B_{i,X}^0 - B_{i,W})$$

where

$B_{i,X}$ Burden for exchange i from pollutant X in leachate

m_X Mass of pollutant X in leachate 0-100a (in kg)

$B_{i,X}^0$ Burden for exchange i from 1 kg pollutant X in 1 m³ of wastewater (from WWTP model)

$B_{i,W}$ Burden for exchange i from 1 m³ of *unpolluted* wastewater (from WWTP model) = 'base burden'

The used factors $(B_{i,X}^0 - B_{i,W})$ represent a sparsely filled (i, X) matrix of 215 rows (burdens) by 216 columns (pollutants + uncertainty columns). A representation in print would take up over 42 pages. The full matrix is contained in the sanitary landfill tool (file '13_MSOLFv2.xls' on the CD-ROM).

For all pollutants X in the leachate a list of burdens $B_{i,X}$ is obtained from the multiplication step according to Eq. 6.16. Most burdens $B_{i,X}$ in this list are zero. To calculate the total burdens B_i from leachate treatment, all $B_{i,X}$ are summed up over all pollutants, e.g. the total electricity demand from leachate treatment is obtained by summing up over all pollutants X generating an electricity demand. The base burdens created by the leachate volume over 100 years are expressed by $(B_{i,W} \cdot V \cdot 100a)$. The mean annual leachate volume V is $2.5 \cdot 10^{-5}$ m³ per year and per kilogram waste⁶⁹. Over 100 years a total leachate volume of 0.0025 m³ per kilogram waste are treated. The GSD of this figure is assumed to be rather small as variations in precipitation over 100a can be assumed to compensate each other. A GSD of 105% of the total leachate volume is assumed. base burdens and pollutant-specific burdens $B_{i,X}$ for all i are added up to yield the total burden for leachate treatment, as expressed in Eq. 6.17.

$$\text{Eq. 6.17} \quad B_i = B_{i,W} \cdot V \cdot 100a + \sum_X B_{i,X}$$

where

B_i Total burden for exchange i from leachate treatment

$B_{i,W}$ Burden for exchange i from 1 m³ of *unpolluted* wastewater

V Mean annual leachate output from the landfill per kg waste [m³/a per kg waste].

$B_{i,X}$ Burden for exchange i from pollutant X in leachate

⁶⁷ This is the main difference to the similar procedure for sludge incineration in MSWI, where certain expenditures are allocated between a list of causing agents (cf. part IV, chapter 5.1).

⁶⁸ These base burdens are responsible for the effect that the cumulated LCI results of wastewater treatment a e.g. cadmium-free wastewater can still show minor cadmium emissions.

⁶⁹ The mean annual leachate output V , is calculated from the landfill height (20 m), the waste density (1000 kg/m³) and a rainwater infiltration rate of 0.5 m³/m²a ($0.5/(20 \cdot 1000) = 2.5 \cdot 10^{-5}$ m³/kg*a). See also chapter 6.1.2 'Re-precipitation and the release factor' on page 45.

Uncertainty of leachate treatment

The burden data calculated with the WWTP model contains also uncertainties. I.e. the treatment of the leachate adds to the uncertainty of the landfill leachate output. To pass on this additional uncertainty the following procedure is applied.

First, the calculation of the burden factors $B_{i,X}^0$ and $B_{i,W}$ must be performed with a minimal input uncertainty. I.e. all uncertainty factors of the pollutant input in wastewater are set to $GSD^2 = 100\%$. This way, the uncertainty in the resulting burdens represents the uncertainty introduced *by the WWTP process and further downstream processes alone* and not the combined uncertainty of the wastewater plus WWTP process.

The resulting uncertainty values of the burden factors can be combined with the uncertainty values of the pollutants in leachate to obtain the uncertainty of $B_{i,X}$ ⁷⁰. For lognormally distributed burdens (with GSD^2) value, the uncertainty of the wastewater can be combined with this lognormal uncertainty using the formula for multiplication.

As for the mean values (cf. Eq. 6.17) the final uncertainty values for B_i are combined from the contributing burdens from all pollutants in the leachate. For lognormally distributed values the final GSD is calculated from the contributing GSD in the sum using the approximate formula of Wilkinson-Fenton for addition.

6.2 Landfill fires

Sanitary landfills contain a lot of burnable material. Conventional ignition sources like cigarettes or hot ashes, but also the large temperatures encountered during microbial decomposition can ignite the waste. Landfill fires on the surface are usually detected quickly and abated. Fires can also start or continue underground, where they are harder to detect and can continue for weeks or even months. Spontaneous underground fires can be a sign of inferior landfill construction, i.e. insufficient compaction, which allows ignitable methane pockets to form. Also important to prevent and fight underground fires is to minimise air supply to the landfill. In the United States an average frequency of 3000 landfill fires are reported to occur in MSW landfills and open dumps⁷¹ (USFA 2001). A Swiss landfill fire occurred in Raron VS on Mai 30, 2001⁷². The uncontrolled incineration of waste is very burdensome and releases a lot of air pollutants like nitrogen oxides, carbon monoxide, VOCs and dioxins – apart from posing a severe explosion risk during the methane phase of the landfill. Due to the larger emission factors compared to regular operation, landfill fires could be relevant even when they are rare. However, accurate frequency data is not available for Swiss landfills. Waste-specific emissions from landfill fires could be estimated with the inventory model for uncontrolled incineration of wastes (illegal, open burning) described in (Doka 2000).

⁷⁰ For burdens which contain 'base burdens' this involves a subtraction ($B_{i,X}^0 - B_{i,W}$). Uncertainty propagation during subtraction ($A-B=C$) is estimated with an experimental formula $GSD_C = (m_A * GSD_A^2 - m_B) / m_C$; where m_A is the mean value of A. The sheet 'leachate treat' in MSWLF.xls contains these difference values.⁷¹ Of 8300 reported incidents, 64% were burning waste containers, and not landfill fires.

⁷² Cf. newspaper report 'Walliser Nachrichten' on <http://www.vs-wallis.ch/unwetter/0501.html>. According to the address list of Swiss landfills from BUWAL, the landfill in Raron VS is supposed to be an inert material landfill storing inorganic material with a maximum of 5% carbon (BUWAL 2001a)!

In this study **landfill fires are not heeded**, due to lack of frequency data. Information on Swiss landfill fires is sparse and probably reflects a low frequency. In other regions of the world, landfill fires are known to be common (cf. section on waste scavengers in part I).

6.3 Infrastructure of sanitary landfills



Fig. 6.3 Construction of the sanitary landfill Chrüzlen near Oetwil am See, ZH, in 1995 with a capacity of 600'000 t (Chrüzlen 2002).

For the inventory the landfill shape is approximated with a rectangular box. The depth is assumed to be 20 m and the area 90'000 m². The landfill volume is thus 1.8 million m³. The average density of the waste is 1000 kg/m³. The landfill capacity is thus 1.8 million tons of waste⁷³. These figures are taken from the landfill Elbisgraben BL and are representative for average Swiss sanitary landfills (Zimmermann et al. 1996:B.57).

Humus, loam, tree trunks and earth needs to be removed from the surface. A volume of 1 m³ per m² landfill surface is estimated. The landfill is assumed to be partly submerged below the existing surface level and will rise above it after closure. So, additionally an estimated 50% of the landfill volume needs to be excavated. A total of 990'000 m³ of excavated material with a density 1500 kg/m³ is transported 20 km with trucks for recycling (cut-off boundary, i.e. no disposal of this material). Similar material is used for the recultivation after landfill closure.

The base and the flanks of the landfill need to be sealed watertight for the leachate collection system. In reality the flanks are inclined at an angle. For this estimate the flanks are assumed to be perpendicular to the base and have a height identical to the landfill height⁷⁴. The total outer surface to be sealed is 114'000 m². A layer of 80 cm of gravel of 1600 kg/m³ density is applied. A 7 cm layer of bituminous concrete (15% bitumen, 85% gravel, density 2500 kg/m³) follows, which is covered with a polyethylene PE sheet (0.25 mm thickness, density 960 kg/m³). The material thickness applied are the legal minimum defined in (TVA 2000). For the construction a specific energy demand of 0.5 litre diesel per m² sealed surface is inventoried (density 840 kg/m³, heating value 42.8 MJ/kg) (Zimmermann et al. 1996:B.77). A total of 2'481'000 MJ diesel in construction equipment are consumed.

⁷³ Swiss legislation prescribes a minimal size for new sanitary landfills of 500'000 m³ (TVA 2000:Art.31).

⁷⁴ The mistake in assessed surface is smaller than 1.5%.

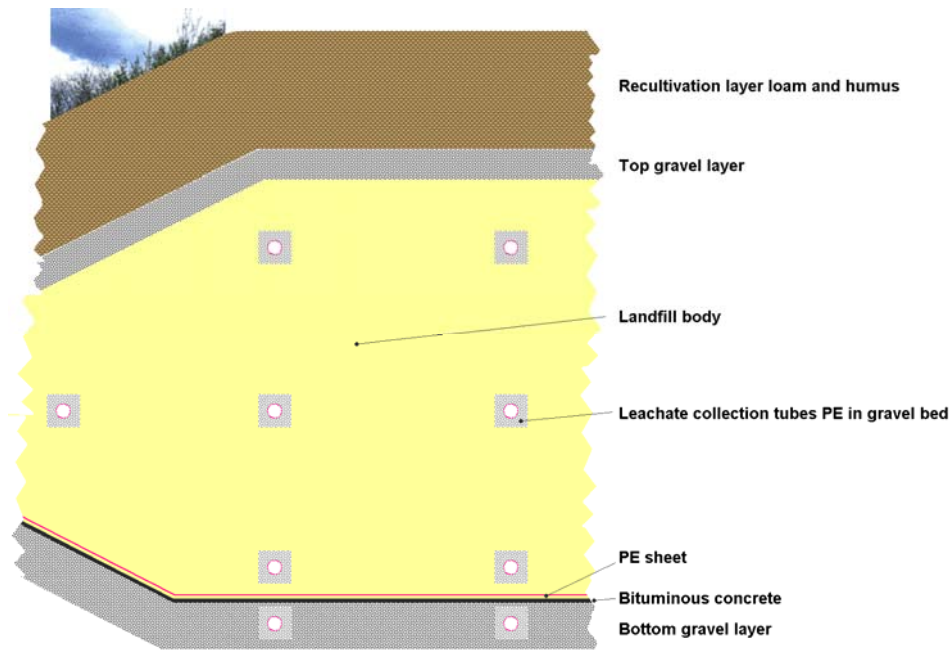


Fig. 6.4 Schematic section through a sanitary landfill. Not to scale

At the landfill bottom perforated PE tubes collect the leachate (radius 10 cm, thickness 2.4 cm, density 960 kg/m^3). A total grid length of 2000 m is assumed. An identical collection grid is located just below the base seal to detect seal breaks, so the tube material demand is doubled.

A further horizontal layer of an identical tubular collection grid of 2000 m length is installed every 5 m vertical distance within the landfill body. With a landfill depth of 20 m the total grid length of the drain system inside the landfill is 8000 m. The drain tubes are surrounded along their path with a 30 cm by 30 cm sized gravel bed (density 1600 kg/m^3).

Eight concrete collection tanks buffer the leachate and allow for measurements. The tanks have a rectangular 1 m by 2 m by 3 m geometry and a wall thickness of 15 cm. The leachate tank is connected to the local sewer system for subsequent wastewater treatment. A distance of 3 km is assumed. The concrete sewer pipe is assumed to have a radius of 25 cm and a thickness of 5 cm. The total concrete demand for the sewer pipe is 518 tons. This figure is complemented with proportional amounts of expenditures for a class 5 sewer pipe described in part IV 'Wastewater treatment' of this report⁷⁵, see Tab. 6.5.

Tab. 6.5 Expenditures for 3 km of a class 5 sewer pipe

unalloyed steel	15'688 kg	EPDM rubber	161 kg
high-alloyed steel	5'471 kg	sand	120'359 kg
cast iron	2'655 kg	water	3'343'895 kg
PVC	402 kg	earth excavation work	885 m ³
PE	7'321 kg	diesel	8'528 MJ
PP	402 kg	concrete	518'000 kg

⁷⁵ The expenditures concrete + gravel + cement for a class 5 sewer are taken to represent to the figure for concrete above. The amounts of the other expenditures are calculated in proportion to that sum.

Usually large landfills are charged in stages. Each partition must be sealed off from the others with watertight seals, like the outward border of the landfill. The construction of these walls is inventoried identically to the outward border (80 cm gravel, 7 cm bituminous concrete, 0.25 mm PE sheet, 0.5 l diesel per m² sealed surface). Again, the walls that are inclined in reality are assessed as being vertical. Four inside walls divide the landfill into eight partitions of 225'000 m³ volume. The total area for inside walls is 24'000 m².

To access the landfill, a road of 3000 m length and 6 m width is built specifically (Zimmermann et al. 1996:B.77). The total road surface is 18'000 m². The road features a 50 cm gravel bed (density 1600 kg/m³) and a 10 cm tarmac surface (15% bitumen, 85% gravel, density 2500 kg/m³). A diesel consumption of 0.2 litres per m² road surface (7.2 MJ/m²) is assumed for road construction and subsequent road renovation. The road is renovated every 50 years. The material is assumed to be recycled on site. The access road is provided during landfill construction (5 a), operation (30 a) and the whole period of aftercare (150 a). After use, the transformation of the road area is *not* inventoried in accordance with ecoinvent methodology.

After landfill closure the area covered with a final gravel layer of 50 cm (density 1600 kg/m³). For restoration a 4 m layer of humus and loam is applied. It is assumed that the restoration materials are excavation materials from other construction sites and the application as restoration material is recycling. Accordingly no production nor transport expenditures must be heeded for this material (except gravel). However the gravel and restoration materials with a total of 405'000 m² per landfill must be distributed with loaders and diggers (split 50:50).

All necessary materials are transported to the landfill site using the standard transport distances defined in the ecoinvent 2000 methodology report (Frischknecht et al. 2003a) except where noted. Gravel and concrete is transported 20 km by truck. Plastics, bitumen and rubber is transported 50 km by truck and 200 km by train. Metals are transported 50 km by truck and 600 km by train. Excavated material is transported 20 km by truck for re-use/recycling.

Tab. 6.6 Infrastructure for one complete Swiss sanitary landfill facility (capacity 1.8 million tons waste)

Name	Location	Unit	sanitary landfill facility	GSD ²
Location			CH	
InfrastructureProcess			1	
Unit			unit ¹	
excavation, hydraulic digger	RER	m ³	697'900	306.0%
excavation, skid-steer loader	RER	m ³	697'900	306.0%
diesel, burned in building machine	GLO	MJ	2'968'000	306.0%
gravel, round, at mine	CH	kg	288'500'000	306.0%
bitumen, at refinery	CH	kg	4'298'000	306.0%
concrete, sole plate and foundation, at plant	CH	m ³	257.2	306.0%
polyethylene, HDPE, granulate, at plant	RER	kg	512'200	306.0%
extrusion	CH	kg	181'800	306.0%
reinforcing steel, at plant	CH	kg	15'690	306.0%
chromium steel 18/8, at plant	RER	kg	5471	306.0%
cast iron, at plant	RER	kg	2655	306.0%
polyvinylchloride, at regional storage	RER	kg	402.3	306.0%
polypropylene, granulate, at plant	RER	kg	402.3	306.0%
synthetic rubber, at plant	RER	kg	160.9	306.0%
sand, at plant	CH	kg	120'400	306.0%
tap water, at user	RER	kg	3'344'000	306.0%
transport, lorry 28t	CH	tkm	35'730'000	315.1%
transport, freight, rail	RER	tkm	976'400	286.5%
Transformation, from pasture and meadow ²		m ²	18'000	207.0%
Transformation, to traffic area, road network ²		m ²	18'000	207.0%
Occupation, traffic area, road network ²		m ² a	3'330'000	158.0%

1 All infrastructure data modules in ecoinvent 2000 have one entire facility as functional unit.

2 Land use exchanges are for the access road only; not for the whole landfill. Landfill land use is inventoried in the process-specific exchanges.

The uncertainty in energy consumption (diesel) is set equal to the uncertainty in material demand. This although the Pedigree approach suggests smaller uncertainties for energy consumption (basic uncertainty of 105%). For energy demand in building activities it seems unlikely that the demand in materials has a basic uncertainty of 300%, but the energy used to process and handle those materials has only a uncertainty of 105%. Consequently, energy consumption for infrastructure has the same basic uncertainty as the materials (300%).

Tab. 6.7 Uncertainty of landfill infrastructure expenditures

Exchange	GSD ² value	Pedigree codes	Comment
landfill infrastructure material	306%	(2,3,3,1,1,5)	Basic uncertainty of 3
landfill land transformation	207%	(2,3,3,1,1,5)	basic uncertainty of 2
landfill land occupation	158%	(2,3,3,1,1,5)	basic uncertainty of 1.5

6.4 Operation of sanitary landfills

Within the ecoinvent 2000 methodology, the transport of the waste from the waste producer to the landfill site must be inventoried in the waste-generating process, and not in the waste disposal process.

Similarly, the municipal collection of waste is inventoried at the waste generating process⁷⁶. Municipal collection is inventoried in part II on incineration.

During the landfill operation, special loaders are used to distribute and compact the waste (cf. Fig. 5.1 on page 7). For that, an average consumption figure of 1.3 litre diesel per ton of waste (0.0467 MJ/kg waste) is applied from (Hunziker & Paterna 1995). Landfill gas is collected through the drain tubes within the landfill body. For the operation of the landfill gas pumps a value of 0.00135 kWh medium power grid electricity per kilogram of waste is applied (Hunziker & Paterna 1995). It is assumed that the leachate can flow by gravity to the wastewater treatment plant and no pumps are necessary. The energy demand of a simple administrative house is estimated at 0.000015 kWh low power grid electricity per kilogram of waste and 0.045 litre of heating oil per ton of waste (0.0016 MJ/kg waste) (Hunziker & Paterna 1995). The administrative energies needed equate to 3240 MJ electricity and 96'600 MJ fuel oil per year during a 30 year operation time. Uncertainty for heating fuel demand is assumed to be 120%, due to temperature variation (GSD).

Land use exchanges are based on the occupied landfill surface of 90'000 m². The original land type is assumed to be pasture and meadow (Corinair type 231). For five years the location is a construction site⁷⁷. Landfill operations last approximately 30 years, where the land is inventoried as sanitary landfill site (Corinair type 132b). After closure renaturation is promoted by planting of shrubs. For five years the site is assumed to be of type 'sclerophyllous shrub land' (Corinair type 323). After that transformation to forest land is assumed. The land occupation as forest land is attributed to forestry products (wood) and not to the landfill.

The inventoried process-specific burdens are displayed in Tab. A.10 on page 122 in the appendix.

Tab. 6.8 Uncertainty of landfill infrastructure expenditures

Exchange	GSD ² value	Pedigree codes	Comment
landfill energy demand	125%	(2,3,3,1,1,5)	Basic uncertainty of 1.05
landfill land transformation	207%	(2,3,3,1,1,5)	basic uncertainty of 2
landfill land occupation	158%	(2,3,3,1,1,5)	basic uncertainty of 1.5

6.5 Energy production of sanitary landfill

During the methane phase the landfill gas is collected and incinerated either in open flares, furnaces or in gas motors. Gas motors produce electricity. Heat from furnaces and waste heat from gas motors can be converted to useful heat. In 2000, Swiss landfills produced 44.25 GWh electricity from gas motors, 19.27 GWh heat from furnaces and 2.23 GWh heat from gas motors (BUWAL 2001h). With a total landfill gas input of 159.3 GWh/a, a thermal efficiency for the utilised landfill gas of 13.5% and an electric efficiency for utilised landfill gas of 27.8% results. This is the weighted efficiency for the converting motors and furnaces installed in Swiss landfills. The electric efficiency of the converter mix is larger than the thermal efficiency, as few landfills can put the heat into a district heating system.

However, not all landfills utilise the landfill gas for energy. About 34% of the landfills incinerate the captured landfill gas in open flares with no energy utilisation (Zimmermann et al. 1996:B.63). The *system mix* efficiency is therefore lower than the conversion efficiency above. In the system mix, 18.3% of the captured landfill gas is converted to electricity and 8.9% to useful heat.

⁷⁶ This is a deviation from former methodology applied in ETH studies like (Frischknecht et al. 1996, Zimmermann et al. 1996, Hellweg 2000, Doka 2002), where collection and transport was included within the system boundary of the waste disposal process.

⁷⁷ Land transformations to and from construction sites are not inventoried inecoinvent 2000, but land occupations are ().

The methane phase of the landfill lasts several decades (depending on landfill depth and climate). Typically gas production during the methane phase is stable for an intermediate time but then starts to decrease with a long fade-out phase. At some point, gas production becomes so small, that energy conversion becomes unfeasible. Also the gas collection system cannot guarantee a complete capture of all landfill gas generated due to seeps, i.e. there is a difference between the amount of microbiologically *generated* landfill gas and the amount that can actually be *captured*. Zimmermann et al. (1996:B.63) estimate that in Swiss landfills 47% of the generated landfill is emitted directly to the atmosphere and only 53% are recovered. Even with well-designed gas collection systems, few landfills are thought to capture more than 60% of generated landfill. Normal recovery rates are considered to be in a range of 40% to 50%⁷⁸ (Johannessen 1999).

The degradability rate used to characterise the decomposition, and hence gas production potential, of a specific waste encompasses a time span of 100 years. To calculate the energy generated from a specific waste, the overall energy conversion efficiency has to be reduced to take into account that part of the waste might indeed decompose during the methane phase, but that the generated gas might not be captured and not incinerated. Using the reported Swiss recovery rate of 53%, 9.7% of the *generated* landfill gas is converted to electricity and 4.7% to useful heat for the system mix over 100 years. I.e. the net system efficiency relating to landfill gas *generated* is 65% lower than the system efficiency relating only to the landfill gas *captured and utilised*.

The full burden of the landfill is allocated to the disposal function of the landfill. Generated energy is free of any burden. Substitution is not sensible within the context of the ecoinvent 2000 database.

6.5.1 Waste heat balance

The waste heat liberated from a specific waste depends on its (upper) heating value and its decomposability. A homogenous approach is taken here, i.e. the decomposition rate for the waste describes also the 'decomposition' or liberation of the heating value in the first 100 years (short-term). A small amount of the chemical energy of the waste is preserved in the directly emitted methane. Another part of the generated landfill gas is incinerated and the methane energy is released. This leads to a waste heat emission to air. A heat content of 40 MJ/kg carbon in incinerated landfill gas is assumed. The remainder of the waste heat is emitted to ground. A distinction is made between the short-term and the long-term release. A total heat release in the long-term is assumed as carbon and hydrogen, which are the main 'carriers' of chemical energy, will be completely degraded and emitted in the long-term. The calculation procedure for waste heat is shown schematically in Fig. 6.5.

