A model for waste-specific and climate-specific life cycle inventories of open dumps and unsanitary landfilling of waste

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1 Introduction

The LCA landfill models in Doka (2003) are based on well-managed landfills in Switzerland and Germany. Here a landfill model is developed that is applicable to a wide range of climates and allows also the calculation of process inventories for uncontrolled landfills (unsanitary landfills) to be used in LCA calculations. As with the previous instalments, the model focuses on the emission of pollutants from specific waste materials and less on management of average waste.

The present tool was developed 2016–2017 by Gabor Doka in a project for Sustainable Recycling Industries SRI, financed by the Swiss State Secretariat for Economic Affairs SECO, commissioned by the Swiss Federal Laboratories for Materials Science and Technology EMPA, mandated by the ecoinvent Association. The reports were reviewed by Ivan Muñoz of 2.-0 LCA consultants.

2 Influence of climate on landfill degradation

Several models exist to predict annual emissions of methane from landfills. Important parameters in such models are the speed of decay and the ultimate convertible amount of decay. Speed of decay is often expressed with a half life constant \( t_{1/2} \) [years] or an exponential rate constant \( k = \ln(2)/t_{1/2} \) [years\(^{-1}\)]. Ultimate amount of decay is expressed with a constant \( L_0 \) [kg CH\(_4\)/ton waste]. Such models can be considered to cover a limited part of a more circumspect landfill model for LCI. Figures of \( L_0 \) were used in (Doka 2003-III) to estimate degradabilities of different waste materials.

2.1 Influence on speed of methanogenic decay

The widely-used IPCC landfill model features a dependency of the speed of decay on the climatic regions, e.g. dry versus wet temperate/boreal and dry versus wet tropical (IPCC 2006-3). These figures suggest a strong influence of water availability on the decay speed, and a relatively smaller effect of the mean (surface) temperature. For average waste in a dry temperate climate the default half-life of methanogenic decay is 14 years, meaning that within a century 99.3% of all decomposable carbon will have been converted.\(^1\) But even for slowly degrading materials, like wood, a half live of 35 years is given, resulting in a 86% decay within 100 years, and 95% within 150 years.\(^2\) Oonk (2010:23) quotes maximal decay half times of 35 years for average waste in arid climates from the Landgem model.

The LCA landfill model in the present study has only a very coarse temporal resolution. It only distinguishes emissions in the short-term (up to 100 years after waste placement) and emissions in the long-term (after 100 years). Thus this model is not intended to replace models with an annual temporal resolution, but to supply life-cycle inventory data on the total circumspect burdens inflicted by landfilling, especially including the long-term time frame and also regarding pollutants and expenditures other than methane. For the purposes of this model, the effects on methanogenic decay speed are not really relevant, since even for slow decay speeds the methanogenic phase of a landfill will be well over or close to over a century after waste placement (the short-term timeframe for the present model). While the speed of decay plays a crucial role in estimating annual landfill emissions, for the LCA landfill model it is of little importance. For the purposes of the current model, all methanogenic emissions are considered short-term emissions, i.e. the negligible to small amount of

\(^1\) From \( 1 - 0.5^{100/14} = 0.993 \)

\(^2\) From \( 1 - 0.5^{150/35} = 0.949 \)
landfill gas theoretically expected to be generated after the short-term period of 100 years ends is nevertheless inventoried as leading to short-term emissions.

2.2 Influence on ultimate amount of methanogenic decay

More important for LCA is the question whether the ultimate amount of methanogenic decay ($L_0$) might be affected by climate. Since methanogenic decay requires humidity, it can be expected that in climates with no precipitation ultimate methanogenic decay is close to zero. A small activity might be possible from humidity drawn up from groundwater or from water already contained in water, which however will quickly dissipate in arid environments. Essentially a dry environment will preserve even potentially highly degradable materials for a long time when buried.\(^3\) Similarly sufficiently low temperature in the landfill will prevent methanogenic conversion.

In the following the effect of climate conditions on the ultimate amount of methanogenic decay is investigated.