⁷⁸ Johannessen includes losses from lateral movement of landfill gas, which is not possible in Swiss landfills with bituminous base and flank seals.

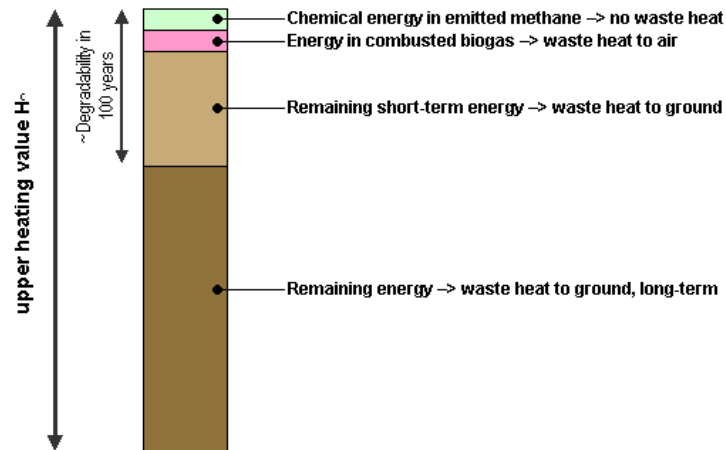


Fig. 6.5 Schematic of the waste-specific waste heat calculations for sanitary landfills.

6.6 Unit Process Inventories

The unit process inventories (i.e. non-cumulated data) of all waste materials to sanitary landfill is a very large table of 50 columns and 170 rows. A table representation in print is not very sensible, as many cells contain no data and some comment cells contain long text entries. The inventoried data is therefore contained in Excel tables on the CD-ROM. These workbooks contain the list of exchanges in ECOSPOLD format. Tab. 6.9 gives an overview of the file names and their contents.

Tab. 6.9 Names of files with sanitary landfill inventory data and their contents.

Excel file	Contents
13_MSRLF_LCI.xls	Waste disposal inventories
13_proc_landfills_LCI.xls	Process-specific burdens (for all landfill types)
13_infra_landfills_LCI.xls	Incinerator infrastructure (for all landfill types)

7 Life Cycle Inventory for slag compartments

7.1 Emissions from slag compartments

7.1.1 Transfer coefficients for slag compartments

Parameters

The transfer coefficients for slag compartments are calculated according to the procedure described in chapter 5.5.2 'Modelling of landfill emissions' on page 19. For these calculations, the elemental waste composition m and the initial leachate concentrations c_o must be known. Measurements for m and c_o are collected from literature sources (see Tab. A.2 and Tab. A.3 on page 112ff. in the appendix).

The average precipitation rate in the plateau of Switzerland (Mittelland) is approximately 1000 mm/m²a. Some of the precipitated water is evaporated or flows off over the surface. In a slag compartment approximately only 50% of the precipitation actually enters the landfill as infiltration water (Schweizer 1999:96, based on AGW 1991). The infiltration rate I for slag landfills is therefore 500 mm/m²a = 500 l/m²a. The preferential flow share in leachate output $w\%$ is 22%. Landfill height h is 15 m, slag density is 1500 kg/m³. These parameters are used to determine the effective annual leachate volume V_{eff} with chapter Eq. 5.20 on page 32. V_{eff} for the slag compartment is 0.0174 l per kg waste per year.

Using Eq. 5.15 on page 29, and V_{eff} from above, the time t_e when the carbonate phase ends is calculated to be 22'918a. This parameter is used in the calculation of the long-term transfer coefficients Eq. 5.13 and Eq. 5.14 on page 29.

Extrapolations to other elements

For some elements no literature data is available. Following assumptions for the short-term transfer coefficients are used. Bromine and iodine are given short-term transfer coefficients of 100%, as they are soluble halogens and form monovalent anions, like chlorine. There is little anion exchange capacity in the landfill. High solubility of iodine is also suggested by its use as a leachate tracer in landfill studies (Johnson et al. 1998). Silver is given the same short-term transfer coefficient as copper (0.0035%), based on similar chemistry. Additional information is derived from transfer coefficients for hard coal ash deposits reported in Frischknecht et al. (1996:VI.118). This data was compiled from (Tauber 1988, van der Sloot et al. 1982, Swaine & Goodarzi 1995). The geometric mean of the range for transfer coefficients of *alkaline* coal ashes were used with priority (Tab.VI.9.60 in Frischknecht et al. 1996:VI.118). Secondly, average, unspecified transfer coefficients for hard coal ash given in Tab.VI.9.61 were used; see Tab. A.9 on page 121 in the appendix. Beryllium, strontium and thallium are given the same short-term transfer coefficient as calculated for nickel (0.063%), based on information for hard coal ash, where these elements have identical transfer coefficients. Selenium and tungsten form oxianions (SeO_4^{2-} , WO_4^{2-}) and are soluble at high alkaline pH. High solubility is also suggested from hard coal ash. A short-term transfer coefficient of 1.4% derived from the mean of other known, oxianion-forming metals (Mo, Sb, V) is chosen. For scandium a value between potassium and phosphorus is chosen (6%), suggested from data for hard coal ash. To complete the mass balance, short-term transfer coefficients for oxygen and hydrogen are set equal to calcium.

All transfer coefficients of the slag compartment are given in chapter Tab. A.7 on page 119 in the appendix.

7.1.2 Calculation of slag compartment emissions

The short-term and long-term emissions are calculated by multiplying the waste composition (kg element/kg waste) with the transfer coefficients (kg emitted element/kg element content). See also section 'Uncertainty of transfer coefficients and emissions' on page 36.

7.1.3 Speciation of slag compartment emissions

On categorisation of landfill emissions see section Categorisation of landfill on page 36.

Chromium

Kersten et al. (1998) measured the total chromium and the Cr^{III} concentrations in 10 leachate samples from the Swiss bottom ash landfill Im Lorstorf AG. While for total chromium an average concentration of 0.21 ± 0.02 mmol/l was found, the Cr^{III} concentration was below the detection limit of 0.002 mmol/l. This suggests that <1% of the chromium in leachate is Cr^{III} and over 99% is Cr^{VI}. This is relevant since Cr^{VI} has higher toxicity than Cr^{III} ⁷⁹. Fruchter et al. (1990) also find Cr^{IV} instead of Cr^{III} in leachate of coal fly ash landfills (cited in Kersten et al. 1998). This suggests that the high Cr^{VI} speciation is not only a feature of bottom ash landfills/slag compartments, but also applies to fly ash and residual material landfills.

A rationalisation of this is that Cr^{VI} is soluble as chromate anion CrO₄²⁻. This oxianion is mobile as ashes have no notable anion exchange capacity. On the other hand 75% of chromium in slag is bound as Cr^{III} in chromite (FeCr₂O₄) which is very stable and not available even under most aggressive conditions (Huber et al. 1996:47). So the chromium which is actually mobile is bound to be transported as Cr^{VI} in CrO₄²⁻. In this study the short-term and long-term emissions of chromium to water from slag compartments will be inventoried as 100% Cr^{VI}.

Nitrogen, Sulfur and Phosphor

Emissions of nitrogen, sulfur, and phosphor are inventoried as nitrate NO₃⁻, sulfate SO₄²⁻, and phosphate PO₄³⁻, respectively, heeding the resulting mass increase from oxygen.

Carbon

Carbon emissions to water are inventoried *simultaneously* as TOC, DOC, BOD, and COD in the ecoinvent database (Frischknecht et al. 2003a). Carbon emissions in leachate are interpreted as total organic carbon TOC. It is safe to assume that all this carbon is dissolved. Any particulate carbon would precipitate during the long residence times in the landfill. Hence the dissolved organic carbon DOC is set equal to the TOC value. Average BOD-to-DOC and COD-to-DOC ratios derived from literature (Tab. 7.1) are then used to calculate BOD and COD values from the DOC value.

Tab. 7.1 Leachate data for hydrocarbons in slag landfill leachate and average ratios

		AIB 1993	Lechner 2001	Lechner 2001	BLU 1983	BLU 1983	
	unit	mean	Min	Max	min	max	Geomean
BOD	mg O ₂ /l	137	0.5	1700	0.5	350	28.96
COD	mg O ₂ /l		14	560			88.54
DOC	mg C/l	30	10	400	10 ¹	44 ¹	35.04
BOD/DOC ratio	–						0.827
COD/DOC ratio	–						2.527
BOD/COD ratio	–						0.327

1 given as TOC, assumed to be equal to DOC

⁷⁹ The former LCI landfill models of ETH Zürich (e.g. Zimmermann et al. 1996, Hellweg 2000) assumed a release as the more common Cr^{III}. The higher toxicity of Cr^{VI} has but moderate effects in LCIA. In CML'01 (Guinée et al. 2001) the factors for Cr^{VI} are a factor 1 – 4 larger than for Cr^{III}. In Eco-indicator'99 a factor of 1664 for emissions of water results, mainly the consequence of the carcinogenicity score (human health) for Cr^{VI} but none for Cr^{III}.

For all other emissions, no speciation is necessary. The inventory lists them as chemical elements and not compounds.

7.2 Infrastructure of slag compartment

The slag compartment is not an individual landfill, but a part of a larger landfill, usually a sanitary landfill. The infrastructure of the slag compartment is therefore inventoried based on the information given in chapter 7.2 'Infrastructure' on page 67. The infrastructure inventory of the slag compartment heeds the smaller size parameters of the slag compartment. The following text mentions only the geometrical or constructional differences to the sanitary landfill infrastructure.

The compartment shape is approximated with a rectangular box. The depth is assumed to be 15 m and the area 25'000 m². The compartment volume is thus 375'000 m³. The average density of the waste is 1500 kg/m³. The compartment capacity is thus 562'500 tons of waste.

The slag compartment is excavated and sealed at base and flanks in the same manner as the sanitary landfill. The sealed area is 34'500 m². The infrastructure inventory of the slag compartment heeds the different size parameters of the slag compartment. The two base drain tubes are only 1000 m long each and an additional 3000 m drain tubes inside the compartment body are inventoried. All material surface or thickness and specific diesel consumption figures are identical to the sanitary landfill. The compartment must be sealed off against the rest of the landfill. The flank seals included above are assumed to cover that expenditure. No inside walls are inventoried, as the compartment is small enough.

Since the slag compartment is part of a larger sanitary landfill, certain expenditures are shared with the rest of the landfill. Only a part of some expenditures must be allocated to the slag compartment. These expenditures are the access road, the leachate storage tanks and the sewer pipe (and administrative energy use, cf. below). As an allocation key the relative volumes of MSW landfill and slag compartment are taken⁸⁰. The modelled MSW landfill has a total volume of 1'800'000 m³, of that volume the slag compartment occupies 375'000 m³ or 21%. So instead of 3000 m road, 8 concrete storage tanks and 3000 m of sewer as in the MSW landfill, only 21% of that (625 m road, 1.67 storage tanks and 625 m sewer) are allocated to the slag landfill. All construction details are identical to what is inventoried for the sanitary landfill. Uncertainty is derived from Tab. 6.7 on page 61.

The road is provided during landfill construction (5 a), operation (30 a) and the whole period of aftercare (75 a). The road is renovated every 50 years. The allocated road surface is 3750 m². After use, the transformation of the road area is *not* inventoried in accordance with ecoinvent methodology.

⁸⁰ The reasoning to take volume and not mass is that landfills provide space for waste. Slag is a denser waste than MSW and uses up less space. With the allocation key 'volume' this advantage of slag is respected. However the overall effect of this choice is small as the infrastructure expenditures of landfills are usually not ecologically dominant.

Tab. 7.2 Infrastructure for one slag compartment (capacity 562'500 tons of waste)

Name	Location	Unit	slag compartment	GSD ²
Location			CH	
InfrastructureProcess			1	
Unit			unit ¹	
excavation, hydraulic digger	RER	m ³	162'600	306.0%
excavation, skid-steer loader	RER	m ³	162'600	306.0%
diesel, burned in building machine	GLO	MJ	681'000	306.0%
gravel, round, at mine	CH	kg	74'370'000	306.0%
bitumen, at refinery	CH	kg	1'046'000	306.0%
concrete, sole plate and foundation, at plant	CH	m ³	53.59	306.0%
polyethylene, HDPE, granulate, at plant	RER	kg	243'500	306.0%
extrusion	CH	kg	160'900	306.0%
reinforcing steel, at plant	CH	kg	3268	306.0%
chromium steel 18/8, at plant	RER	kg	1140	306.0%
cast iron, at plant	RER	kg	553.1	306.0%
polyvinylchloride, at regional storage	RER	kg	83.81	306.0%
polypropylene, granulate, at plant	RER	kg	83.81	306.0%
synthetic rubber, at plant	RER	kg	33.52	306.0%
sand, at plant	CH	kg	25'070	306.0%
tap water, at user	RER	kg	696'600	306.0%
transport, lorry 28t	CH	tkm	7'930'000	306.0%
transport, freight, rail	RER	tkm	260'900	270.9%
Transformation, from pasture and meadow ²		m ²	3750	207.0%
Transformation, to traffic area, road network ²		m ²	3750	207.0%
Occupation, traffic area, road network ²		m ² a	412'500	158.0%

1 All infrastructure data modules in ecoinvent 2000 have one entire facility as functional unit.

2 and use exchanges are for the access road only; not for the whole landfill. Landfill land use is inventoried in the process-specific exchanges.

7.3 Operation of slag compartment

During the landfill operation, loaders are used to distribute the slag. Unlike for MSW, compaction is less an issue for slag. The diesel consumption is adopted from the figure for residual material landfill waste distribution. An average consumption figure of 0.75 litre diesel per ton of waste (0.027 MJ/kg waste) is inventoried. No active gas collection with pumps is performed and no electricity is needed for that. The administrative building is shared with the whole landfill. The administrative demand is assumed to be mainly connected with waste volume (truckloads). The original energy demand figures of the sanitary landfill of 0.000054 MJ electricity and 0.0016 MJ fuel oil per kilogram municipal waste are decreased, since slag is more dense than municipal waste (1500 kg/m³ instead of 1000 kg/m³). Per ton of slag 0.000036 MJ low power grid electricity and 0.00107 MJ heating oil per kilogram slag are inventoried for the administrative energy demand of the slag compartment. All these energies are assigned unspecifically to each kilogram of landfilled slag.

Land use exchanges are based on the occupied landfill surface of 25'000 m². The original land type is assumed to be pasture and meadow (Corinair type 231). For five years the location is a construction site⁸¹. Landfill operations last approximately 30 years, where the land is inventoried as sanitary landfill site (Corinair type 132b). After closure renaturation is promoted by planting of shrubs. For five years the site is assumed to be of type 'sclerophyllous shrub land' (Corinair type 323). After that

⁸¹ Land transformations to and from construction sites are not inventoried in ecoinvent 2000, but land occupations are ().

transformation to forest land is assumed. The land occupation as forest land is attributed to forestry products (wood) and not to the landfill.

The inventoried process-specific burdens are displayed in Tab. A.10 on page 122 in the appendix.

7.4 Unit process inventories

Waste to slag compartment is only generated in municipal incineration plants MSWI. The landfilling of bottom ash from MSWI is integrated in the inventories of municipal incineration, which are explained in part II and given in the Excel table '13_MSWI_LCI.xls' on the CD-ROM. Tab. 7.3 gives the file names for tables of the inventoried process-specific and infrastructure burdens.

Tab. 7.3 Names of files with slag compartment inventory data and their contents.

Excel file	Contents
13_proc_landfills_LCI.xls	Process-specific burdens (for all landfill types)
13_infra_landfills_LCI.xls	Incinerator infrastructure (for all landfill types)

8 Life Cycle Inventory for residual material landfills

8.1 Emissions from residual material landfill

8.1.1 Transfer coefficients for residual material landfill

Parameters

The transfer coefficients for residual material landfills are calculated according to the procedure described in chapter 5.5.2 'Modelling of landfill emissions' on page 19. For these calculations, the elemental waste composition m and the initial leachate concentrations c_o must be known. Measurements for m and c_o are collected from literature sources (see Tab. A.4 on page 116 in the appendix).

The average precipitation rate in the plateau of Switzerland (Mittelland) is approximately $1000 \text{ mm/m}^2\text{a}$. Some of the precipitated water is evaporated or flows off over the surface. In a residual material landfill approximately only 20% of the precipitation actually enters the landfill as infiltration water (Schweizer 1999:96, based on AGW 1991). The infiltration rate I for residual material landfills is therefore $200 \text{ mm/m}^2\text{a} = 200 \text{ l/m}^2\text{a}$. The preferential flow share in leachate output $w\%$ is 22%. Landfill height h is 10 m, waste density δ is 1600 kg/m^3 . These parameters are used to determine the effective annual leachate volume V_{eff} with Eq. 5.20 on page 32. V_{eff} for the residual material landfill is 0.0098 l per kg waste per year.

Using Eq. 5.15 on page 29, and V_{eff} from above, the time t_e when the carbonate phase ends is calculated to be over 660'000a, i.e. in presumably 60'000a, when the next glacial period occurs and destroys the landfill, the landfill is still in its carbonate phase. As explained in section 'Long-term emissions' on page 25, the minimal values of the transfer coefficients will be set equal to the mean values of the transfer coefficients (which define the emissions that occur up to the next glacial period).

Extrapolations to other elements

For some elements no literature data is available. Following assumptions for the short-term transfer coefficients are used. Iodine is given a short-term transfer coefficient of 100%, as it is a soluble halogen and forms a monovalent anion, like chlorine and bromine. High solubility is also suggested by the use of iodine as a tracer in (Johnson et al. 1998). Silver is given the same short-term transfer coefficient as copper (0.0064%), based on their similar chemistry. Titanium is given the same short-term transfer coefficient as aluminium (0.050%). Likewise, barium is given the same transfer coefficient as manganese (0.0014%), based on similarity in leaching behaviour observed from slag compartment data. For nitrogen, boron, and iron the transfer coefficients from slag landfill are used. Additional information is derived from transfer coefficients for hard coal ash deposits reported in Frischknecht et al. (1996:VI.118). This data was compiled from (Tauber 1988, van der Sloot et al. 1982, Swaine & Goodarzi 1995). The geometric mean of the range for transfer coefficients of *alkaline* coal ashes were used with priority (Tab.VI.9.60 in Frischknecht et al. 1996:VI.118). Secondly, average, unspecified transfer coefficients for hard coal ash given in Tab.VI.9.61 were used; see Tab. A.9 on page 121 in the appendix. Beryllium, strontium and thallium are given the same short-term transfer coefficient as calculated for nickel (0.0604%), since in the hard coal ash data these elements have identical transfer coefficients. Selenium, antimony, and tungsten form oxianions (SeO_4^{2-} , SbO_4^{3-} , WO_4^{2-}) and are soluble at high alkaline pH. High solubility is also suggested from hard coal ash data. An arithmetic mean value based on other calculated oxianions is used (35%). For scandium a value between potassium and phosphorus is chosen (14%), based on data from hard coal ash. To complete the mass balance, short-term transfer coefficients for oxygen and hydrogen were set equal to calcium.

All transfer coefficients of the residual material landfill are given in Tab. A.8 on page 120 in the appendix.

8.1.2 Calculation of residual material landfill emissions

The short-term and long-term emissions are calculated by multiplying the specified waste composition (kg element/kg waste) with the transfer coefficients (kg emitted element/kg element content). See also section 'Uncertainty of transfer coefficients and emissions' on page 36.

8.1.3 Speciation of residual material landfill emissions

On categorisation of landfill emissions see section Categorisation of landfill on page 36.

The same speciation as outlined in chapter 7.1.3 'Speciation of slag compartment emissions' on page 66 are used for the residual material landfill emissions. In short, chromium is inventoried as Cr^{VI}, nitrogen, sulfur, and phosphor are inventoried as nitrate NO₃⁻, sulfate SO₄²⁻, and phosphate PO₄³⁻.

No information on the average speciation of the organic carbon emissions (TOC, DOC, BOD, COD) is available. The same distribution as for slag compartments is assumed (see chapter 7.1.3 'Speciation of slag compartment emissions' on page 66). For all other emissions, no speciation is necessary. The inventory lists them as chemical elements and not compounds.

8.2 Infrastructure of residual material landfill

Residual material landfills must have essentially the same constructional details as sanitary landfills (TVA 2000). The infrastructure of the residual material landfill is therefore inventoried based on the information given in chapter 7.2 'Infrastructure' on page 67. The infrastructure inventory of the residual material landfill heeds the different size parameters of the residual material landfill. The following text mentions only the constructional differences to the sanitary landfill infrastructure.

The landfill shape is approximated with a rectangular box. The depth is assumed to be 10 m and the area 30'000 m². The landfill volume is thus 300'000 m³. The average density of the waste is 1600 kg/m³. The landfill capacity is thus 480'000 tons of residual material⁸².

The residual material landfill is excavated and sealed at the base and the flanks in the same manner as the sanitary landfill. The two base drain tubes are only 1000 m long each. Four leachate collection tanks are built. No sewer connection is necessary. No additional drain tubes inside the landfill body nor compartment walls are inventoried. An access road of 3000 m length is inventoried. The road is provided during landfill construction (5 a), operation (30 a) and the whole period of aftercare (40 a) and renovated every 50 years. After use, the transformation of the road area is *not* inventoried in accordance with ecoinvent methodology. All material surface or thickness figures and specific diesel consumption figures are identical to the sanitary landfill. Uncertainty is derived from Tab. 6.7 on page 61.

⁸² Swiss legislation prescribes a minimal size for new residual material landfills of 100'000 m³ (TVA 2000:Art.31).

Tab. 8.1 Infrastructure for one complete Swiss residual material landfill facility (capacity 480'000 tons)

Name	Location	Unit	residual material landfill facility	GSD ²
Location			CH	
InfrastructureProcess			1	
Unit			unit ¹	
excavation, hydraulic digger	RER	m ³	157'500	306.0%
excavation, skid-steer loader	RER	m ³	157'500	306.0%
diesel, burned in building machine	GLO	MJ	858'000	306.0%
gravel, round, at mine	CH	kg	94'990'000	306.0%
bitumen, at refinery	CH	kg	1'644'000	306.0%
concrete, sole plate and foundation, at plant	CH	m ³	10.8	306.0%
polyethylene, HDPE, granulate, at plant	RER	kg	117'600	306.0%
extrusion	CH	kg	28'950	306.0%
transport, lorry 28t	CH	tkm	7'388'000	282.3%
transport, freight, rail	RER	tkm	352'400	257.7%
Transformation, from pasture and meadow ²		m ²	18'000	207.0%
Transformation, to traffic area, road network ²		m ²	18'000	207.0%
Occupation, traffic area, road network ²		m ² a	1'350'000	158.0%

1 All infrastructure data modules in ecoinvent 2000 have one entire facility as functional unit.

2 Land use exchanges are for the access road only; not for the whole landfill. Landfill land use is inventoried in the process-specific exchanges.

8.3 Operation of residual material landfill

8.3.1 Cement consumption for solidification

In Swiss residual material landfills, the waste is often solidified with cement. For MSWI residues this happens at the incinerator plant or at the landfill site. The LCI calculations for MSWI residues already contain the expenditures for cement solidification (cf. part II 'Waste Incineration').

For special waste, usually industrial waste, that is *directly* landfilled in residual landfills, additional solidification may be necessary. An example of such a waste is the inorganic sludge from brine filtration in chloralkali electrolysis. The user can specify in the calculation tool, if additional solidification of the waste is necessary. If waste is solidified, cement and water is added in the proportion waste–cement–water of 50%–20%–30%. Hence, for one kilogram of unsolidified waste (= functional unit of the data module) 2 kilograms of material are landfilled. The assumed cement composition, including traces, is given in part II on waste incineration.

In the residual landfill model, the inventoried emissions are based on waste composition and the *average* elemental transfer coefficients. No *waste-specific* adaptation of transfer coefficients was performed and thus also solidification with cement results in *no change of the inventoried emissions*. This is a simplification, as at least in the short term the degradation and disintegration of the waste can be slowed down with solidification. Solidification is then a means to postpone emissions from the short term into the long term. As the technical barrier functions of landfills have limited lifetime (cf. Future emissions in LCA on page 14), the abating effect of solidification will cease to be effective in the long run. In the long term emissions will be determined by the average conditions encountered in the landfill. As the ecoinvent project proposes to include long-term emissions in LCA results with equal weight as short-term emissions (Frischknecht et al. 2003a:10), the effect of solidification on emissions within this framework is minor. It has to be reminded that these models aim to capture *typical* magnitude of burdens from landfilling to complement LCI data from production processes.

They should not be used to assess disposal-specific questions, like if solidification is a sensible thing to do or not.

8.3.2 Energy demand for landfill operations

During the landfill operation, loaders are used to place the solidified residual material. Unlike for MSW in sanitary landfills, compaction of the waste is less an issue here. An average consumption figure of 0.75 litre diesel per ton of waste (0.027 MJ/kg waste) is inventoried (Unreferenced value in Zimmermann et al. 1996:B.169). No gas collection occurs and no electricity is needed for that. A similar administrative building as for the sanitary landfill is inventoried. But its energy demand is distributed to a smaller total mass of waste in the landfill. The 3240 MJ electricity and 96'600 MJ fuel oil per year during the 30 year operation time equate to 0.0002 MJ electricity and 0.006 MJ fuel oil per kilogram waste.

8.3.3 Land use exchanges

Land use exchanges are based on the occupied landfill surface of 30'000 m². The original land type is assumed to be pasture and meadow (Corinair type 231). For five years the location is a construction site⁸³. Landfill operations last approximately 30 years, where the land is inventoried as sanitary landfill site (Corinair type 132b). After closure renaturation is promoted by planting of shrubs. For five years the site is assumed to be of type 'sclerophyllous shrub land' (Corinair type 323). After that transformation to forest land is assumed. The land occupation as forest land is attributed to forestry products (wood) and not to the landfill.

8.3.4 Summary process-specific burdens

The inventoried process-specific burdens (energy demand and land use) are displayed in Tab. A.10 on page 122 in the appendix. Cement consumption for solidification is a waste-specific burden and not contained in this module.

8.4 Unit process inventories

The unit process inventories (i.e. non-cumulated data) of all waste materials to residual landfill is a very large table of 170 columns and 130 rows. A table representation in print is not very sensible, as many cells contain no data and some comment cells contain long text entries. The inventoried data for waste disposal processes is therefore contained in Excel tables on the CD-ROM. These workbooks contain the list of exchanges in ECOSPOLD format. Tab. 8.2 gives an overview of the file names and their contents.

Tab. 8.2 Names of files with residual landfill inventory data and their contents.

Excel file	Contents
13_RMLF_LCI.xls	Waste disposal inventories
13_proc_landfills_LCI.xls	Process-specific burdens (for all landfill types)
13_infra_landfills_LCI.xls	Incinerator infrastructure (for all landfill types)

⁸³ Land transformations to and from construction sites are not inventoried in ecoinvent 2000, but land occupations are ().

9 Life Cycle Inventory for inert material landfills

9.1 Emissions from inert material landfills

The waste material placed in inert material landfills has generally a low pollutant content and is chemically inert to a large extent. In previous ETH inventories no leachate emissions were inventoried for this reason (Frischknecht et al. 1996:F.27). For the inert material landfill no direct leachate emissions will be used.

This practice is not strictly consistent with the current landfill inventories, as emissions from landfills are inventoried irrespective of their concentration. I.e. also for inert material landfills the behaviour of the waste should be modelled waste-specifically and transfer coefficients should be established. In the future a specific model for inert material landfills could be developed to account for the leachate emissions. In the meantime, for sensitivity analysis, the process module 'hydrated cement to residual material landfill' could be used to estimate the relevance of emissions from such 'inert materials'. The average trace contents in hydrated cement are heeded in that module. Residual material landfill development predicts an extended carbonate phase of >60'000 years; i.e. no drop of pH in the modelled time period. This behaviour is also likely for inert material landfills. So the chemical behaviour of residual material landfills is probably comparable that of inert material landfills⁸⁴. A comparison of 'inert' material in residual material landfills and the current inert material landfill model is made in the result section of part V on building materials.

9.2 Infrastructure of inert material landfill

Inert material landfills usually have essentially the same constructional details as sanitary landfills. They can however waive the base seal and leachate collection system, if located outside drinking water zones (TVA 2000). It is not known which percentage of the over 194 inert material landfills in Switzerland do have a base seal and a leachate collection system. In the former ETH LCI database, inert material landfills were inventoried without a base seal or collection system (Frischknecht et al. 1996:F.27). For this report, it is assumed that 50% of the IMLF do have a base seal and a leachate collection system, and 50% do not. The inventoried infrastructure is a 50:50 system mixture of the two types.

The infrastructure of the inert material landfill with base seal and collection system is inventoried based on the information given in chapter 7.2 'Infrastructure' on page 67. The infrastructure inventory of the inert material landfill heeds the different size parameters of the residual material landfill and resembles the residual material landfill. The following text mentions only the constructional differences to the sanitary landfill infrastructure.

The landfill shape is approximated with a rectangular box. The depth is assumed to be 15 m and the area 30'000 m². The landfill volume is thus 450'000 m³. The average density of the waste is 1500 kg/m³. The landfill capacity is thus 675'000 tons of residual material⁸⁵. Per kilogram waste $1.48 \cdot 10^{-9}$ units of landfill infrastructure are needed.

The inert material landfill is excavated in the same manner as the sanitary landfill. A share of 50% of the infrastructure mix have following features:

- a sealed base and flanks (40'400 m²).

⁸⁴ This statement relates to the chemical behaviour of elements in the landfill, as expressed by transfer coefficients. Of course the pollutant content per kilogram waste is higher in residual landfills.

⁸⁵ Swiss legislation prescribes a minimal size for new inert material landfills of 100'000 m³ (TVA 2000:Art.31).

- two base drain tubes 1000 m long each.
- Four leachate collection tanks are built.

No additional drain tubes inside the landfill body nor compartment walls are inventoried. No sewer connection is necessary. An access road of 3000 m length is inventoried. The road is provided during landfill construction (1 a), operation (10 a) and a period of aftercare (5 a). After use, the transformation of the road area is *not* inventoried in accordance with ecoinvent methodology.

All material thickness and specific diesel consumption figures are identical to the sanitary landfill. Uncertainty is derived from Tab. 6.7 on page 61.

Tab. 9.1 Infrastructure for one complete Swiss inert material landfill facility (capacity 675'000 tons)

Name	Location	Unit	inert material landfill facility	GSD ²
Location			CH	
InfrastructureProcess			1	
Unit			unit ¹	
excavation, hydraulic digger	RER	m ³	195'000	306.0%
excavation, skid-steer loader	RER	m ³	195'000	306.0%
diesel, burned in building machine	GLO	MJ	404'500	306.0%
gravel, round, at mine	CH	kg	71'080'000	306.0%
bitumen, at refinery	CH	kg	1'205'000	306.0%
concrete, sole plate and foundation, at plant	CH	m ³	5.4	306.0%
polyethylene, HDPE, granulate, at plant	RER	kg	62'950	306.0%
extrusion	CH	kg	14'480	306.0%
transport, lorry 28t	CH	tkm	9'135'000	316.4%
transport, freight, rail	RER	tkm	253'600	259.5%
Transformation, from pasture and meadow ²		m ²	18'000	306.0%
Transformation, to traffic area, road network ²		m ²	18'000	306.0%
Occupation, traffic area, road network ²		m ² a	288'000	306.0%

1 All infrastructure data modules in ecoinvent 2000 have one entire facility as functional unit.

2 Land use exchanges are for the access road only; not for the whole landfill. Landfill land use is inventoried in the process-specific exchanges.

9.3 Operation of inert material landfill

During the landfill operation, loaders are used to place the landfilled material. Unlike for MSW in sanitary landfills, compaction of the waste is less an issue here. An average consumption figure of 0.75 litre diesel per ton of waste (0.027 MJ/kg waste) is adapted from residual material landfills (Unreferenced value in Zimmermann et al. 1996:B.169). No gas collection occurs and no electricity is needed for gas pumps. A similar administrative building as for the sanitary landfill is inventoried. But its energy demand is distributed over a smaller total mass of waste in the landfill. The 3240 MJ electricity and 96'600 MJ fuel oil per year during the 10 year operation time equate to 0.00005 MJ electricity and 0.0014 MJ fuel oil per kilogram waste.

Land use exchanges are based on the occupied landfill surface of 30'000 m². The original land type is assumed to be pasture and meadow (Corinair type 231). For one year the location is a construction site⁸⁶. Landfill operations last approximately 10 years, where the land is inventoried as sanitary landfill site (Corinair type 132b). After closure renaturation is promoted by planting of shrubs. For

⁸⁶ Land transformations to and from construction sites are not inventoried in ecoinvent 2000, but land occupations are ().

five years the site is assumed to be of type 'sclerophyllous shrub land' (Corinair type 323). After that transformation to forest land is assumed. The land occupation as forest land is attributed to forestry products (wood) and not to the landfill.

The inventoried process-specific burdens (energy demand and land use) are displayed in Tab. A.10 on page 122 in the appendix.

9.4 Inventoried burdens per kilogram waste

Tab. 9.2 shows the inventoried burdens per kilogram waste in the Ecospol format. All burdens are equal for all waste modules. The last column give the 'StandardDeviation95%' (=GSD²). The uncertainty is heeded in the requested modules.

Tab. 9.2 Inventoried burdens per kilogram waste in the Ecospol format.

Name Location InfrastructureProcess Unit	Location	Category	SubCategory	InfrastructureProcess	Unit	disposal, , to inert material landfill CH 0 kg	StandardDeviation95%
inert material landfill facility	CH	waste management	inert material landfill	1	unit	1.48148E-09	100%
process-specific burdens, inert material landfill	CH	waste management	inert material landfill	0	kg	1	100%

10 Life Cycle Inventory for Underground Deposits

10.1 Emissions from underground deposits

Underground deposits for hazardous waste are located in salt mines. The operators of German deposits emphasize the fact, that these salt veins have lasted over 240 million years without being dissolved or destabilised. Clay layers seal off the mine from rainwater intrusion and salt wash-off over geological times. In Herfa-Neurode, the example of an intrusion of volcanic basalt 20 Million years ago asserts the long-term stability of the salt vein (Brendel 2000). The absence of any free-flowing water secures the stability of the waste storage site and guarantees the immobility of waste materials.

Rock salt is less dense than other rocks. This exerts a buoyancy force on the salt and it will move upwards towards the surface ('diapirism'). This process shapes the salt layers into 'domes' of salt (German 'Salzstock') in the course of millions of years. The rock salt is also ductile and exhibits flow. Cracks or openings in the salt therefore can 'heal' off ('halokinetic mobility'). The deposited waste is thought to be completely enclosed by salt after several hundred years. This is another safety feature of salt mine deposits.

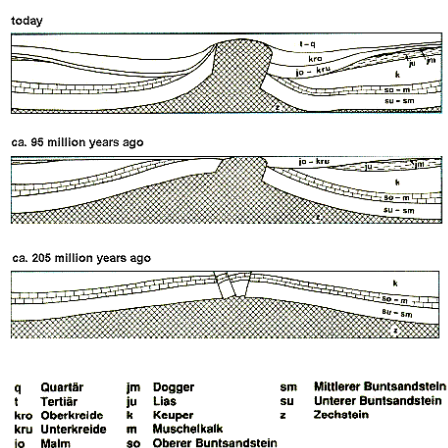


Fig. 10.1 Development of the salt dome Gorleben, Germany, over geological time frames. Adapted from (GNS 2002)

Risk of water intrusion and mine flooding

The principal safety feature of underground salt mine deposits is the absence of free water, which prevents pollutants to be washed-off and dispersed, even if containers were corroded.

However, if water were to enter the salt mine several detrimental processes would ensue. Water intrusion would pose the risk that pollutants could be washed to the biosphere. Also, water represents a reactand media. The salt water would corrode steel containers much faster. Chemicals from leaking containers could react with the water or with each other. If such reactions are exothermic – generating heat – more water could be liberated from crystalline-bound water⁸⁷. Thermal gradients also induce transport of water *towards the heat pool*, so called 'thermomigration' (Grimmel 1993:41). If gases were formed by reactions of the waste, the gas-impermeability of rock salt could lead to pressure buildup and fissures. Water intrusion would also compromise the mechanical stability of the mine by dissolution of rock salt.

⁸⁷ For example carnallit $\text{KCl} \cdot \text{MgCl}_2 \cdot 6\text{H}_2\text{O}$ or kieserit ($\text{MgSO}_4 \cdot \text{H}_2\text{O}$), that are encountered in salt veins, contain crystalline-bound water.

Water intrusion in salt mines is not merely a remote, theoretical risk. For example, the commercial Retsof salt mine in the Genesee Valley, USA, was destroyed 1994 after unexpected water intrusion from the surface and a complete flooding of the mine. This also led to collapses on the surface (Fig. 10.2). Retsof was up to 1994 still producing salt before it was destroyed and was one of the largest salt mines in the world (Tepper et al. 1997, Kappel et al. 1998).



Fig. 10.2 Surface damages from the 1994 collapse of the Retsof salt mine, Genesee Valley, USA (Kappel et al. 1998)

A frequent fate of old salt mines is complete *flooding with groundwater entering from the surface* within years (Jockel 2001). Consequently, special measures must be taken in hazardous waste deposits to prevent water entering the mine after closure. In Herfa-Neurode, for this reason a 48 m thick, 4645 m³ brick wall⁸⁸ was erected within the mine to seal off the waste deposit from the conventional part of the Herfa potash mine (Brendel 2000). See Fig. 10.3. A problematic weak point in the mine is the vertical access shaft. It is not definite, how this opening will be sealed off watertight from the surface after closure. Possibly by a clay seal at the surface will be used (Brendel 2000).

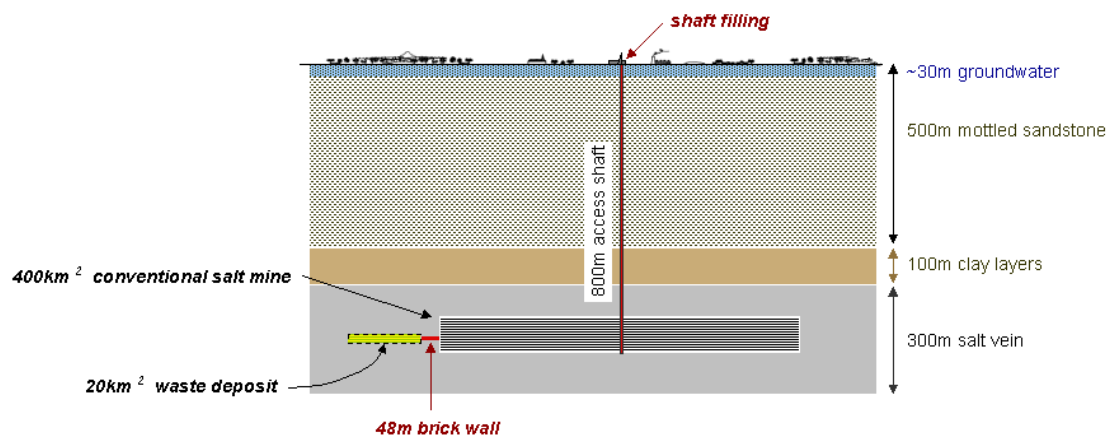


Fig. 10.3 Schematic situation of the Herfa-Neurode waste deposit and post-closure safety measures according to information by the operator Kali+Salz GmbH (Brendel 2000). Correct relative scales

In the experimental radioactive salt mine repository⁸⁹ Asse II near Hannover, Germany, inexplicable, permanent and discontinuously increasing water intrusion was observed since 1991. The water is expected to fill the remaining cavities with water and the whole salt mine is expected to be filled with

⁸⁸ This is *not* the compartment brick wall depicted in Fig. 5.5 on page 12, which is much smaller.

⁸⁹ Radioactive repositories in salt mines are *not* within the scope of this report. However the presumed inherent safety features of salt mines (absence of water) apply to underground deposits for hazardous waste as well as to radioactive repositories.

water within approximately 120 years. Transport of radionuclides to the biosphere cannot be excluded⁹⁰. Also the salt mine repository for low and medium radioactive wastes in Morsleben is in danger of collapsing from water intrusion and was commissioned in 1999 to be closed down (Musiol 2001).

Leuenberger (1999) argues that it might be misleading to extrapolate the past million-year-long stability of the salt mine into the future. The stability of the mine has been compromised by human mining and blasting activities. Indeed, pressure measurements in salt pillars in the mine show that a new equilibrium has not been reached yet. Movements in the salt are known and expected (cf. 'halokinetic mobility' mentioned above). The current mine system with shafts and tunnels is mechanically not identical to the original state and might well be damaged by earthquakes or basalt intrusions, that in earlier instances showed no effect on stability.

Evidence for endangered mine stability

That the stability of geological formations can indeed be compromised by mining can be illustrated by the records of seismic events. Seismic records are available for Germany from Leydecker (2001a) covering 12 centuries. The Werra potash mining district south of Göttingen, where Herfa-Neurode is located, has a small natural seismic activity. Since 800 A.D. the few natural earthquakes had a maximal local intensity of 2, which is comparatively small⁹¹. This can be regarded as a natural local advantage regarding the long-term stability of the Herfa waste deposit. In the 20th century however 5 rockburst events took place in the Werra district with local intensities of 3.7 to 5.6. In 1953 such a rockburst event resulted in human injuries and cracks in the surface (location Heiringen). Rockbursts are *not* the seismic waves from mine blasting operations but collapses of mines, triggered by weak carnallitic columns⁹². Rockbursts in the Werra district are indeed some of the largest recorded rockburst events in the world (Leydecker 2001b). The maximum rock burst happened on March 13, 1989 in Völkershausen with a sudden collapsed mine area of 6.8 km², resulting in an event of local intensity 5.6 and a felt area of 140 km in radius. In September 11, 1996 a former potash mine collapsed in Halle/Saale (Germany).

Hence, the risk of seismic damages to salt mines seems larger from *induced effects of mining activities* than from natural seismicity. And indeed this supports the doubt expressed in (Leuenberger 1999) that it might be wrong to extrapolate the past million-year-long stability of the salt mine into the future.

10.1.1 Conclusions for the inventory

It can be concluded that waste deposits in salt mines have intrinsic properties that make them exceptional for deposition of hazardous waste – most notably the absence of free water. Also the ordered and recorded deposition of waste makes future retrieval possible (this is in contrast to the 'filling-up' of old coal mines (back-filling, German '*Bergversatz*'), partly with hazardous wastes, but also in contrast to other types of common surface landfills).

Observations from other salt mines and the safety and research efforts of deposit operators, however, cast doubt on the notion that salt mines are – and will continue to be – intrinsically stable systems over geological timeframes. Predictions on long-term stability are however very complex. Dynamics of underground deposits over centuries or more are not possible today, since the number of influential

⁹⁰ Presentation of Dr. Gerd Hensel, of GSF Gesellschaft für Strahlen- und Umweltforschung (operating organisation of Asse II), October 20, 2001 in Wolfenbüttel, Germany. See (Hensel 2001) for full text, (GSF 2001) for research abstract and (Kleber 2001) for a summary.

⁹¹ Local intensities given in units of the Medvedev-Sponheuer-Karnik macroseismic scale (Leydecker 2001a).

⁹² Carnallite is a potassium-magnesium chloride (KCl·MgCl₂·6H₂O) occurring in salt veins, apart from rock salt (NaCl) and Sylvite (KCl).

parameters increases with the duration modelled. Interaction mechanisms grow more complex and simulations won't give usable results⁹³.

There is no direct evidence on emissions from underground deposits, i.e. no emissions from the waste occur today – in contrast e.g. to surface landfills. The probability of a detrimental development with transport of pollutants to the biosphere cannot be calculated within this study, but can be assumed to be non-zero. Such emissions would have to be discussed within a risk assessment study of underground deposits. Remote risks and rare accidents are not the object of life cycle inventories in ecoinvent 2000, but rather 'regular' and normal operation releases (Frischknecht et al. 2003a). **For these reasons no direct emissions from the waste in underground deposits are inventoried.** The only burdens in the inventory stem from transport, packaging and conditioning⁹⁴.

10.2 Operation of underground deposits

Information on the operation process in Herfa-Neurode is available from the operators of the mine (K+S 2002a, K+S 2002b) and secondary sources (Brendel 2000, WEKA 1993).

The expenditures to create the mine are fully allocated to the produced salt. I.e. it is assumed that the creation of the mine is or was motivated by the exclusive goal to produce salt/potash. The disposal function of the salt mine, which historically originated later, receives the necessary salt mine infrastructure (shafts and tunnels) without any burden. This understanding can also be substantiated by financial data of Kali+Salz GmbH, where the disposal branch contributes only 3% to the total annual turnover (K+S 1998).

Columns of salt remain in the mine to support the layers above ("room and pillar" system). The mine is designed to support the fourfold of the present weight to ensure mechanical stability (K+S 2002a:6). The columns make up 35% to 45% of the total area in the mine (WEKA 1993). Only 55% to 65% are usable and a further 15% of that (9% of the total mine area) are assumed to be used for access roads. Tunnel height is between 2.5 and 2.9 metres (Brendel 2000). Tunnel width is 16 meters on average.

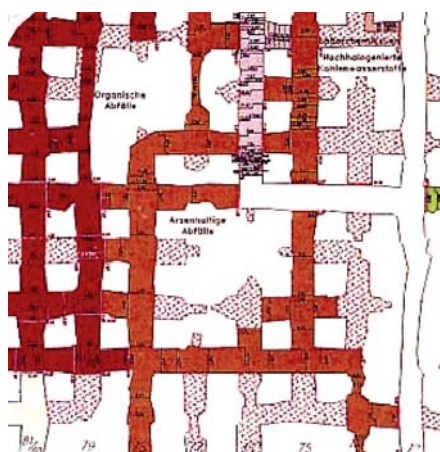


Fig. 10.4 Layout plan of a part of the Herfa-Neurode underground deposit. The nominal width of a tunnel is approximately 16 metres (K+S 2002a)

Wastes are deposited in three kinds of containers:

⁹³ Personal communication with Prof. Dr. Eckhard Grimmel, Geographical Institute, University Hamburg, in March 25, 2002.

⁹⁴ However, the LCIA method of ecological scarcity (BUWAL 1998) features a midpoint eco-factor per kilogram waste to underground deposits. This might be interpreted as a valuation of long-term risks associated with underground deposits.

- Steel drums of 200 l with an inner plastic lining (Fig. 5.5 on page 12)
- Large steel containers (Fig. 10.5 left)
- Big bags (plastic) for dusts (Fig. 10.5 right)

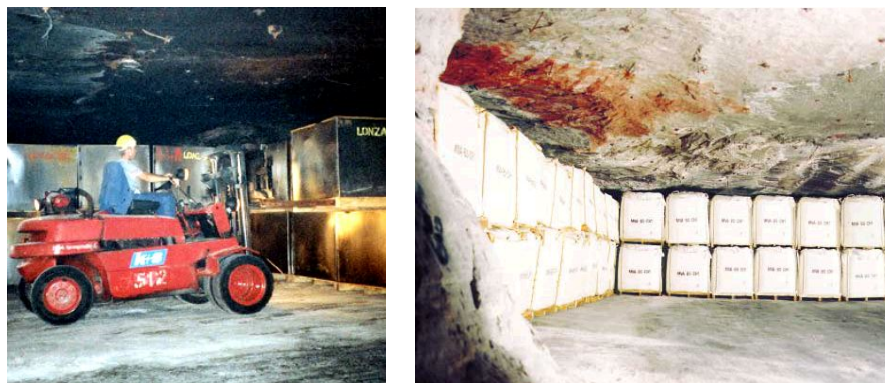


Fig. 10.5 Deposition of large steel containers in Herfa-Neurode (left, K+S 2002b) and deposit of big bags (right, K+S 2002a)

The containers are stacked on wood pallets in two or three layers. An average waste density of 1600 kg/m^3 is assumed. Steel drums (200 l) are estimated to consist of 31 kg of steel, 1.37 kg of plastic lining and 360 g of surface paint. Four steel drums are placed on one wood pallet of 32 kg (assumed to be softwood with 20% humidity and density 540 kg/m^3 , one pallet is 0.059 m^3 wood). One steel drum holds 320 kg of waste. For some wastes solidification with cement is necessary. For solidified waste, a steel drum is estimated to contain 25 V% cement. Such a steel drum holds only 240 kg of waste (density 1600 kg/m^3) plus 110 kg of cement (density 2200 kg/m^3), and hence uses more volume and materials per kg of waste.

The large steel containers are estimated to consist of 166 kg steel and 1.9 kg surface paint⁹⁵. One steel container is placed on one wood pallet of 32 kg (0.059 m^3). One container holds 3600 kg of waste (2.25 m^3).

Big bags are estimated to consist of 23 kg of LDPE plastic. One steel big bag is placed on one wood pallet of 32 kg (0.059 m^3). One big bag holds 5400 kg of waste (3.4 m^3).

Compartments are separated from the rest of the mine with 24 cm brick walls as an odour barrier⁹⁶ and also to economise mine ventilation. One compartment is estimated from plans to contain on average 2600 tons of waste. One compartment wall is estimated to have a width of 16 m, a height of 2.7 m, a depth of 20 cm and is estimated to contain 16.6 tons of bricks and 1.3 tons of cement mortar.

A field with 1 km by 1.5 km overall dimensions is additionally separated with a large anhydrite dam (WEKA 1993). This dam consists of two brick walls and a 6 m layer of anhydrite filling (CaSO_4). The width and height of the field wall are estimated to be 16 m and 2.7 m. One field wall contains 33.2 tons of bricks, 2.65 tons of cement mortar and 260 tons of anhydrite. One 1 km by 1.5 km field is estimated to contain 3.1 million tons of waste.

The deposit is separated against the conventional, unprotected potash mine with a 48 m wall of 4645 m^3 volume (Brendel 2000). The wall is assumed to consist of 9 brickwall layers filled with anhydrite. The whole mine separation wall is estimated to contain 334 tons of bricks, 26.8 tons of

⁹⁵ Estimated measures of steel container $1.5 \text{ m} \times 1.5 \text{ m} \times 1 \text{ m}$.

⁹⁶ Personal communication with Jörg Glienke, Kali und Salz Entsorgung GmbH, Kassel, in November 15, 2002.

cement mortar and 4440 tons of anhydrite. With a total running time of approximately 45 years⁹⁷ and an average deposition rate of 100'000 tons of waste per year, the expenditures of the mine separation wall are distributed amongst a total of 4.5 million tons of waste⁹⁸.

An additional expenditure for 'plaster mixing' is inventoried in the magnitude of the sum of cement mortar (in brick walls) and anhydrite (in dams).

Electricity demand for illumination and ventilation⁹⁹ is estimated to be 0.47 kWh/ton of waste (German medium voltage mix including imports).

Closure procedure

When deposition operations cease, the 800 m deep access shaft of the deposit will probably be completely filled with gravel and clay to seal off surface water (¹⁰⁰, Brendel 2000, K+S 2002a:15). The shaft cross section is estimated to be 5 by 3 meters, resulting in a volume of 12'000 m³ to be filled. Clay is estimated to be filled from the surface to a depth of 60 m (twice the groundwater depth). With a clay density of 2200 kg/m³, a total clay demand of 1980 tons results. The rest of the shaft (740 m or 11'100 m³) is filled with gravel. With a gravel density of 2200 kg/m³, a total gravel mass of 24'420 tons results. Per kilogram of waste 0.44 grams of clay and 5.4 grams of gravel are needed. 50% of these amounts are assumed to be recycled material from excavation works, the other 50% is clay from a clay pit and new crushed gravel, respectively. 100% of the material is transported 50 km with a lorry. Only the virgin material is inventoried as a material demand; recycled materials are not inventoried as a demand (a cut-off boundary resulting in no burden, except for transportation).

As a natural resource demand, the deposit volume occupied by waste is inventoried in m³. This figure is the inverse of the average waste density (1/1600 kg/m³). For solidified waste this figure is larger, as the solidifying cement is included.

10.2.1 Transport of materials

Standard transportation distances in Europe are used for material transport (Frischknecht et al. 2003a). The transport of the waste itself to the deposit (500 km from Switzerland) is *not* inventoried here. Steel, paint, plastic is transported 200 km by train and 100 km by lorry (32t). Bricks, cement, mortar, and anhydrite are transported 100 km by train and 100 km by lorry. Wood for pallets is transported 100 km by train and 50 km by lorry. Clay and gravel is transported 50 km by lorry.

10.2.2 Uncertainty for underground deposits

The GSD² value for material inputs for underground deposits is shown in Tab. 10.1. The same value is used for energy demand. GSD values for totals of transport services (in tkm) are calculated from the product of material and standard distance for any single material type and the approximated formula for addition (Wilkinson-Fenton) over all material types (see section on uncertainty of standard distance in part I). GSD values for plaster mixing are calculated from uncertainties of anhydrite and cement mortar and the Wilkinson-Fenton formula.

⁹⁷ Herfa-Neurode started operations in 1972 and in 1997 had a licensed capacity for approximately another 20 years, until 2017.

⁹⁸ With an average waste density of 1600 kg/m³ this corresponds to a total volume of 2.8 million cubic meters.

⁹⁹ Based on online calculations according to SIA-Norm 196 for tunnel ventilation at <http://www.tunnel.ch/solve1.asp>.

¹⁰⁰ Personal communication with Jörg Glienke, Kali und Salz Entsorgung GmbH, Kassel, in November 15, 2002.

Tab. 10.1 GSD² values for material inputs for underground deposits

Exchange	GSD ² value	Pedigree codes	Comment
Underground deposit material	301%	(2,3,1,1,1,n.a.)	Basic uncertainty of 3; raw data from 1 operator (Herfa-Neurode) and estimates

10.2.3 Infrastructure of underground deposits

It is debatable, which parts of an underground deposit can be considered as the 'infrastructure'. In most industrial processes the infrastructure or production means are used to perform a desired task on the product. The product enters, proceeds through and leaves the infrastructure. The infrastructure is removed or replaced when aged. The only doubtless infrastructure part of the deposits are the tunnels and shafts in the salt vein, created by the salt extraction. The walls and dams erected for the underground deposit are part of the technical barrier system of the deposit and can be seen as an extended form of deposit packaging. Energy used for illumination and ventilation is process energy.

It is decided not to create any separate infrastructure modules for underground deposits. The only actual infrastructure (creation of tunnels and shafts) is fully allocated to the salt production and does not appear in the inventories of underground deposition.

10.2.4 Inventory data

Tab. 10.2 shows the inventory data for 1 kg hazardous waste in underground deposit for four different types of packaging. Tab. 10.3 lists the different waste materials to underground deposit and their packaging.

Tab. 10.2 Inventory data for 1 kg hazardous waste in underground deposit for four different types of packaging

3702 Name	3703 Location	3508 Infrastructure Process	3706 Unit	1 kg waste large steel container	1 kg waste 200l steel drum cement- solidified	1 kg waste 200l steel drum unsolidified	1 kg waste Big Bag
steel, converter, unalloyed, at plant	RER	0	kg	0.0461	0.131	0.098	0
alkyd paint, white, 60% in solvent, at plant	RER	0	kg	0.000525	0.00149	0.00112	0
sawn timber, softwood, raw, kiln dried, u=20%, at plant	RER	0	m ³	0.0000165	0.0000617	0.0000463	0.000011
packaging film, LDPE, at plant	RER	0	kg	0	0.00569	0.00427	0.0043
cement, unspecified, at plant	CH	0	kg	0	0.458	0	0
clay, at mine	CH	0	kg	0.00022	0.0002933	0.00022	0.00022
gravel, crushed, at mine	CH	0	kg	0.005427	0.007236	0.005427	0.005427
brick, at plant	RER	0	kg	0.006438	0.008584	0.006438	0.006438
cement mortar, at plant	CH	0	kg	0.000515	0.0006867	0.000515	0.000515
anhydrite, at plant	CH	0	kg	0.00107	0.001427	0.00107	0.00107
plaster mixing	CH	0	kg	0.001586	0.002114	0.001586	0.001586
electricity, medium voltage, at grid	DE	0	kWh	0.0004436	0.0005915	0.0004436	0.0004436
transport, lorry 32t	RER	0	tkm	0.006201	0.06275	0.01269	0.001822
transport, freight, rail	RER	0	tkm	0.01101	0.07781	0.02398	0.002255
Volume occupied, underground deposit			m ³	0.000625	0.0008333	0.000625	0.000625

Tab. 10.3 Waste materials to underground deposit and their packaging.

Packaging	Waste
200 liter steel drum with cement solidification	sludge from FeCl ₃ production, 30% water
200 liter steel drum without cement solidification	catalytic converter NOx reduction, 0% water hazardous waste, 0% water (average) catalytic converter for cars, 0% water catalyst for EDC production, 0% water
Big Bag	spent activated carbon with mercury, 0% water
large steel container	waste, silicon wafer production, 0% water

The unit process inventories (i.e. non-cumulated data) of all waste materials to underground deposit are also given as Excel tables on the CD-ROM. The workbook contains the list of exchanges in ECOSPOLD format and has the file name '13_Underground_deposit_LCI.xls'.

11 Life Cycle Inventory for Landfarming

11.1 Introduction

A wide range of wastes and by-products of industrial processes is being spread on the land in agriculture, forestry and land reclamation operations. These procedures are varyingly called landfarming, land spreading or surface spreading (DGE 2001). A common practice is the land spreading of municipal wastewater treatment sludge on agricultural areas. Some materials generated from industrial processes are considered, by the producers at least, to be by-products rather than wastes so that they can be recycled to land as soil amendments and fertilisers with minimum restriction. Various regulations apply to the practice of land spreading but most share the underlying assumption that it is for the benefit of the soil. Chief motivation, however, is likely to be the low cost of this disposal option. More than 90% of the waste spread on land in the European Union is farm waste and predominantly animal manure. Of the remaining 10%, the most important are food production wastes, dredgings from waterways and paper waste sludge (DGE 2001). Leaving aside farm wastes, it would require access to only about 1% of agricultural land in the EU to landspread the industrial wastes to the current extent (DGE 2001).

It is expected that land spreading will increase in Europe following the implementation of national restrictions on the disposal of organic-rich materials in landfills and increases in the treatment of organic-rich industrial effluent from different branches of the food and drink and other sectors (DGE 2001).

In Switzerland, however, landfarming of wastewater treatment sludge is in the process to be phased out for all applications until 2005. In the year 2000 a share of 39% of the wastewater treatment sludge was recycled in agriculture, while the rest was incinerated (BUWAL 2001g). The trend is declining. The reason for this development is that most Swiss farmers in both integrated and organic production schemes balance the nutrition flows and losses on their land and have decreasing need for additional nutrition input. Other reasons are the pollution risk from waste water contaminants like AOX or heavy metals as well as concerns regarding 'mad-cow-disease' (BSE) type of risks (Chassot & Mühlethaler 2001).

11.2 System description and inventory

Direct emissions

Only three wastes are inventoried to landfarming¹⁰¹. No separate disposal model is created for these wastes. *The waste composition is completely inventoried as direct emission to soil.* Waste compositions are detailed in part I.

A distinction of the area type (agricultural or not) is important, because on agricultural soils much larger LCIA scores result, due to large human toxicity burdens via the agricultural food chain. According to (Concawe 1989) landfarming in the oil industry is performed on specially designated areas, which are not used for food or crops. Landfarming of these wastes is assumed to take place on *non-agricultural* fields for the two oil industry wastes (drilling waste, refinery sludge). For wood ash an application to agricultural areas (meadows, crops) or non-agricultural areas (forest) is possible (Noger et al. 1996:23). Here an application of wood ash on agricultural surface (i.e. on a food crop area) is inventoried.

¹⁰¹ Drilling waste (oil & gas drilling), refinery sludge (oil refinery), and wood ash mixture from untreated wood. Spreading of wastewater treatment sludge is not included in the report of agricultural products (Nemecek et al. 2003). Spreading of wastewater treatment sludge is heeded in a waste-specific manner in the treatment of wastewater (cf. part IV on wastewater treatment).

Speciation of emissions

All chemical elements (C, N, P, S, etc.) are emitted as such. No conversion (e.g. phosphorus to phosphate) is performed.

Spreading process

For the spreading the agriculture data module 'slurry spreading, by vacuum tanker' (in m³) is adopted as a proxy. A general waste density of 1000 kg/m³ is assumed (pumpable waste). For one kilogram of waste 0.001 m³ slurry spreading are inventoried. The uncertainty of the spreading process is estimated with the Pedigree approach heeding the different technology and different materials (see Tab. 11.1).

Tab. 11.1 Uncertainty of vacuum tank spreading for landfarming

Exchange	GSD ² value	Pedigree codes	Comment
Vacuum tank spreading for landfarming	150%	(n.a.,n.a.,1,1,4,n.a.)	Basic uncertainty of 1.05; vacuum tank spreading (in m ³) for landfarming of waste (in kg)

Uncertainty

Uncertainty in soil emissions are directly adopted from the uncertainty in waste composition. The uncertainty in waste composition in turn is estimated with the generic approach described in chapter 'Uncertainty of waste composition data' in part II (incineration).

11.3 Unit process inventories

The unit process inventories (i.e. non-cumulated data) of all waste materials to landfarming residual landfill is a very large table of 80 rows and 20 columns. A table representation in print is not very sensible, as many cells contain no data and some comment cells contain long text entries. The inventoried data for waste disposal processes is therefore contained in Excel tables on the CD-ROM. These workbooks contain the list of exchanges in ECOSPOLD format. Unit process data for landfarming is given in file '13_Landfaming_LCIv2.xls'.

12 Cumulative Results and Interpretation

12.1 Introduction

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

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

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

Name			disposal, municipal solid waste, 22.9% water, to sanitary landfill	disposal, paper, 11.2% water, to sanitary landfill	disposal, polyethylene, 0.4% water, to sanitary landfill	
			CH	CH	CH	
Location			kg	kg	kg	
Unit			0	0	0	
Infrastructure						
LCIA results			Unit			
cumulative energy demand	non-renewable energy resources, fossil	MJ-Eq	0.3400	0.3510	0.3190	
cumulative energy demand	non-renewable energy resources, nuclear	MJ-Eq	0.0699	0.1020	0.0205	
cumulative energy demand	renewable energy resources, water	MJ-Eq	0.0244	0.0361	0.0064	
cumulative energy demand	renewable energy resources, wind, solar, geothermal	MJ-Eq	0.0011	0.0015	0.0005	
cumulative energy demand	renewable energy resources, biomass	MJ-Eq	0.0023	0.0030	0.0011	
LCI results			Unit			
resource	Land occupation	total	m2a	0.00461	0.00466	0.00452
air	Carbon dioxide, fossil	total	kg	0.0194	0.0138	0.033
air	NMVOG	total	kg	0.0000446	0.000045	0.0000437
air	Nitrogen oxides	total	kg	0.000267	0.000277	0.00025
air	Sulphur dioxide	total	kg	0.0000503	0.0000737	0.0000204
air	Particulates, < 2.5 um	total	kg	0.000027	0.0000297	0.0000243
water	BOD	total	kg	0.0189	0.0203	0.0541
soil	Cadmium	total	kg	3.91E-11	3.93E-11	3.82E-11
Further LCI results			Unit			
water	Cadmium, ion	total	kg	0.0000117	0.00000163	0.0000339
water	Copper, ion	total	kg	0.00121	0.0000609	0.0000414
water	Lead	total	kg	0.000502	0.0000804	0.0000226
water	Zinc, ion	total	kg	0.00108	0.000104	0.000283

These life cycle inventories include the landfilling itself, but also the treatment of leachate in the first 100 years after waste deposition. Looking at the inventory data, we find that paper burdens are often the highest, while polyethylene burdens are the lowest, and the municipal waste is intermediate. The reason for this is that paper, as a well degradable waste, leads to additional burdens from landfill gas combustion and leachate treatment as opposed to a hardly disposable waste which leaves the leachate relatively unburdened. Waste decomposition after 100 years will not lead to any air emissions, because at that point the methane phase of the landfill is terminated and no landfill gas is produced. The increased expenditures for leachate treatment and landfill gas combustion make themselves felt in the cumulative energy demand CED, but also in land occupation and most air pollutants. The most relevant contributions to the total burden originate from water emissions, so the statement that the CED of polyethylene disposal is lower than the CED of paper disposal should not be taken as indicative of a relatively low *overall* score for polyethylene compared to paper, nor should it indicate that it is altogether advantageous to landfill hardly degradable wastes..

Paper landfilling has the lowest fossil carbon dioxide air emissions, because the waste itself contains no fossil carbon and all emissions come from background processes. The fossil carbon dioxide air emissions for polyethylene are increased because there are some direct emissions from the waste¹⁰². The fossil carbon dioxide air emissions for average municipal waste are intermediate, because the waste contains a mixture of biomass and plastic wastes.

The BOD emissions contain emissions from wastewater treatment of the short-term leachate and the long-term emissions directly from the landfill. BOD emissions very roughly represent carbon content of the waste. Paper has a lower carbon content than polyethylene and also a part of the carbon in the leachate will be converted to CO₂ and sewage sludge during wastewater treatment and thus not show up as BOD in the WWTP effluent.

Other water emissions are dominated by waste-specific emissions which depend on waste composition. Municipal solid waste has high emissions of copper, lead and zinc due to presence of bulk metal waste fractions. Cadmium concentration in polyethylene is 36 ppm, in paper 1.7 ppm which reflects the differences in total cadmium emissions.

12.3 Results for slag compartment

Waste to slag compartment is only generated in municipal incineration plants MSWI. The landfilling of bottom ash from MSWI is integrated in the inventories of municipal incineration, which are discussed in part II. The behaviour of slag compartments is compared to that of other landfill types in chapter 12.8 'Landfill model comparisons' on page 93.

12.4 Results for residual material landfill

Tab. 12.2 shows some arbitrary results of the cumulated inventory of waste disposal in a residual material landfill. Two wastes were chosen:

- Average residual waste
- Slag/dust from alloyed electric arc furnace (EAF) steel production

Average residual waste is the residual material from the incineration of municipal solid waste and is used as a proxy for residual waste materials of unknown composition, e.g. for APME/Bousted sources (cf. part I on waste compositions). Both wastes are solidified with cement prior to landfilling.

¹⁰² Polyethylene with 822 grams fossil carbon per kg waste, and an overall PE degradability of 1%, releases 0.008 grams of carbon to the landfill gas, of which 0.0023 grams are emitted directly as methane and 0.0056 grams are combusted, leading to 0.02 grams of carbon dioxide (weight increase factor C→CO₂ is 3.66). This accounts for the difference in fossil carbon dioxide emissions between paper and polyethylene.

Tab. 12.2 Selected LCI results and the cumulative energy demand for residual material landfill

Name				disposal, average incineration residue, 0% water, to residual material landfill	disposal, slag, alloyed EAF steel, 15.4% water, to residual material landfill
Location				CH	CH
Unit				kg	kg
Infrastructure				0	0
LCIA results				Unit	
	cumulative energy demand	non-renewable energy resources, fossil	MJ-Eq	2.0300	2.0300
	cumulative energy demand	non-renewable energy resources, nuclear	MJ-Eq	0.2450	0.2450
	cumulative energy demand	renewable energy resources, water	MJ-Eq	0.1060	0.1060
	cumulative energy demand	renewable energy resources, wind, solar, geothermal	MJ-Eq	0.0044	0.0044
	cumulative energy demand	renewable energy resources, biomass	MJ-Eq	0.0534	0.0534
LCI results				Unit	
resource	Land occupation	total	m2a	0.0142	0.0142
air	Carbon dioxide, fossil	total	kg	0.32	0.32
air	NMVOG	total	kg	0.000147	0.000147
air	Nitrogen oxides	total	kg	0.000865	0.000865
air	Sulphur dioxide	total	kg	0.000192	0.000192
air	Particulates, < 2.5 um	total	kg	0.000372	0.000372
water	BOD	total	kg	0.0449	0.00384
soil	Cadmium	total	kg	1.12E-10	1.12E-10
Further LCI results				Unit	
water	Cadmium, ion	total	kg	0.00000578	0.00000285
water	Copper, ion	total	kg	0.000374	0.0000556
water	Lead	total	kg	0.000108	0.0000605
water	Zinc, ion	total	kg	0.000923	0.000724

For each kilogram of waste to the residual landfill the same amounts of infrastructure, solidification cement and process energy are inventoried. No air emissions originate from the landfill directly, so all air emissions are indirect. Accordingly, there are no differences in cumulative energy demand, air or soil emissions between the two wastes. The only differences are in the water emissions which depend largely on waste composition. BOD emissions reflect roughly carbon content of the wastes.

The cumulative energy demand for these wastes is higher than for waste to sanitary landfill (cf. chapter Tab. 12.1 on page 87). The reason for this is the increased energy demand for the solidification cement, slightly better economy of scale of the sanitary landfill infrastructure and increased total landfilled mass per kg waste due to solidification for the residual wastes. Solidification of residual wastes is optional, i.e. not all residual material wastes are solidified (e.g. ash from the incineration of WWTP sludge from paper production is not solidified).

12.5 Results for inert material landfill

Tab. 12.3 shows some arbitrary results of the cumulated inventory of waste disposal in a inert material landfill. For wastes to inert material landfill currently no waste-specific emissions are inventoried and hence all wastes receive the same burden per kilogram, irrespective of composition.

Tab. 12.3 Selected LCI results and the cumulative energy demand for inert material landfill

Name		disposal, inert waste, 5% water, to inert material landfill
Location Unit Infrastructure		CH kg 0
LCIA results		
cumulative energy demand	non-renewable energy resources, fossil	MJ-Eq 0.2030
cumulative energy demand	non-renewable energy resources, nuclear	MJ-Eq 0.0059
cumulative energy demand	renewable energy resources, water	MJ-Eq 0.0016
cumulative energy demand	renewable energy resources, wind, solar, geothermal	MJ-Eq 0.0002
cumulative energy demand	renewable energy resources, biomass	MJ-Eq 0.0004
LCI results		
resource	Land occupation	total m2a 0.00154
air	Carbon dioxide, fossil	total kg 0.00679
air	NMVOG	total kg 0.0000269
air	Nitrogen oxides	total kg 0.000149
air	Sulphur dioxide	total kg 0.0000105
air	Particulates, < 2.5 um	total kg 0.0000138
water	BOD	total kg 0.0000208
soil	Cadmium	total kg 2.56E-11
Further LCI results		
water	Cadmium, ion	total kg 3.59E-09
water	Copper, ion	total kg 0.000000016
water	Lead	total kg 2.64E-08
water	Zinc, ion	total kg 0.000000372

The cumulative energy demand CED is lower than for sanitary landfills, which can be explained by a more modest infrastructure construction and no compaction energy for inert material landfills.

The emissions to waster originate entirely form indirect processes, and not from the landfill.

12.6 Results for underground deposits

Tab. 12.4 shows some arbitrary results of the cumulated inventory of waste disposal in a residual material landfill. Four wastes were chosen:

- Hazardous waste (average), deposited in a 200 l steel drum without cement solidification
- Sludge from FeCl₃ production, deposited in a 200 l steel drum with cement solidification
- Spent activated carbon with mercury, deposited in a polyethylene Big Bag
- Silicon wafer production waste, deposited in a large steel container

Tab. 12.4 Selected LCI results and the cumulative energy demand for underground deposit

Name	Location	Unit	Infrastructure	disposal, hazardous waste, 0% water, to underground deposit	disposal, sludge from FeCl ₃ production, 30% water, to underground deposit	disposal, spent activated carbon with mercury, 0% water, to underground deposit	disposal, waste, silicon wafer production, 0% water, to underground deposit
				DE kg	DE kg	DE kg	DE kg
LCIA results				0	0	0	0
		Unit					
cumulative energy demand	non-renewable energy resources, fossil	MJ-Eq		1.9600	4.2500	0.4200	0.7500
cumulative energy demand	non-renewable energy resources, nuclear	MJ-Eq		0.1900	0.5120	0.0493	0.0709
cumulative energy demand	renewable energy resources, water	MJ-Eq		0.0427	0.1720	0.0110	0.0159
cumulative energy demand	renewable energy resources, wind, solar, geothermal	MJ-Eq		0.0090	0.0164	0.0016	0.0038
cumulative energy demand	renewable energy resources, biomass	MJ-Eq		0.6070	0.8690	0.1320	0.2250
LCI results							
		Unit					
resource	Land occupation	total	m2a	0.113	0.153	0.0267	0.0405
air	Carbon dioxide, fossil	total	kg	0.0836	0.46	0.0144	0.0347
air	NMVO	total	kg	0.00012	0.00025	0.0000392	0.0000396
air	Nitrogen oxides	total	kg	0.000432	0.0012	0.0000695	0.000177
air	Sulphur dioxide	total	kg	0.000336	0.000639	0.0000499	0.000137
air	Particulates, < 2.5 um	total	kg	0.000113	0.000544	0.00000567	0.0000515
water	BOD	total	kg	0.00137	0.00192	0.0000104	0.000641
soil	Cadmium	total	kg	2.82E-10	4.8E-10	2.48E-11	1.24E-10
Further LCI results							
		Unit					
water	Cadmium, ion	total	kg	9.13E-09	2.67E-08	1.67E-09	3.87E-09
water	Copper, ion	total	kg	0.000000416	0.000000667	6.41E-08	0.000000168
water	Lead	total	kg	0.000000129	0.000000286	2.11E-08	5.36E-08
water	Zinc, ion	total	kg	0.0000765	0.000148	0.00000137	0.0000356

No emissions that depend on the waste composition were inventoried for underground deposits. Accordingly all burdens originate mainly from background processes and most prominently from packaging. For all shown results the FeCl₃ production sludge has the highest burdens, the average hazardous waste comes next, and spent activated carbon and silicon wafer production waste are a clearly less burdening. All these differences come from the differences in packaging.

The FeCl₃ production sludge is deposited in a 200 l steel drum with cement solidification. The average hazardous waste is also in a 200 l steel drum, but has no cement solidification. The difference between the former and the latter are due to cement solidification which clearly plays an important role.

Silicon wafer production waste is deposited in a large steel container. Large steel containers have a volume of 2250 l and therefore have a better economy of scale (e.g. 74 kg of steel per m³ packed as compared to 155 kg/m³ for a 200 l steel drum, cf. chapter 10.2 'Operation of underground deposits' on page 80). This reduced the burdens from packaging even further.

Spent activated carbon is deposited in a polyethylene Big Bag with a volume of 3400 l. Only 23 kg of PE are needed per bag or 6.8 kg per m³ packed. It wasn't determined here if the burdens *per kilogram* are higher for PE or for steel, but apparently the even larger economy of scale of Big Bags makes this type of packaging the least burdening.

12.7 Results for landfarming

Only three wastes to landfarming are inventoried. Tab. 12.5 shows some arbitrary results of the cumulated inventory of waste landfarming.

- Drilling waste from oil and natural gas production
- Refinery sludge from oil refining
- Wood ash from natural wood incineration

Tab. 12.5 Selected LCI results and the cumulative energy demand for landfarming

Name			disposal, drilling waste, 71.5% water, to landfarming	disposal, refinery sludge, 89.5% water, to landfarming	disposal, wood ash mixture, pure, 0% water, to landfarming	
Location			CH	CH	CH	
Unit			kg	kg	kg	
Infrastructure			0	0	0	
LCIA results			Unit			
	cumulative energy demand	non-renewable energy resources, fossil	MJ-Eq	0.0176	0.0176	0.0176
	cumulative energy demand	non-renewable energy resources, nuclear	MJ-Eq	0.0016	0.0016	0.0016
	cumulative energy demand	renewable energy resources, water	MJ-Eq	0.0006	0.0006	0.0006
	cumulative energy demand	renewable energy resources, wind, solar, geothermal	MJ-Eq	0.0000	0.0000	0.0000
	cumulative energy demand	renewable energy resources, biomass	MJ-Eq	0.0005	0.0005	0.0005
LCI results			Unit			
resource	Land occupation	total	m2a	0.000146	0.000146	0.000146
air	Carbon dioxide, fossil	total	kg	0.00108	0.00108	0.00108
air	NMVOG	total	kg	0.00000221	0.00000221	0.00000221
air	Nitrogen oxides	total	kg	0.0000108	0.0000108	0.0000108
air	Sulphur dioxide	total	kg	0.00000173	0.00000173	0.00000173
air	Particulates, < 2.5 um	total	kg	0.00000115	0.00000115	0.00000115
water	BOD	total	kg	0.0000041	0.0000041	0.0000041
soil	Cadmium	total	kg	2.14E-11	0.0000102	0.0000142
Further LCI results			Unit			
water	Cadmium, ion	total	kg	2.92E-09	2.92E-09	2.92E-09
water	Copper, ion	total	kg	8.14E-09	8.14E-09	8.14E-09
water	Lead	total	kg	2.06E-08	2.06E-08	2.06E-08
water	Zinc, ion	total	kg	0.000000126	0.000000126	0.000000126

The inventory for landfarming only consists of a expenditure for spreading by vacuum tanker and direct emissions to soil. All cumulative energy demand results originate from the spreading process, as well as land occupation, air and water emissions.

Cadmium emissions to soil from refinery sludge and wood ash are from the waste itself which contain 10 ppm and 14 ppm cadmium in the waste composition, respectively. No cadmium concentration is known for drilling waste. The given cadmium emission to soil is therefore entirely from background processes.

12.8 Landfill model comparisons

12.8.1 Comparison of short-term transfer coefficients in inorganic landfills

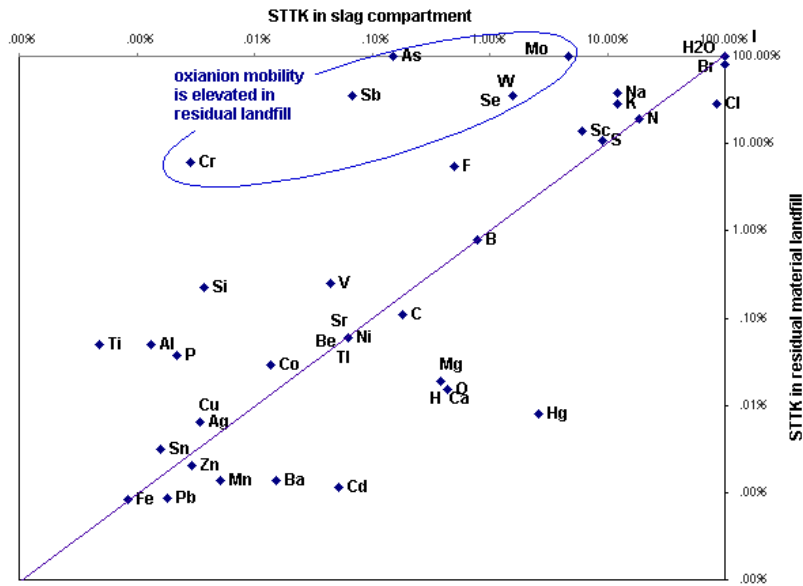


Fig. 12.1 Comparison of short-term transfer coefficients in slag compartments vs. residual landfills. Please note the double-logarithmic scale

The comparison of short-term transfer coefficients (STTK) in slag compartments vs. residual landfills allows a comparison of the different release and hence mobility of pollutants over 100 years in these two landfill types (Fig. 12.1). The comparison reveals a significantly larger discharge for Cr, Sb, Se, W, Mo, and As in residual landfills. This effect is less pronounced for V and Si. All these elements form oxianions¹⁰³. The mobility of anions and hence their emission vs. retention depends on the anion exchange capacity AEC. The anion exchange capacity is often associated with presence of oxide surfaces, notably iron, manganese, and aluminum oxides, carbonate surfaces, and insoluble organic matter (McLean & Bledsoe 1992:6). The concentrations of these elements are higher in slag compartments than in residual material landfills (e.g. 7 times higher for iron, 16 times for Mn and 3 times for Al; cf. average waste compositions in Tab. A.2 and Tab. A.4 on page 112f.) The reason for this difference is that these elements are predominantly transferred to bottom ash in MSWI (transfer coefficients to bottom ash in average municipal waste is 97.2% (Fe), 86.7% (Mn), and 91%(Al)). The relative scarcity of these AEC-forming elements in residual landfills lowers the AEC and increases the oxianion mobility as observed in the model data.

Chromium is much more mobile in residual landfills than in slag landfills. This can be explained by the presence of Cr as a Cr^{VI} oxianion in leachate (CrO_4^{2-}). Additionally this may be due to a relatively low chromium mobility in slag compartments by incorporation of Cr to initially *uncorroded alloyed*

¹⁰³ Silicon was *not* truly regarded as an oxianion-forming element in this project, but behaves like one in the model (low solubility at high pH). However, silicon actually *is* able to form oxianions (like from silicic acid, SiO_4H_4 which will dissociate to an anion like $\text{SiO}_4\text{H}_2^{2-}$ in high pH environments, or the silicate anion SiO_4^{4-} , respectively). For the last remaining oxianion-forming element in the model, boron, the STTK in the residual material landfill was adopted directly from the slag compartment, so no difference is visible in the model. Though based on these findings the STTK for boron in the residual material landfill should probably be around 13–19% and not 0.78% as currently used.

steel phases in MSWI slag, i.e. material heterogeneity. Another observation is that mercury and cadmium are much less mobile in residual material landfills.

Caution must be taken in interpreting Fig. 12.1, as some elements were extrapolated from coal ash or other elements, and some STTK for residual material landfills were adopted from slag compartments (N, B, Fe; see chapter 'Extrapolations to other elements' on page 70). That the derived short-term transfer coefficients are able to show realistic and reasonable phenomena, like increased oxianion mobility, is however encouraging.

12.8.2 Comparison of long-term transfer coefficients in landfill models

Since the landfill models in this study are new and include a lot of new elements it is interesting to cross-examine the model behaviour. The most relevant parameter is the long-term transfer coefficient, which expresses the percentage of emitted element over the course of 60'000 years since waste placement (i.e. including the short-term releases 0-100a). Together with the elemental waste composition the transfer coefficients determine the direct releases from the landfill, which are the usually dominant contribution in the LCA result of waste disposal activities.

Three types of landfill are regarded for the comparison:

- Slag compartment (MSWI bottom ash landfill)
- Residual material landfill (inorganic hazardous landfill)
- Sanitary landfill (landfill for untreated municipal waste)

The slag compartment only receives bottom ash from MSWI. No direct landfilling to slag compartments is inventoried in this report. Bottom ash from incinerated waste is however landfilled in slag compartments and included in the inventories of MSWI waste disposal.

The transfer coefficients for slag compartments and residual material landfills are constant for every waste. The transfer coefficients for sanitary landfills are variable according to waste degradability (cf. chapter 6.1.4 'Long-term landfill development' on page 49). For the comparison presented here the transfer coefficients calculated for an average waste municipal waste composition are used.

This report also inventories inert material landfills. There however no relevant direct emissions from waste were assumed and all transfer coefficients are zero. For this reason inert material landfills are not included in this analysis.

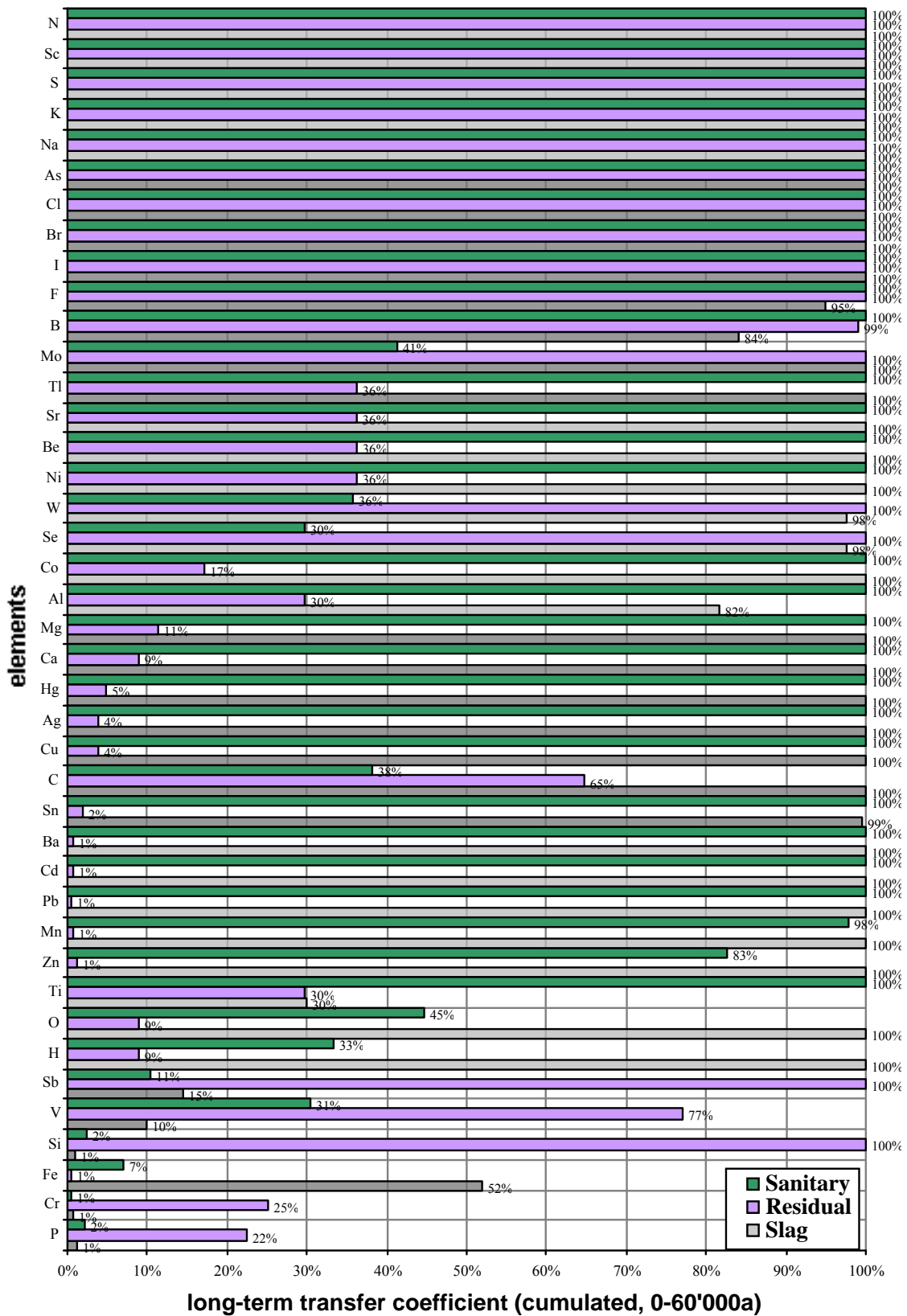


Fig. 12.2 Synopsis of the long-term transfer coefficients for MSWI slag compartments (bottom ash landfill), residual material landfill and sanitary landfill in the case of average municipal solid waste. Sorted according to average over the three values.

Distinct rankings or groups can be established when looking at the retention behaviour of elements in the different landfill types.

Group 1: The comparison of the long-term transfer coefficients (Fig. 12.2) shows that there are some elements which behave **without difference** irrespective of landfill type. These are all typically highly soluble elements, which are easily washed of from the landfill site: potassium K, sodium Na, the halogens (Cl, Br, I), nitrogen N (as nitrate) and sulfur S (as sulfate).

A bit surprising is the presence of scandium Sc and arsenic As in this highly soluble group. The high solubility of scandium is suggested from coal ash landfills (between phosphorus and potassium, cf. Tab. A.9 on page 121). This information was used in all three landfill types. Arsenic forms oxianions (HAsO_4^{2-}) and high solubility of arsenic is actually measured in sanitary landfills and slag compartments (cf. Tab. A.1 and Tab. A.2 on page 111f.). The LTTK for As in slag compartments was adopted to residual landfills. As shown in chapter 12.8.1 'Comparison of short-term transfer coefficients in inorganic landfills' on page 93 the solubility of oxianions in residual landfills tends to be *higher* than in slag compartments. This justifies in retrospect this extrapolation from slag compartments to residual landfills, as it does not lead to exaggerated transfer coefficients.

Group 2: For the vast majority of environmentally relevant elements the **residual material landfill shows the best retention behaviour**: For most elements the residual material landfill has smaller transfer coefficients than the slag compartment or the sanitary landfill. This is an expected result, since the residual material landfill has a large carbonate buffer which is not depleted within 60'000 years. Also it bears waste with lower carbon content than sanitary or slag landfills and hence less potential for biogenic activity.

Group 3: However for some elements the residual material landfill shows not the best retention behaviour. The retention of phosphorus P, vanadium V, and silicon Si is better in **slag compartments** than in residual material landfills or sanitary landfills¹⁰⁴. Phosphorus and vanadium both form oxianions (vanadate HVO_4^{2-} and phosphate PO_4^{3-}). Oxianions have elevated solubility at high pH but solubility drops, if pH turns acidic. As can be seen in Fig. 12.1 on page 93, V and P have already lower short-term transfer coefficients in slag compartments compared to residual material landfills. In slag landfills the depletion of the acid buffer occurs 23'000 years after waste deposition (cf. chapter 7.1.1 'Transfer coefficients for slag compartments' on page 65). After depletion of the buffer, the pH drops which *decreases* further oxianion emissions (but *increases* other metal emissions). On the other hand, in residual material landfills, oxianions are emitted with a relatively high rate which is not abated by a pH drop, since the carbonate phase lasts over 60'000 years. Retention for oxianions is therefore low in residual landfills. In slag compartments the mobility of oxianions is lower plus a pH drop after 23'000 years decreases the release even further. **Retention for oxianions is therefore higher in slag compartments than in residual landfills**¹⁰⁵. This also appears to be true for silicon Si, which is modelled – like oxianions – with decreasing solubility at low pH (cf. arrow 5 in Fig. 5.8 on page 27). Emissions of silicon are however not regarded as environmentally relevant and retention of silicon is of no importance.

Group 4: In a few cases the best retention behaviour is shown by the **sanitary landfill**. The sanitary landfill (average waste) has the lowest long-term transfer coefficients for Cr, Sb, Se, W, Mo and carbon. The metal species are again oxianions; and also carbon can be said to form an oxianion in the form of the carbonate anion CO_3^{2-} . However, for carbon the likely explanation is that the microbial

¹⁰⁴ For fluorine F and boron B the advantage of slag compartments is only very slight and not significantly different from group 1 (100% release in all three landfills). Fluorine is a soluble halogen and boron forms oxianions.

¹⁰⁵ This appears to be true for all oxianions with regard to long-term transfer coefficients (P, Cr, V, Sb, Se, W, B). For the oxianions Mo and As the long-term transfer coefficient are 100% for both landfill types: i.e. mobility of these elements is so high that the differences between slag compartment and residual material landfill in the early centuries are not large enough to make themselves perceptible in the long-term time scales.

activity tends to assimilate a soil-like substrate, which retains organic carbon better than matrix of the inorganic landfills. For the metal oxianions the explanation is likely to be sought in the short buffer phase of sanitary landfills. After 'only' 4500 years the carbonate buffer is depleted in sanitary landfills. After that the oxianion release is abated due to a low pH, as explained above. For Cr, Sb, Se, W, and Mo the oxianion mobility in the sanitary landfill is small enough in the early centuries to profit from this effect and promote better retention.

In short: residual material landfills will in general tend to have the best retention behaviour compared to slag compartments and sanitary landfill. However, for oxianion-forming elements (Cr, Sb, Se, V, W, Mo, As) the reverse is true and slag compartments or sanitary landfills will tend to have the best retention behaviour.

Please, remember that the statements for sanitary landfill made here refer to average municipal waste only. For specific waste fractions with different degradability the transfer coefficients can change substantially.

12.8.3 Do landfills really make a difference?

For quite a few elements the landfills have long-term coefficients of 100%. I.e. over 60'000 years the landfill will release those pollutants *completely*. It is however an exaggeration to consider landfills to have 'no effect' regarding the retention of pollutants¹⁰⁶. There is a *distinct difference in retention* in the various landfills types, as shown in Tab. 12.6. The residual landfill succeeds in retaining almost two thirds of different harmful elements. Even the *least* retaining landfill (sanitary for average waste) actually has a retaining effect *for a third of the modelled substances over 60'000 years*. Thus, it can be said that landfills are indeed purposeful and do show a mitigating effect. Whether this retention effect actually reduces the environmental burden for a certain waste, depends on the waste composition, i.e. if the elements present in this waste are retained. The main reason however today to run landfills is mitigation of *acute effects* and protection of groundwater from harmful concentrations of pollutants. Concentrations issues are however not heeded in the LCA approach.

Tab. 12.6 Count of the elements with a 100% average long-term transfer coefficient in the different landfill models

Landfill type	Elements with a 100% average long-term transfer coefficient				
	Harmful elements ¹	share ²	Harmless elements ³	share ²	Total
Sanitary landfill ⁴	22	52%	6	14%	67%
Slag compartment	22	52%	7	17%	69%
Residual material	12	29%	4	10%	38%

1 Harmful elements are heavy metals, halogens, carbon, sulfur, nitrogen, phosphorus, and boron.

2 Percentages relate to the 42 modelled elements (32 harmful + 10 harmless).

3 Harmless are water and oxygen, hydrogen, Ca, K, Mg, Na, Al, Fe, Si.

4 Only the transfer coefficients for average municipal waste are compared here. Transfer coefficients for specific waste fractions might differ due to different degradability.

¹⁰⁶ This study assumes an unrestrained leaching potential as expressed by (Sabbas et al. 1998, Lechner 2001, Huber et al. 1996) for geochemical reasons, and hence acknowledges the *possibility* of a 100% emission. An ad hoc criticism of this approach encountered during the study was that – in an LCA context – controlled landfills would then not be any different than uncontrolled dumping which could be considered a direct and complete emission of the waste pollutants. This criticism can be refuted.

12.9 LCI exchanges and available LCIA valuation

In the disposal inventories an extensive vector of 39 chemical elements were heeded¹⁰⁷. The current disposal models can calculate emissions for any of these elements. A prerequisite is however that waste composition data is available for that element. Even if composition data is available and used in the inventory, the burden associated with an element might not be visible in an LCIA result. LCIA methods can have gaps or missing classification factors. In Tab. 12.7 an overview of current and outdated LCIA methods is shown with their classification factors for the elements modelled in this report. The table is sorted according to 'popularity' of metals, i.e. counts per element (right column).

A tendency towards more complete valuation is observable with modern methods. However 14 entries in the current elements vector, or 36%, remain unheeded with these methods. Iodide, scandium, strontium, titanium, tungsten are elements with possible toxic effects. Aluminum and silver are toxic to fish. Potassium and BOD can contribute to eutrophication. Calcium, magnesium, sodium, chloride, and silicon are probably of little relevance in LCIA.

Cumulative Energy Demand CED

The method of Cumulative Energy Demand CED only values energetic resources and is not capable of detecting *any* burdens associated with emissions from waste materials. Energy consumption in disposal processes is usually of minor importance and bears no correlation to the total ecological burden of disposal processes. The common practice of associating energy consumption with environmental burden breaks down for disposal processes¹⁰⁸. Since disposal processes can – depending on material – become relevant for the whole life cycle, the notion that energy consumption is a proxy for environmental burden should generally be used with caution in LCA. Other methods are needed to adequately assess the environmental burdens over the life cycle.

One popularly assumed advantage of energy consumption as an 'environmental indicator' is the 'objectivity' of calculation. While the devised unit, usually megajoules, is indeed a defined physical unit, the procedure *how* to calculate Cumulative Energy Demand or primary energy consumption is not unambiguous and requires also several value judgements. One example of ambiguity is the energetic valuation of uranium in the nuclear electricity chain. Another is which energy resources shall be included. See (Frischknecht et al. 2003b) for implementation of CED in ecoinvent 2000.

CML'01

The method CML'01 incorporates the results from (Huijbregts 1999), who modelled fate for a large number of pollutants. With these results, 18 elements in the inventory (46%) can be valued. The shown category human toxicity HTP for a time horizon for 500a is only *one of several* categories from Huijbregts work. Others are freshwater aquatic ecotoxicity FAETP, marine aquatic ecotoxicity MAETP, freshwater sedimental ecotoxicity FSETP, marine sedimental ecotoxicity MSETP, and terrestrial ecotoxicity TETP. For all categories time frames of 20a, 100a, 500a, and infinite time are available.

¹⁰⁷ This is an improvement in comparison to the 10 elements which were fully inventoried in the landfill models of the former ETH inventories (marked with a '♥' in Tab. 12.7).

¹⁰⁸ In general terms and e.g. on a national scale it is often true that a lot of pollutants (especially air pollutants) come from energy consumption (e.g. road and air traffic, space heating). Hence there is some justification in using energy consumption as a proxy for environmental burdens. The reverse is often also true: the processes that are the most polluting often do consume a lot of energy. However for specific processes – like disposal – this simplification can be utterly wrong. There is essentially no point in trying to assess the environmental burdens of disposal by only looking at energy consumption.

The large number of heeded elements makes this a preferable method to assess environmental damages calculated from the disposal inventories of this study. Within the scope of this study the only time frame choice *compatible with the LCI* would be infinite time¹⁰⁹. However, the infinite time scores of (Huijbregts 1999) are distorted due to over- or underestimated residence times of pollutants in the ocean sub-model. This is especially the case for the two marine categories and/or for easily soluble elements like fluorine. A correction only for fluorine is available in (Huijbregts 2000). Since no consistent correction for all elements is available, a partial correction of fluorine alone was not heeded in the implementation of CML'01 in ecoinvent 2000 database v1.0¹¹⁰. If infinite time scores of CML'01 are used, fluorine often obtains an uncharacteristically high share of the burden. **The user is advised to use the 500 years time horizon scores instead.** Although this represents a time frame mismatch between the inventory and the valuation.

Apart from the mentioned generally missing factors for I, Sc, Sr, Ti, W, Al, and Ag, the CML'01 method lacks also factors for B, Br, and Mn (marked grey in Tab. 12.7).

Other methods

The popular methods Ecoscarcity 1997 and Eco-indicator'99 have comparatively poor coverage of 12 resp. 8 elements (31%, resp. 21%). If coverage is indeed poor for a specific waste also depends on the available waste composition data. Nitrate and COD/DOC/TOC are emissions that are only valued with Ecoscarcity 1997, but not within CML'01 toxicity scores. For the v2.1 of the ecoinvent database (2009) changes were implemented that *ignore* long-term emissions to groundwater in the LCIA scores for Ecoscarcity 1997, Ecoscarcity 2006, Eco-indicator'99 and EDIP 2003¹¹¹. These methods – as employed by ecoinvent – are therefore not suitable to properly assess the "grave" in cradle-to-grave analyses.

In general, a valuation of disposal inventories with CML'01 toxicity scores 500a is advisable, until the corrected infinite toxicity scores of CML'01 toxicity become available.

The relevance of completeness in the models can be shown for waste compositions where a lot of chemical elements are known. Such wastes are for example hard coal ashes. With the current model for residual material landfills, the most relevant landfill emissions of e.g. German hard coal ash, turn out to be (in decreasing order) vanadium, thallium, arsenic, selenium, molybdenum, antimony, nickel, and beryllium. *All these emissions were not heeded in the former ETH models for residual material landfills.* The burden calculated *only for the heeded emissions* in the former ETH model is almost 4 orders of magnitude lower than the score for the emissions from the present models. The relevance of the assessment gap is apparent and justifies in retrospect the extension of the element vector in the present models. However, even with the current models not all elements are covered and a few gaps for rare elements remain.

¹⁰⁹ Due to the long inventoried time frame of 60'000 years for leachate long-term emissions.

¹¹⁰ A future correction is expected at CML, depending on financing.

¹¹¹ For Eco-indicator'99 and EDIP 2003 new impact categories called "stored toxicity" were created in v2.1 that would assess the *remaining toxicity potential* of landfills after 100 years. In the ecoinvent implementation only the long-term *emissions*, not the landfill remainder, are assessed with these "stored toxicity" categories, and thus underestimate the toxicity potential. In any case, ecoinvent management chose not to include either "stored toxicity" categories as part of the sum total LCIA scores for Eco-indicator'99 and EDIP 2003.

Tab. 12.7 Synopsis of classification factors of emissions to (ground)water for elements inventoried in this study. Not shown are BOD, I, Sc, Sr, Ti, W, Al, Ag, Ca, Mg, Na, and Si.

Method	Cumulative Energy Demand	Swiss Ecoscarcity 1990	CML'92 HumTox water emissions	CML'92 Aquatic Ecotoxicity	Eco-indicator'95 heavy metals	Swiss Ecoscarcity 1997	Eco-indicator'99 (H,A)	CML'01 HumTox 500a	counts
score type	Aggregated score	Aggregated score	HumTox water emissions	Aquatic Ecotoxicity	heavy metals	Groundwater emissions	Aggregated score	HumTox water emissions	
unit	MJ/kg	Ecopoints90 / kg	kg TLV burdened body weight /kg	m ³ TLV polluted /kg	kg Pb equivalents / kg	Ecopoints97 / kg	millipoints / kg	kg 1,4-DCB equivalents /kg	
Cd ♥			2.9	200	3	120000	1890	12.6	6
Cr VI			4100	1 ¹	0.2	1300 ¹	5.4 ^{2,6}	2.03	6
Cu ♥			0.02	2	0.005	1900	11.5	0.515	6
Pb ♥			0.79	2	1	2900	0.576	6.29	6
Hg ♥			4.7	500	10	120000	15.4	122	6
As			1.4	0.2	1		1710	153	5
Ni			0.057	0.33		1900	11.2 ⁶	48.9	5
Zn ♥			0.0029	0.38		520	1.27	0.251	5
Mo			0.29		0.14	19000		126	4
Nitrate ♥		204	0.00078			27			3
Ba			0.14		0.14			17.1	3
Co			2			3800		64.4	3
Phosphate		246700			1 ³				2
Sb					2			413	2
F			0.041					9.43	2
Tl						96000		6680	2
Sn			0.0014					0.00887	2
B					0.03				1
Br			0.0029						1
Cl ♥		26							1
Fe ♥			0.0036 ⁴						1
Sulfate ♥		76.6							1
Be								464	1
Mn					0.02				1
Se								619	1
V								312	1
COD						5.9 ⁵			1
DOC						18 ⁵			1
TOC ♥						18 ⁵			1

- ♥ Elements fully inventoried in the landfill models of the former ETH inventories (Frischknecht et al. 1996:F)
- 1 characterised as Cr
- 2 For Ecotox characterised as Cr
- 3 For eutrophication
- 4 0.0057 for iron oxides
- 5 Only to surface water
- 6 Corrected by Pre Consultants in August 2003

12.9.1 Uncertainty in landfill emissions

The uncertainty of the landfill emissions is influenced by two factors:

1. The uncertainty of the **waste composition** and
2. The uncertainty of the **landfill model** (transfer coefficients).

The uncertainty in waste composition is often more influential than the uncertainty in landfill development as expressed by transfer coefficients (see for example the results of municipal waste incineration in part II). If one were to argue that long-term emissions have to be neglected because of the associated large uncertainty in landfill development, then consequently some of the more uncertain waste components would have to be excluded from analysis, too. However, it is often the (uncertain) trace elements (e.g. copper, cadmium) which make up a dominant share of the total burden from disposal¹¹². Neglecting these emissions on the grounds of associated uncertainty is against the goals of LCA, as a dominant (but uncertain) part of the life-cycle burden would be obscured. Waste disposal processes and especially landfills must be seen as inherently highly variable processes. This is not a result of poor inventory data, but of system characteristics.

¹¹² For example the single, most important exchange in the LCA of average municipal solid waste incineration (with a share of 46% of the total burden) is the long-term emission of copper from landfilled incineration remains (valuation with Eco-indicator'99 HA). The uncertainty of this emission is almost entirely determined by the uncertainty of waste material composition and not by the uncertainty of the landfill transfer coefficient.

13 Conclusions

The elaborate landfill models developed in this report give reasonable results and are able to differentiate between different landfill types. The inclusion of a long modelling time horizon diminishes the risk of neglecting relevant burdens of the life cycle of products. Care needs to be taken to use Impact Assessment methods that do perceive landfill emissions. The extension to 41 heeded chemical elements in the models has proven to be relevant, i.e. in earlier studies relevant gaps in coverage were possible (depending on waste composition). Although the applied models are uncertain and will remain to be so (because future development is *inherently* uncertain), there is no reason to exclude these burdens from life cycle assessments. Any neglect of long-term emissions in product comparisons studies bears the danger to favour the product that has its burdens in the future. Such choices can then hardly be called 'sustainable' regarding inter-generational equity. The results should be taken as indicative of realistic pollution potentials, based on current knowledge and available data. Uncertainty ranges give an estimate of the associated uncertainty.

13.1 Further work

13.1.1 Tailings and overburden waste

The focus of this report is on controlled Swiss landfills (sanitary landfill, slag compartment, residual material landfill, inert material landfill). An issue not explicitly covered by these models are overburden and tailings from mining of coal, metal and other resources.

In the ecoinvent database, tailings are produced in various processes:

v1.0 (2003) – v2.0 (2008):	- Redmud from alumina production
	- Residues from production of dichromate and titanium dioxide
	- Salt tailings from potash mining
	- Phosphogypsum stacks
New in v2.1 (2009):	- Sulfidic tailings stacks from non-iron metal mining
	- Uranium mill tailings
Expected for v2.2 (~2010)	- Hard coal tailings

In ecoinvent v1.0 (2003) the indicated tailings were assessed with the residual landfill model as a proxy. Emissions from sulfidic tailings and uranium mill tailings¹¹³ are now assessed with new inventory models documented in the respective ecoinvent reports for metals (No. 10) and nuclear energy (No. 6, part VII), see also (Doka 2008) and (Doka 2009). These are dedicated models similar to those for landfills, to complement the LCIs that formerly only contained land-use and no or an inadequate vector of emissions. For 2010 a dedicated inventory model for hard coal tailings is planned.

13.1.2 Modelling of individual substances

For the waste compositions usually only chemical elements are available. In some cases also data for single compounds like dioxins, PAH etc. is available. In the current model those compounds are only heeded as carbon, hydrogen, oxygen and chlorine, which underestimates the toxicity of those

¹¹³ *Radioactive* emissions of uranium tailings are assessed in the ecoinvent database since its early beginnings in 1993. The extension made in 2009 includes the modelling of *non-radioactive* emissions.

compounds. A possible extension of the landfill models could be heeding the fate of such chemical substances. Such models were created for LCA purposes e.g. in (Nielsen & Hauschild 1998), but require a lot of a priori substance-specific parameters as data input.

13.1.3 Inert material landfill model

Inert material landfills are modelled here without any direct emissions from waste, assuming a low pollutant potential and chemical inertia. Measurements of inert material landfill leachate could be used to verify this statement and compare the emission potential of inert material landfills to other landfill types. Results from 'inert' materials in residual landfills suggest that the pollution potential of inert materials is relevant¹¹⁴.

13.1.4 Landfill model as an LCIA model?

It has become apparent in the inventories presented here that the calculations of *chemical fate in landfill models* resemble the calculations of *substance fate in LCIA*¹¹⁵. A difference here is that the landfills are considered a part of the technosphere, while LCIA fate calculations are chiefly concerned with the biosphere¹¹⁶. As the LCIA models become more sophisticated, the 'landfill environment' could be attempted to be modelled using the methods similar to the ones used for LCIA fate calculations of substances in soil. Especially the long-term behaviour of landfills might be calculated with parameter variations from soil models¹¹⁷. If this is a suitable procedure, pollutants could be inventoried as 'emitted to landfill', and the generic fate would be taken care of in LCIA.

Obstacles of this approach are probably correct modelling of the buffer consumption in a landfill, heeding degradability of different waste fractions and modelling of gas emissions in the anaerobic methane phase of sanitary landfills. Also with the usually generic valuation in LCIA it becomes more difficult to account for different local conditions (climate, sub-soil etc.), while on an LCI level those local differences are comparatively easy to consider.

However, a cross-examination of models might be fruitful.

13.1.5 Further improvements

The landfill models were created using field measurements wherever possible. However for some elements not all necessary information was available and had to be extrapolated from other landfill types or from chemical elements with behaviour presumed to be similar (see e.g. section 'Application of the model for short-term behaviour' on page 21ff.). In future versions these data gaps should be eliminated.

¹¹⁴ For example cement which used for solidification in *residual landfills* has an environmental burden that is a factor 6 higher than landfilling of wastes in an *inert material landfill* (valuation with Eco-indicator'99 HA). About 85% of that increase is due to landfill leachate emissions, the rest is mainly caused by the larger energy demand in residual material landfills. I.e. accounting for the pollutant content in cement, the burden of its disposal would probably rise by a factor 5 compared to the current inert material landfill model.

¹¹⁵ Apart from landfills, such fate calculations in 'biosphere-resembling' surroundings are performed also in the inventories of agricultural production fertiliser application (Nemecek et al. 2003) and wastewater treatment sludge application to agricultural areas (see part IV of this report). Also heeding the atmospheric conversion of H₂S from uncaptured landfill gas to SO₂ constitutes such a 'fate in LCI' consideration.

¹¹⁶ This is especially true for ecotoxicity calculations. For calculations of human toxicity the technosphere can become relevant again in LCIA, e.g. via drinking water production or fish consumption.

¹¹⁷ E.g. adapting for pH, organic carbon content, loam content etc.

14 Calculation manual

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

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

14.1 For waste to sanitary or residual landfill

Setting up

1. Open the Excel file '13_MSWiV2.xls' that contains the waste definitions.
2. Open the file '13_MSWLFv2.xls' (acronym for 'municipal solid waste landfill') to calculate inventories for wastes to that type of landfill. Inventories of wastes to residual landfills are included in '13_MSWiV2.xls'.
3. Do not update links to other sheets. There is no need. Choose 'No' in the dialog.
4. On slower machines you might want to disable automatic calculation. Choose the menu command 'Extras/Preferences', choose 'calculation' and set the calculation to 'manual'.
5. In '13_MSWiV2.xls' go to the sheet 'waste input'. If you want to calculate inventories of sanitary landfills set the 'disposal code' in cell B4 to '**R**' (the 'R' signifying the German 'Reaktordeponie'). If you want to calculate inventories of residual material landfills set the 'disposal code' in cell B4 to '**U**' (the 'U' standing for 'user-defined waste directly to residual material landfill', instead of residual waste from MSWI). The code will set the residual landfill calculation sheet to use the correct residual material as an input.
6. Go to the waste composition database (located on the same sheet 'waste input') starting at cell K9. Examine the waste compositions listed there. If you find something suitable for you go to 'Choosing the waste fractions' further below. If not, you need to define a new waste composition.

Enter a new waste fraction

7. Find an empty column in the waste composition database in the sheet 'waste input'. **You can only enter data in the white cells.** You can also overwrite existing columns. In case you need to insert new, additional columns, make sure the functionality of the formulas in columns H and I is retained, i.e. the SUMPRODUCTs must refer the *complete* database area. To do that, enter new columns *starting on the left* of the last column in the database (then copy the contents/formulas of an existing data column, to retain sheet functionality).
8. In a suitable column, enter your data for the waste. If you have waste to sanitary landfill that is heterogeneous in degradability, e.g. plastic compounded with paper, you need to divide the waste up into an readily degradable part (e.g. paper) and a robust part (e.g. plastic). Degradability for the first 100 years (*D*) is entered in row 65. More information on degradability of different materials can be found in chapter 6.1.1 'Waste-specific degradability in sanitary landfills' on page 43.
9. Enter a name for the waste in row 15 of the free column.
10. Enter heating values, composition data and other characteristics in rows 18 to 65 of the free column. Use the correct units (MJ/kg and kg/kg). *All figures are per kilogram wet waste.* More information and guidance in chapter 5.7 'Necessary waste-specific data' on page 39. An error message appears in row 14, if the entered composition is not reasonably close to 100%. Please note that unnecessary rounding to significant digits at this stage can lead to 'mass gaps', resulting in an error message. You can skip the parameter for iron recycling, since there is no iron recycling

in sanitary or residual landfills. You might want to fill it in anyway, if you want to calculate the disposal of the same waste in a MSWI.

11. For waste to residual landfill you can specify if the waste is solidified with cement prior to landfilling in row 66 (enter 1 for 'yes' or zero for 'no'). More on solidification in chapter 8.3.1 'Cement consumption for solidification' on page 72.

12. Save the sheet.

Choosing the waste fraction(s)

13. In row 12 of the sheet 'waste input' you can enter percentages for the waste fractions you want to be present in your waste. These shares make up the final waste you inventorise. Usually you just select one column, i.e. one waste fraction with 100%. You can enter any combination of fractions as long as the sum total of all fractions is 100%¹¹⁸. The functional unit is fixed to 1 kg waste to landfill. Check if the degradability of the waste over 100a in row 65 is entered correctly (possible between 100% and 0%, usually between 35% and 1%).

14. Go to cell E15. Enter an identifying name in English for the waste you just defined. In cell E16 you can enter a German name. These names will be used to generate module names in the sheets 'X-Process' and 'X-Exchange'. With the restriction that module names can't be longer than 80 characters, and that the landfill disposal must have the format 'disposal, WASTE_XY, WW% water, to ZZ landfill' (German 'Entsorgung, WASTE_XY, WW% Wasser, in ZZdeponie'), the identifier name should not be longer than 29 to 31 characters (depending on WW%).

15. Force Excel to recalculate the workbook(s) by pressing 'Alt = ' on Windows machines or '⌘ = ' on Mac machines or by choosing the menu command 'Extras/Preferences', 'calculation' and clicking on 'Calculate now'. Important: wait until calculation is complete. Excel's status bar comment in the lower left corner of your screen shows you the progress of the calculation.

16. For disposal of your waste in a *sanitary landfill*: Go to the sheets 'X-Process' and 'X-Exchange' in '13_MSWLFv2.xls' from where you can copy the inventory data in ECOSPOLD format. Paste the tables *as values* to another sheet (press '⌘-Shift-V' and choose 'values').

17. For disposal of your waste in a *residual landfill*: Go to the *far right side* of the sheets 'X-Process' and 'X-Exchange' *in '13_MSWIv2.xls'* from where you can copy the inventory data in ECOSPOLD format. Paste the tables *as values* to another sheet (press '⌘-Shift-V' and choose 'values').

¹¹⁸ This scheme allows to inventorise degradable and robust materials, which have different emissions, in one single exchange module.

Abbreviations

2+, 3+ etc.	Arabic numerals given with chemical elements or compounds refer to their ionic charge (not to their oxidation state), e.g. NH_4^+ is ammonia, a monovalent cation; Cl^- is chloride, a monovalent anion.
a	annum = year, used in expressions like "kg/a" for "kilogram per year" or "100a" for "100 years".
AOX	adsorbable organic halogenides (adsorbable organic halogenated hydrocarbons). A summary parameter for a group of substances that includes such diverse substances as dioxin (2,3,7,8-tetrachlorodibenzo-p-dioxin), pentachlorobenzene, 3,4-chlorobenzaldehyde, or dichloromethane.
GSD	(also SDg) Geometric Standard Deviation. Uncertainty measure for lognormally distributed values. The GSD cannot be smaller than 1 (i.e. 100%). The squared GSD value (GSD^2) can be used to calculate 95% confidence intervals around the mean. More information on the uncertainty calculations in ecoinvent 2000 can be found in the methodology report (Frischknecht et al. 2003a).
I, II etc.	Roman numerals given with chemical elements refer to their oxidation state (not to their ionic charge), e.g. Cr^{VI} is chromium in the oxidation state 6.
IMLF	Inert material landfill. Inorganic landfill for largely unpolluted materials like construction debris or clean excavation material.
kWh	Kilowatt-hours, = 3.6 megajoule.
L/S	liquid (extraction agent) to solid (extractee) ratio in extraction tests.
LCI	Life Cycle Inventory: list of environmentally relevant exchanges crossing the system boundary of the assessed system or process. Result of the system data collection step in an LCA.
LCIA	Life Cycle Impact Assessment aims to examine the product system from an environmental perspective using impact categories and category indicators derived from the LCI results.
LFG	Landfill gas. Gas created from sanitary landfills.
MLV	Maximum Likelihood Value. Value with the peak probability in a triangular distribution.
MSW	Municipal Solid Waste (Swiss expression: Kehricht, German expression: Haushaltabfall / Hausmüll,)
MSWI	Municipal Solid Waste Incinerator (Swiss expression: KVA Kehrichtverbrennungsanlage, German expression: MVA Müllverbrennungsanlage)
MSWLF	municipal solid waste landfill, sanitary landfill (Swiss expression: Rekatordeponie, German expression: Hausmülldeponie)
n.a.	Not applicable (for Pedigree scores) or not available (for waste composition data).
RMLF	Residual material landfills. Inorganic landfill for polluted materials like incineration ashes.
SC	Slag compartment. Compartment within a sanitary landfill for slag (bottom ash) from municipal solid waste incinerators.
TJ	Terajoule; 1'000'000 megajoule
ZH	Canton of Zürich, Switzerland.

Glossary of terms

Categorisation	Definition of the emission media within ecoinvent 2000 database, e.g. emissions to water can occur to marine water, river water, ground water etc. Not to be confused with → speciation.
Closure	Point of time a filled compartment of a landfill is closed.
Infiltration water	Part of the precipitation water that enters a landfill body and becomes the leachate.
Leachate	Water that leaves a landfill body after flowing through the landfill water pores. Leachate is generated by a part of the precipitation water entering the landfill surface. Pollutants present in the leachate are the ecologically most important source of emissions from landfills.
Oxianion	(or oxo anion). Metals usually form positively charged ions (cations). Certain metals do not naturally occur as cations, they form negatively charged oxides (oxianions). Such elements are tungsten (W), vanadium (V), hexavalent chromium (Cr^{VI}), molybdenum (Mo), boron (B), selenium (Se), antimony (Sb), arsenic (As). The corresponding oxianions are: WO_4^{2-} , HVO_4^{2-} , chromate CrO_4^{2-} or dichromate $\text{Cr}_2\text{O}_7^{2-}$, MoO_4^{2-} , HBO_3^{2-} , HSeO_4^- , (or SeO_3^{2-}), SbO_4^{3-} , HASO_4^{2-} (or AsO_2^-). Oxianions have usually good solubility at high or neutral pH. Mobility of oxianions depends on anion exchange capacity.
Pedigree	Method devised in ecoinvent 2000 to assign uncertainty figures to data in case of incomplete information. Uncertainty figures are coarsely estimated for different exchanges using information (Pedigree scores) on source reliability, completeness, sample size, and match or mismatch regarding technology, geography and time. See Frischknecht et al. 2003a.
Precipitation	Water falling in a land surface, either as rain or snow.
process-specific	Most of the inventoried burdens of waste disposal are dependent on waste composition in this study ('waste-specific' burdens). Some burdens are constant and the same for every waste. These constant burdens are separated into distinct data modules (infrastructure and other process-specific burdens)
Slag compartment	Term adopted from Swiss waste ordinance (TVA 2000) which essentially means 'bottom ash landfill' i.e. a landfill for bottom ash incineration residues of municipal solid waste incinerators. Slag compartments are only realised as a separate compartment of a sanitary landfill. (German 'Schlackekompartiment'). The term slag is generally used for MSWI bottom ash in this report except where noted (e.g. slags from metal producing industries described in part I).
Speciation	Chemical form of a compound, e.g. chromium can be emitted as trivalent Cr^{III} or as hexavalent Cr^{VI} which have different toxicological behaviour. Not to be confused with →categorisation.
Static reserve time	(or static reserve, static reserve index) time the known reserves of a non-renewable, scarce good would last at the current rate of consumption = known reserves divided by the annual consumption.
Waste placement	Point of time a waste is deposited at the landfill site.
waste-specific	In this study, burdens from the disposal of different waste materials are calculated. Wherever possible burdens are allocated to waste components (chemical elements). Hence, the waste composition is a chief input parameter. All data items or data modules that change for different waste materials are called 'waste-specific', i.e. 'dependent on the waste composition'. Burdens that are constant for every waste are called 'process-specific'.

15 Appendix

Remark on displayed vs. significant digits

Figures in the tables of this report often feature several digits. This is not to imply that all the digits are significant or that the data displayed is very precise (it is mostly not). Showing several digits helps to minimise the avoidable accumulation of rounding mistakes along the chain of calculations within this study, and in possible future studies referring to the data published in this study. See also chapter 1 in part I.

Tab. A.1 Literature data on municipal waste compositions *m* and leachate compositions *c_o* in sanitary landfills

source	Belevi et al. 1989a	calculated this study part I	This study	Belevi et al. 1989a	Ehrig 1987	Ehrig 1987	Ehrig 1987	NPI 1999	Krumpelbeck 1999	Krumpelbeck 1999	Krumpelbeck 1999	Krumpelbeck 1999	Belevi et al. 1989b:395	Qasim & Chiang 1994	Qasim & Chiang 1994	Qasim & Chiang 1994	SFWM 1987	SFWM 1987	SFWM 1987	Pohland & Harper 1985	Pohland & Harper 1985	Pohland & Harper 1985	This study	
unit	g/kg	g/kg	g/kg	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	
sample type	before methane phase	mean	m	after methane phase	2mo acidic phase and 30a methane phase avg	2mo acidic phase and 30a methane phase Min.	2mo acidic phase and 30a methane phase Max.	generic leachate values average	avg leachate value for various MSF landfills 1 to 5 years after placement	avg leachate value for various MSF landfills 6 to 10 years after placement	avg leachate value for various MSF landfills 11 to 20 years after placement	avg leachate value for various MSF landfills 21 to 30 years after placement	leachate after 9.4 years	min	max	avg	min	max	avg	min	max	avg	c _o	
data type	mean	mean	m						mean	mean	mean	mean	mean	min	max	avg	min	max	avg	min	max	avg	c _o	
O		257.063604	257.0636																					
H		48.2504081	48.25041																					
C	290	334.229989	311.3305	650					1235	845	520	1120	1000										856.596703	
DOC																							1624.80768	
BSB					252.420091	42.283105	772.83105		2285	1210	465	290		80	28000	1496.66295				4	57700	480.416486	754.000092	
CSB					3122.55708	533.424658	4834.24858		3810	3255	1830	1225		400	40000	4000	530	3000	1260.95202		31	71700	1490.87223	2391.36
TOC/CSB									0.324146982	0.259600614	0.284153005	0.914285714											0.38452136	
S		1.11902494	1.119025		2.7	41.3926941	5.19497717	214.874429	49	73	46.25	41.5	2.7										40.0032651	
s2									5.6	6.1	2	1.1											2.94432086	
N	4	3.12384609	3.534881	1200	1250	50	5000						1200										852.398158	
NH4-N					750	30	3000	163.333333															115.797324	
Norg					600	10	4250	137.5						43.5555556	374.888889	127.782995	7.31111111	1042.22222	87.2914799	1.55555556	801.111111	35.3011733	243.33844	
NO3-N					3	0.1	50																2.46621207	
NO2-N					0.5		25																3.53553391	
P	1	0.89379254	0.945406	6.8	6	0.1	30	7.5					6.8	8	35	16.7332005	1.5	130	13.96424	0.2	120	4.89897949	7.32426127	
B		0.00719327	0.007193						5.9	6	5.6	9.1											6.5171568	
Cl	7.2	6.8661886	7.031114	1300	2100	100	5000	590					1300	70	1330	305.122926	112	2360	514.120608	30	5000	387.298335	650.198002	
Br		0.01355198	0.013552																					
F		0.05635809	0.056358	0.65				0.39					0.65										0.54823123	
I		0.0000121	1.21E-05																					
Ag		0.000714	0.000714																					
As		0.00062521	0.000625		0.16	0.005	1.6	0.014															0.01534512	
Ba		0.14903802	0.149038																					
Cd	0.011	0.01174756	0.011369	0.002	0.006	0.0005	0.14	0.014					0.002	0.025	0.025	0.025					70	3900	522.494019	0.06896908
Co		0.00134527	0.001345		0.055	0.004	0.95																0.05934472	
Cr		0.31521067	0.315211		0.3	0.03	1.6	0.06															0.17219794	
Cu	0.4	1.21275952	0.696494	0.1	0.08	0.004	1.4	0.054															0.08541204	
Hg		0.00144239	0.001442		0.01	0.0002	0.05	0.0006															0.00152982	
Mn		0.25935991	0.25936		0.83926941	0.03167123	45.3621005																3.03604755	
Mo		0.00195506	0.001955																					
Ni		0.10738252	0.107383		0.2	0.02	2.05	0.17															0.19610062	
Pb	0.4	0.5024304	0.448299	0.07	0.09	0.008	1.02	0.063					0.00008	0.5	1	0.70710678		0.105		0.001	1.44	0.03794733	0.07887093	
Sb		0.02256364	0.022564					0.066															0.066	
Se		0.00031969	0.00032																					
Sn		0.07343994	0.07344																					
V		0.00921466	0.009215																					
Zn	1.2	1.31124686	1.254391	0.6	0.62785388	0.03055708	4.66849315	0.68					0.6	0.1	30	1.73205081					0.06	220	3.63318042	1.11986779
Be								0.0048															0.0048	
Sc																								
Sr					2.52785388	0.30278539	7.08356164																1.75676169	
Ti																								
W																								
Si		48.5097277	48.50973																					
Fe	50	29.996336	38.72747	8	19.3452055	3.11141553	291.69863						8	0.6	325	13.96424	1.8	22	6.29285309	4	2200	93.8083152	20.4976366	
Ca		14.0621081	14.06211		66.6849315	20.0557078	613.926941																160.138485	
Al		12.4195645	12.41956																					
K		2.05969662	2.059697																				218.460933	
Mg		3.37766484	3.377665		182.618265	40.2785388	356.406393																208.814459	
Na		5.14385611	5.143856		1350	50	4000																537.631607	

Tab. A.2 Literature data on compositions of MSWI bottom ash *m* in slag compartments or bottom ash landfills.

source unit	Stark 1993 g/kg	Stark 1993 g/kg	Goetz 1989 g/kg	Reimann et al.1989 g/kg	Reimann et al.1989 g/kg	EKESA 1992 g/kg	EKESA 1992 g/kg	EKESA 1992 g/kg	Rey 1992 g/kg	AIB 1993 g/kg	Anthonissen et al. 1993 g/kg	Kraxner et al. 2001:55 g/kg	Kraxner et al. 2001:55 g/kg	Kraxner et al. 2001:55 g/kg	Kraxner et al. 2001:55 g/kg
data type	min	max	mean	mean	mean	min	max	mean	mean	mean	mean	mean	mean	mean	mean
sample type				Bamberg	Bamberg	Winterthur	Winterthur	Hagenholz	Hagenholz	Basel	Niederlande	MSWI slag IGLO new	MSWI slag IGLO, aged 3 years in parking lot	MSWI slag RAU-S2 new	MSWI slag RAU-S2 aged 12 years in landfill
H2O															
O												338.068858	288.523401	402.125081	331.425955
H															
C															
S						4	10	3.5	6.4	3.5		6	4.24	12	7.48
N						0.2	0.6			0.7					
P						3	5	0.5	2.4			15.2297168	9.42584191	1.39642102	4.71292095
B												0.416	0.547	0.629	0.331
Cl	1	6		3.51	1.97	9	12	4	2.7	2	2.96				
Br															
F				0.22	0.22	0.3	0.6	0.45	0.8		0.393				
I															
Ag															
As											0.006				
Ba											1.509	2.233	2.733	1.164	1.419
Cd	0.03	0.03	0.01	0.004	0.0038	0.05	0.06	0.022	0.006	0.01	0.004	0.015	0.012	0.017	0.066
Co												0.052	0.045	0.027	0.024
Cr	1	10	0.63	0.66	0.66	0.7	1.1	0.55	0.45		0.125	0.598	0.621	0.513	0.369
Cu	1	4	3	1.52	1.52	1.2	2.4	2.3	1.3	2.5	2.05	5.533	7.702	1.754	4.257
Hg	0.0001	0.0001		0.00072	0.0007			0.00003	0.0001	0.002	0.1				
Mn						0.6	1	0.5	0.7			0.77445093	1.23912149	0.92934111	0.69700584
Mo											0.013	0.027	0.025	0.024	0.024
Ni	0.3	0.3		0.26	0.26	0.1	0.3	0.13	0.13		0.104	0.177	0.199	0.079	0.284
Pb	1	17	4	1.01	1.01	1.8	2.3	1.8	2.3	2	1.248	0.915	0.994	0.743	2.096
Sb												0.102	0.109	0.021	0.071
Se															
Sn								0.1	0.2		0.071				
V											0.045	0.06	0.057	0.102	0.071
Zn	4	15	4	4.57	4.57	4.9	5.9	3.3	3	2.5	1.804	3.141	4.596	4.012	3.548
Be															
Sc															
Sr															
Ti															
Tl															
W															
Si	140	140				210	230	205	227			155.794425	124.429871	190.244017	166.77903
Fe	40	230	117			60	75	50	38.5			78.4759243	85.6100992	72.3209498	63.787917
Ca	25	100	144			110	120	155	121			106.131954	97.1982882	151.514979	118.35321
Al	20	180				51	58	55	53			39.4813785	38.5816688	57.7931171	36.7293253
K	5	20				11	13	12	10			12.7843998	9.96186998	17.4332725	9.71282323
Mg	4	18				17	19	20	16.8			15.6201826	12.7253225	10.8557253	12.5443937
Na	10	60	70			16	25	8	23.7			21.7387097	21.6645161	33.3870968	18.1774194

Tab. A.2 (cont.)

source unit	Faultstich 1993 g/kg	Faultstich 1993 g/kg	Faultstich 1993 g/kg	Huber et al. 1996:24 g/kg	Huber et al. 1996:24 g/kg	Huber et al. 1996:24 g/kg	Schachermerer et al. 1994 g/kg	Huber et al. 1996:24 g/kg	Huber et al. 1996:46 g/kg	Zeltner & Lichtensteiger 2002 g/kg	Zeltner & Lichtensteiger 2002 g/kg	Zeltner & Lichtensteiger 2002 g/kg	Schweizer 1999:14 g/kg	This Study g/kg	
data type	min	mean	max	min	max	geomean	mean	mean arithmetic mean of 2 RFA measurements	mean Slag from MSWI Spittelau A in April 1996	mean	min	max			
sample type	MSWI slag	MSWI slag	MSWI slag							bottom ash of Hagenholz Zürich	bottom ash of Hagenholz Zürich	bottom ash of Hagenholz Zürich	MSWI slag	m	
H2O															
O		508								350	310	390		359.57252	
H															
C	5	28	50	15	40	24.4948974	16			8	7.7	8.3	18	15.78396	
S	2	3.3	4.1	2	4	2.82842712	8.5		7.5	4	3.2	4.8	2.88	4.48857	
N														0.43795	
P	3.46	13.7	33.7	7	14	9.89949494	5.3	2.15	5.67296041	3	2.4	3.6	0.3097	4.22203	
B														0.46654	
Cl	0.27	2.8	6.3	3	6	4.24264069	3.2			2	4	3	5	3.19791	
Br															
F	0.02	0.31	1.1											0.31078	
I															
Ag															
As									0.05	0.01	0.003	0.017		0.01089	
Ba									1.8					1.73608	
Cd	0.0001	0.021	0.079	0.1	0.3	0.17320508	0.0038		0.019	0.005	0.0044	0.0056	0.02248	0.01516	
Co									0.025					0.03279	
Cr	0.15	1.2	9.6	1	10	3.16227766			0.349	1.2	0.85	1.55	0.52	0.90013	
Cu	0.4	2.1	7	1	4		1.934		2.28	8.3	6.3	10.3	2.28744	2.55462	
Hg	0.00007	0.0007	0.002	0.0001	0.007	0.00083666	0.00056		0.0009	0.0004	0.0002	0.0006		0.00052	
Mn				5	20	10		1.15	0.57	1	0.8	1.2		1.31723	
Mo														0.02191	
Ni	0.039	0.21	0.76	0.1	0.3	0.17320508			0.068	0.96		1.92		0.20832	
Pb	0.33	2	5.2	1	17	4.12310563	1.973		1.73	1.7	1.2	2.2	1.781834	1.8862	
Sb									0.062					0.06344	
Se															
Sn	0.2	0.45	1.7						0.13	0.03	0.005	0.055		0.10883	
V									0.025					0.05506	
Zn	0.53	4.7	21	4	15	7.74596669	1.953		3.736	4	3.2	4.8	3.2685	4.1458	
Be									0.012					0.012	
Sc															
Sr															
Ti								4.2	5.39459189	8	7.5	8.5		6.49465	
W															
Si	142.5	200.1	320	170	215	191.180543			213.5	207.538928	190	182	198	207.8364	188.11202
Fe	37.1	59.6	86	40	230	95.9166305	50.5		37.95		180	150	210	44.6784	76.86582
Ca	33.2	76.8	134	25	100	50			107.5	137.221113	120	112	128	148.296	94.38923
Al	4.6	54.3	92.8	80	180	120			29.55		85	70	100	62.05745	55.57287
K	2.9	9.6	21.4						5.2	9	8	7.6	8.4	0.39102	8.15195
Mg	3.9	9.4	18.2	6	18	10.3923048			17.2	18.6959714	15	13.5	16.5	7.2936	12.45018
Na	5	23.4	31.1	10	60	24.4948974			15.5	27	33	31	35	9.2	21.89261

Tab. A.3 Literature data on leachate compositions of MSWI bottom ash c_0 in slag compartments or bottom ash landfills

source unit	Zimmermann et al. 1996:B.159 mg/l	Zimmermann et al. 1996:B.159 mg/l	Zimmermann et al. 1996:B.159 mg/l	Zimmermann et al. 1996:B.159 mg/l	AIB 1993 mg/l	EKESA 1992 mg/l	Lechner 2001 mg/l	Lechner 2001 mg/l	Lechner 2001 mg/l	BLU 1983 mg/l	BLU 1983 mg/l	BLU 1983 mg/l	Leuchs 1990 mg/l
data type	mean	mean	mean	mean	mean	mean	Min	Max	geomean	min	max	geomean	mean
sample type	Weiningen 1970 - 74	Mülheim 77 - 78	1A Dänemark 1973 - 91	Grossmehring 1978 - 82		Phase 1							
H2O													
O													
H													
C										10	44	20.976177	
S	655	117.5	1900	100	256		35	2900	318.590646	35	650	150.831031	99.5
N	126.903226		16.3333333		62.8243728					18.2352941	185.294118	57.2711622	44.5294118
P													
B													
Cl	6580	3235	6100	15400	9779		290	19600	2384.11409	6500	20000	11401.7543	15625
Br													
F											1.3		0.75
I													
Ag													
As			0.01				0.0034	0.025	0.00921954				
Ba													
Cd					0.035		0.00005	0.15	0.00273861	0.01	0.15	0.03872983	
Co													
Cr							0.00005	0.1	0.00223607	0.025	0.1	0.05	0.04
Cu		0.01	0.03	0.06	0.06		0.00025	0.46	0.01072381	0.025	0.3	0.08660254	0.1
Hg				0.03	0.018		0.000025	0.106	0.00162788				0.1
Mn	0.16	0.87					0.02	0.06	0.03464102				
Mo													
Ni				0.12	0.155		0.01	0.6	0.07745967	0.025	0.6	0.12247449	
Pb		0.02	0.01	0.22	0.2		0.00005	0.9	0.0067082	0.025	0.9	0.15	0.27
Sb													
Se													
Sn													
V													
Zn		0.18	0.09	0.07	0.12		0.005	2.5	0.1118034	0.025	0.3	0.08660254	0.14
Be													
Sc													
Sr													
Ti													
W													
Si		2.7											
Fe	0.6	0.17	0.12		0.395		0.24	17	2.01990099				
Ca	63	390	404	600			92	4300	628.967408	0.4	1400	23.6643191	799
Al			0.05										
K	1970	1320	2000	6300			100	10000	1000	10	10000	316.227766	5763
Mg	60		6	40			9	80	26.8328157	100	80	89.4427191	39
Na	6170	916	4600	4900			300	9800	1714.64282	2200	8200	4247.35212	4848

15. Appendix

Tab. A.3 (cont.)

source unit	Kraxner et al. 2001:84 mg/l	Kraxner et al. 2001:84 mg/l	Johnson et al. 1999 mg/l	Johnson et al. 1999 mg/l	Johnson et al. 1999 mg/l	Reichelt & Pfrang 1998 mg/l (large particles)	Reichelt & Pfrang 1998 mg/l (small particles)	Reichelt & Pfrang 1998 mg/l (mean (size 5-8 mm))	Kersten et al. 1998 mg/l	Eggenberger & Mäder 2002:13f. mg/l	Sabbas et al. 1998:26 mg/l	This Study mg/l
data type	mean	mean	mean	max	min					1 sample		
sample type	lysimeter leachate with L/S = 1.6 from MSWI slag IGLO aged 3 years as parking lot	lysimeter leachate with L/S = 0.91 from MSWI slag RAU-S2 aged 12 years in landfill	leachate slag landfill Lorstorf	leachate slag landfill Lorstorf	leachate slag landfill Lorstorf	MSWI slag	MSWI slag	MSWI slag		TVA leachate test with water of de-scraped bottom ash of a Zurich MSWI	leachate conc. Of aged slag at pH 8	co
H2O												
O												
H												
C			10.332	44.76	4.404							16.3061
S	164	1063	132.266667	237.866667	58.7733333			400	213.333333			235.15491
N												52.65423
P	0.04	0.13								0.029368421		0.05345
B	0.6	5.1	2.3652	3.726	1.5444							2.10823
Cl	759	976	1669.8363	3651.659	389.983			1300				3662.41036
Br												
F										0.73		0.89284
I												
Ag												
As												0.00941
Ba	0.97	0.041							0.068665		0.2	0.15287
Cd	0.002	0.003	0.00131508	0.0053952	0.00037879					0.003	0.00065	0.00448
Co	0.001	0.003								0.006		0.00262
Cr	0.224	0.009	0.010868	0.01612	0.008528			0.01092		0.013		0.01479
Cu	0.37	0.056	0.1010286	0.4790916	0.00686232					0.09	0.1	0.05095
Hg												0.00784
Mn	0.003	0.26	0.00468621	0.01801966	0.00154376					0.5		0.03873
Mo	0.46	0.51	0.5219136	1.314378	0.1448694				2.01474			0.60068
Ni	0.009	0.089								0.035		0.07536
Pb	0.009	0.005	0.00269347	0.00700302	0.00052833						0.01	0.01978
Sb	0.017	0.026	0.03226375	0.056979	0.0110549							0.02459
Se												
Sn												
V	0.006	0.008	0.02175138	0.04808736	0.01085022						0.001	0.001
Zn	0.044	0.098	0.00567412	0.01052457	0.00285667					2.3	0.07	0.0709
Be												
Sc												
Sr												
Ti						0.13	1	0.35				0.357
Tl										0.018		0.018
Tl												
W			0.11270005	0.2334895	0.03401225							0.09637
Si	1.4	3.2	3.79161	5.841888	2.6007636					23		3.98742
Fe	0.01	0.17								0.7		0.36948
Ca	346	469	329.8584	645.288	123.8472					340		236.60306
Al	3.53	0.09	0.77436905	3.9662805	0.02425637					0.18	4.5	0.41947
K	123	283	461.4036	946.2684	181.04226			170		40		603.18697
Mg	0.8	474	15.219312	26.25696	2.504136					30		27.17357
Na	314	929	1023.5	2097.6	224.71			530		400		1636.87664

Tab. A.4 Literature data on residual material compositions *m* and leachate compositions *c_o* in residual material landfills

source unit data type	Plüss 1993 g/kg	ETH 1992 g/kg	This report in part I g/kg mean	Speiser et al. 2002 g/kg	Schweizer 1999:14 g/kg	This Study g/kg	AIB 1993 mg/l mean	ETH 1992 mg/l mean	Reuter & Schirmer 1988 mg/l mean	Reuter & Schirmer 1988 mg/l mean	BUWAL 1995 mg/l mean	Ludwig & Johnson 1999 mg/l min	Ludwig & Johnson 1999 mg/l max	Ludwig & Johnson 1999 mg/l mean	Baur et al. 1999 mg/l mean	Baur et al. 2001 mg/l mean	This Study mg/l
sample type	solidified residues of 1992	solidified residues of Winterthur 1991	solidified residual material as calculated according to part I of this study	33% boiler ash, 66% APC residue from MSWI in Bavaria, Germany		<i>m</i>	first leachate		25% of first leachate, solidified sample	50% of fourth leachate, solidified sample	Average values from 4 measurements, CO ₂ and water elution	leachate from RMLF Teutal BE, range including strong rain	leachate from RMLF Teutal BE, range including strong rain	leachate from RMLF Teutal BE, range including strong rain	leachate from RMLF Teutal BE		<i>c_o</i>
H ₂ O	302	301.6				-											-
O			4.687828167			4.68783											
H																	
C	11.6		1.13922875		28.8	7.24693		8									8.
S	22	16.31	0.390296457	30.72	41.6	11.23459	7400	406	1400	1507.5	450						1233.3068
N									4.5	6							5.19615
P	2.73	1.63484211	0.106361312		6.194	1.30948	0.5										0.5
B	0.193		0.002487276			0.02191											
Cl	3.92	3.8955	0.116237314	182.7		4.24359	2500	927.666667	2575	2640	938.75				661.90751		1462.77231
Br			0.012020606			0.01202						20					20.
F	0.0575		0.018513632			0.03263			1.1	5.85		1					1.86001
I			2.35188E-07			0.											
Aq			0.000274485			0.00027											
As			0.000281282			0.00028	5.3										5.3
Ba			0.004036259			0.00404											
Cd	0.146	0.2677	0.011531691	0.3	0.48332	0.14556	0.05	0.004	0.0002225	0.00005	0.005	0.001686	0.00281	0.0026976	0.00058774	0.002248	0.00169
Co			0.00020179			0.0002										0.00005893	0.00006
Cr	0.283	0.3645	0.030076174	0.4	0.624	0.23867	3.9	0.3783333	800	1050	0.55						0.37833
Cu	0.469	0.636	0.131869221	1.8	0.88956	0.57523	0.2	0.02366667	0.0475	0.0205	0.0125					0.050832	0.03782
Hg	0.000391	0.08652	0.001158307			0.0034	0.0005	0.0002	0.000125	0.00045							0.00027
Mn	0.167		0.036310387			0.07787										0.00109876	0.0011
Mo			0.000256112			0.00026									0.69153552	2.8782	1.41081
Ni	0.1	0.12845	0.005384463			0.04105		0.00933333	0.01425	0.0075	0.06					0.17613	0.02539
Pb	3.08	2.868	0.28321735	9.1	7.87322	2.82286	0.25	0.00666667	0.007	0.0015	0.025	0.0310785	0.0683727	0.0663008		0.062157	0.02501
Sb			0.013853715			0.01385											
Se			0.000211227			0.00021											
Sn	0.219	0.5955	0.028262062			0.15447					0.005						0.005
V	1.62		0.001013613			0.04052											0.10188
Zn	13.3	12.942	0.97911737	76.8	22.8795	12.42508	4	0.07666667	0.01375	0.01	0.05	0.3269	1.24222	1.17684	1.0768086	1.3074	0.26012
Be																	
Sc																	
Sr																	
Ti	7.2	3.335				4.9002											
Tl																	
W																1.8385	1.8385
Si	91.1	42.596	1.232635769	16.8	162.8988	26.50879	61										61.
Fe	17.7	30.015	0.839419834	11.5	22.3392	10.2757											
Ca	165	181.85	1.203211311	410.7	184.368	77.15248	39.4								3.5992		11.90834
Al	42.4	60.97	1.105978118	7.7	99.83155	18.55229	60				1.475						9.40744
K	8.45	19.26	0.674893695	17.3	7.8204	6.82975	3270		1800	2105							2313.96855
Mg	9.45	12.22	0.203242738	4.9	9.7248	4.07117	5.5		13.8	0.0065							0.79017
Na	10.4	15.83	0.356723688	14.9	18.4	6.94017	8400		1920	2100					3677.7		3340.75039

Tab. A.5 Fraction of the emissions of an element *e* emitted in landfill gas during the methane phase of a sanitary landfill (cf. Eq. 6.2 on page 46ff.)

<i>e</i>	Source/Comment	% <i>gas_e</i>
O	like C	97.100%
H	like C	97.100%
C	Belevi & Baccini 1989a	97.100%
S	based on Belevi & Baccini 1989a ¹	14.900%
N	Belevi & Baccini 1989a	6.440%
P	Belevi & Baccini 1989a	0.000%
B	like P	0.000%
Cl	Belevi & Baccini 1989a	1.380%
Br	like Cl	1.380%
F	Belevi & Baccini 1989a	83.800%
I	like Cl	1.380%
Ag	like Cu	0.029%
As	like Cl	1.380%
Cd	Belevi & Baccini 1989a	0.662%
Cu	Belevi & Baccini 1989a	0.029%
Hg	Belevi & Baccini 1989a	28.600%
Pb	Belevi & Baccini 1989a	0.033%
Zn	Belevi & Baccini 1989a	0.022%
Fe	Belevi & Baccini 1989a	0.025%

The value of iron (0.025%) is adopted for all other metals (Al, Ba, Be, Ca, Co, Cr, K, Mg, Mn, Mo, Na, Ni, Sb, Sc, Se, Si, Sn, Sr, Ti, Tl, V, W).

¹ adjusted from 98.2% (original figure in Belevi & Baccini 1989a) to result in correct average landfill gas and leachate composition from the model.

Tab. A.6 Calculated release factors r_e for average municipal waste (cf. chapter 6.1.2 'Re-precipitation and the release factor' on page 45)

e	Comment	r_e
O	like C	100%
H	like C	100%
C	Corrected to 100% (from a calculated value 109%) to comply with measured carbon conversion rates and waste degradability rate D according to Tab. 6.1 on page 45.	100%
S		43.80%
N ¹		250%
P		5.59%
B ¹		673%
Cl ¹		255%
Br ¹	like Cl	255%
F		45.20%
I ¹	like Cl	255%
Ag	like Cu	0.49%
As		18%
Ba ¹	like Mn	115%
Cd		17.70%
Co		32.20%
Cr		1.14%
Cu		0.49%
Hg		9.59%
Mn ¹		115%
Mo	like Sb (soluble oxianion)	10.50%
Ni		5.82%
Pb		0.59%
Sb		10.50%
Se	like Sb (soluble oxianion)	10.50%
Sn	like Pb	0.59%
V	like Sb (soluble oxianion)	10.50%
Zn		4.74%
Be	like Ni	5.82%
Sc	mean of Ca and Ti (periodic system neighbours)	9.01%
Sr	like Ni	5.82%
Ti	like Al	5%
Tl	like Ni	5.82%
W	like Sb (soluble oxianion)	10.50%
Si	estimated, little relevance	5%
Fe		1.37%
Ca		13%
Al	estimated, little relevance	5%
K		73.10%
Mg		61.70%
Na ¹		414%

Calculations according to Eq. 6.4 on page 47

- 1 Release factors of >100% signify soluble elements, that can be washed off from the waste matrix, even when the waste matrix is still intact and not degraded.

Tab. A.7 Calculated transfer coefficients for slag compartments (cf. chapter 7.1.1 'Transfer coefficients for slag compartments' on page 65)

Element	TK short-term (100a)		TK long-term (cumulated)		
	Mean value	Uncertainty GSD	Minimal value ¹	Mean value	Maximal value
Time frame	100 a		t _e = 22'918 a	t _g = 60'000 a	inf. time
	kg/kg slag		kg/kg slag	kg/kg slag	kg/kg slag
O	0.004363	197.8%	1	1	1
H	0.004363	197.8%	1	1	1
C	0.001798	213.8%	0.4121	1	1
S	0.09119	143.1%	1	1	1
N	0.1888	130.0%	1	1	1
P	0.00002204	293.0%	0.005051	0.01322	1
B	0.007835	187.3%	0.8352	0.8399	1
Cl	0.8638	102.6%	1	1	1
Br	1	100.0%	1	1	1
F	0.004988	195.4%	0.6821	0.9502	1
I	1	100.0%	1	1	1
Ag	0.00003472	284.8%	0.007957	1	1
As	0.001504	217.0%	0.3447	1	1
Ba	0.0001533	258.1%	0.03513	1	1
Cd	0.0005145	236.3%	0.1179	1	1
Co	0.0001391	259.8%	0.03188	1	1
Cr	0.0000286	288.3%	0.006469	0.006572	0.25
Cu	0.00003472	284.8%	0.007957	1	1
Hg	0.0262	165.6%	1	1	1
Mn	0.00005118	277.8%	0.01173	1	1
Mo	0.04659	155.2%	1	1	1
Ni	0.0006297	232.7%	0.1443	1	1
Pb	0.00001825	296.4%	0.004183	1	1
Sb	0.0006743	231.4%	0.1432	0.1454	1
Se	0.0159	174.5%	0.9746	0.9761	1
Sn	0.000016	298.8%	0.003666	0.9944	1
V	0.0004436	239.0%	0.0967	0.09818	1
Zn	0.00002977	287.6%	0.006822	1	1
Be	0.0006297	232.7%	0.1443	1	1
Sc	0.06044	150.5%	1	1	1
Sr	0.0006297	232.7%	0.1443	1	1
Ti	4.824E-06	320.4%	0.001106	0.2999	1
Tl	0.0006297	232.7%	0.1443	1	1
W	0.0159	174.5%	0.9746	0.9761	1
Si	0.0000369	283.7%	0.008456	0.008593	1
Fe	8.367E-06	310.4%	0.001918	0.5202	1
Ca	0.004363	197.8%	1	1	1
Al	0.00001314	302.3%	0.003011	0.8169	1
K	0.1208	138.0%	1	1	1
Mg	0.003799	200.3%	0.8707	1	1
Na	0.122	137.9%	1	1	1

¹ The minimal value is not used in this study (see section 'Uncertainty of transfer coefficients and emissions' on page 36).

Tab. A.8 Calculated transfer coefficients for residual material landfills (cf. chapter 8.1.1 'Transfer coefficients for residual material landfill' on page 70)

Element	TK short-term (100a)		TK long-term (cumulated)		
	Mean value	Uncertainty GSD	Minimal value ¹	Mean value	Maximal value
Time frame	100 a			$t_g = 60'000$ a	inf. time
	kg/kg waste		kg/kg waste	kg/kg waste	kg/kg waste
O	0.0001508	258.4%	0.09051	0.09051	1
H	0.0001508	258.4%	0.09051	0.09051	1
C	0.001079	223.0%	0.6473	0.6473	1
S	0.1073	140.2%	1	1	1
N	0.1888	130.0%	1	1	1
P	0.0003732	242.1%	0.2239	0.2239	1
B	0.007835	187.3%	0.9911	0.9911	1
Cl	0.286	122.5%	1	1	1
Br	0.8033	103.9%	1	1	1
F	0.05419	152.5%	1	1	1
I	1	100.0%	1	1	1
Ag	0.00006426	273.7%	0.03856	0.03856	1
As	1	100.0%	1	1	1
Ba	0.00001379	301.4%	0.008274	0.008274	1
Cd	0.00001133	305.0%	0.0068	0.0068	1
Co	0.0002854	246.9%	0.1712	0.1712	1
Cr	0.06011	150.6%	0.25	0.25	0.25
Cu	0.00006426	273.7%	0.03856	0.03856	1
Hg	0.0000788	270.1%	0.04728	0.04728	1
Mn	0.00001379	301.4%	0.008274	0.008274	1
Mo	0.9954	100.1%	1	1	1
Ni	0.0006044	233.4%	0.3626	0.3626	1
Pb	0.00000866	309.8%	0.005196	0.005196	1
Sb	0.3527	118.8%	1	1	1
Se	0.3527	118.8%	1	1	1
Sn	0.00003163	286.5%	0.01898	0.01898	1
V	0.002454	208.2%	0.7711	0.7711	1
Zn	0.00002046	294.3%	0.01228	0.01228	1
Be	0.0006044	233.4%	0.3626	0.3626	1
Sc	0.1411	135.2%	1	1	1
Sr	0.0006044	233.4%	0.3626	0.3626	1
Ti	0.0004956	237.0%	0.2973	0.2973	1
Tl	0.0006044	233.4%	0.3626	0.3626	1
W	0.3527	118.8%	1	1	1
Si	0.002249	209.8%	1	1	1
Fe	8.367E-06	310.4%	0.00502	0.00502	1
Ca	0.0001508	258.4%	0.09051	0.09051	1
Al	0.0004956	237.0%	0.2973	0.2973	1
K	0.2819	122.8%	1	1	1
Mg	0.0001897	254.3%	0.1138	0.1138	1
Na	0.3753	117.6%	1	1	1

¹ The minimal value is not used in this study (see section 'Uncertainty of transfer coefficients and emissions' on page 36). The minimal value is identical to the mean value, because the carbonate phase lasts longer than the arrival of the next plateau-covering glacial period ($t_g = 60'000a$).

Tab. A.9 Short-term transfer coefficients for coal ash

<i>Element</i>	<i>Short-term transfer coefficients for coal ash</i>	
Al	0.001%	a
As	0.100%	a
B	1.000%	a
Ca	12.247%	a
Cd	0.050%	a
Cr	1.000%	a
Cu	0.010%	a
Mg	0.001%	a
Mo	83.666%	a
Pb	0.010%	a
S	70.711%	a
Sb	0.020%	a
Sc	0.500%	a
Si	0.005%	a
U	0.020%	a
V	0.005%	a
W	1.000%	a
Zn	0.010%	a
Ba	0.100%	b
Be	0.100%	b
Br	0.100%	b
C	0.100%	b
Ce	0.100%	b
Cl	5.000%	b
Co	0.200%	b
Cs	0.100%	b
Eu	0.100%	b
F	5.000%	b
Fe	0.100%	b
Ge	0.100%	b
Hf	0.100%	b
Hg	0.100%	b
I	0.100%	b
K	0.300%	b
La	0.100%	b
Mn	0.100%	b
N	1.000%	b
Na	3.000%	b
Ni	0.100%	b
P	1.000%	b
Rb	0.100%	b
Se	20.000%	b
Sm	0.100%	b
Sr	0.100%	b
Th	0.100%	b
Ti	0.100%	b
Tl	0.100%	b

a = Geometric mean of the range for alkaline ashes from Tab.VI.9.60 in Frischknecht et al. 1996:VI.118.

b = Mean transfer coefficient given in Tab.VI.9.61 in Frischknecht et al. 1996:VI.118.

Tab. A.10 Process-specific burdens for four landfill types

Name	Location	Unit	process-specific burdens, sanitary landfill	process-specific burdens, slag compartment	process-specific burdens, residual material landfill	process-specific burdens, inert material landfill
Location			CH	CH	CH	CH
InfrastructureProcess			0	0	0	0
Unit			kg	kg	kg	kg
diesel, burned in building machine	GLO	MJ	0.04674	0.02696	0.02696	0.02696
electricity, medium voltage, at grid	CH	kWh	0.00135			
electricity, low voltage, at grid	CH	kWh	0.000015	0.00001	0.00005625	0.00001333
light fuel oil, burned in boiler 10kW, non-modulating	CH	MJ	0.00161	0.001074	0.006039	0.001431
Transformation, from pasture and meadow		m ²	0.00005	0.00004444	0.0000625	0.00004444
Occupation, construction site		m ² a	0.00025	0.0002222	0.0003125	0.00004444
Transformation, to dump site, sanitary landfill		m ²	0.00005			
Transformation, to dump site, slag compartment		m ² a		0.00004444		
Transformation, to dump site, residual material landfill		m ²			0.0000625	
Transformation, to dump site, inert material landfill		m ²				0.00004444
Occupation, dump site		m ² a	0.0015	0.001333	0.001875	0.0004444
Transformation, from dump site, sanitary landfill		m ²	0.00005			
Transformation, from dump site, slag compartment		m ²		0.00004444		
Transformation, from dump site, residual material landfill		m ² a			0.0000625	
Transformation, from dump site, inert material landfill		m ²				0.00004444
Transformation, to shrub land, sclerophyllous		m ²	0.00005	0.00004444	0.0000625	0.00004444
Occupation, shrub land, sclerophyllous		m ² a	0.00025	0.0002222	0.0003125	0.0002222
Transformation, from shrub land, sclerophyllous		m ²	0.00005	0.00004444	0.0000625	0.00004444
Transformation, to forest		m ²	0.00005	0.00004444	0.0000625	0.00004444

Uncertainty ranges for these process-specific exchanges are displayed in Tab. 6.8 on page 62.

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