2.2.1 Precipitation

Oonk (2010:23) quotes $L_0$ figures based on measurements in California landfills of 44 kg per ton average waste as being representative for a subtropical dry climate. By contrast in moist temperate climates $L_0$ values of 60 kg CH$_4$/ton waste are used (based on measurements in Dutch landfills in the TNO model). The major Californian landfills are near population centres (Spokas et al. 2015) and are located in regions typically receiving only 30-60 mm of precipitation. This evidences the reduced ultimate decay in dry climates.

This information is used here to derive a generic dependence of the decay amount on precipitation. For zero precipitation a zero decay is surmised. For higher values of precipitation above 1000 mm/year the decay amount shall be equal to that of a temperate climate. It is therefore assumed here that precipitation above 1000 mm/year will not fundamentally alter the ultimate amount of methanogenic decay (while they do have effects on decay speed, which as explained above is of little consequence in this model).

The following adjustment of the mean average decay amount of average waste with annual precipitation is employed. The resulting dependency is illustrated in Fig. 2.1.

\[
L_0' = L_0t \cdot \left(1 - e^{-fp \frac{MAP}{MAP_t}}\right)
\]

Eq. 2.1

where,

$L_0'$ Adjusted methanogenic decay amount of average waste for specified climate
$L_0t$ Methanogenic decay amount of average waste in temperate climate, constant = 60 kg CH$_4$/t waste
$fp$ Factor for precipitation-dependence of decay amount, constant = 29.372 [-] \(^4\)
$MAP$ Mean annual precipitation on landfill site [mm/year]
$MAP_t$ Mean annual precipitation in temperate climate, constant = 1000 mm/year

\(^3\) In such climates mechanical wind erosion might ultimately become the most relevant disintegration and emission pathway (Kraxner et al. 2000:17)

\(^4\) The parameter $fp$ was derived from fitting the curve to the Californian data mentioned above. For Californian landfills $L_0'$ is 44 kg CH$_4$/ton waste and mean precipitation is 45 mm/year. For a temperate climate $L_0t$ is 60 kg CH$_4$/ton waste and mean precipitation is 1000 mm/year. Therefore $fp = - \ln (1 - 44/60) \cdot 1000/45 = 29.372$.
2.2.2 Temperature

While temperature has some effect on the speed of methanogenic decay $k$, no literature references on the temperature dependency of the ultimate methanogenic decay amount $L_0$ could be found.

A coarse modelling for temperature dependence is attempted here. Similar to precipitation it is assumed that any excess temperature above a temperate climate will not have any enlarging effect on the ultimate decay amount. As sanitary landfills can reach internal temperatures of up to 120°C due to decomposition heat, any kind of elevated atmospheric temperature is unlikely to have any additional effect.

For lower temperatures the generated decomposition heat will counteract any significant cooling effect. One limit however can be expected when the deep ground is freezing, i.e. permafrost conditions. In such conditions the deposited waste is likely to be already frozen on delivery, cannot heat up in the ground and microbial activity is likely to remain negligible.

Permafrost conditions can occur in polar and boreal climates, cf. the next two figures.
A comparison of permafrost occurrence (Fig. 2.2) with mean annual temperatures (Fig. 4.2) yields that sporadic permafrost occurs roughly around \(-2\) to \(-6\) °C while continuous permafrost occurs roughly around \(-11\) to \(-15\) °C. This means that around \(-2\)° MAT landfill decay might start to get to be affected and that at \(-15\)° it is most likely completely halted. From this information a curve similar to the precipitation dependence is derived here of the form:
Eq. 2.2 \[ L_0' = L_0 \cdot \left(1 - e^{-\beta [MAT + 15]}\right) \]

where,
- \(ft\) Factor for temperature-dependence of decay amount, constant = 0.3 [-]
- \(MAT\) Mean annual temperature on landfill site [°C]. Please note addition of 15°C occurring in the formula.

Fig. 2.4 Employed dependency of mean average decay amount \(L_0'\) on mean annual temperature (MAT) [°C].

The two derived dependencies of decay amount – from precipitation and from temperature – will be combined in the following (see Eq. 2.5 on page 10).

### 2.2.3 Extension to different waste fractions

The present LCA landfill model is expressly created to be able to inventory separate specific waste materials, not only average municipal waste mixtures. In the sections above only formalisms were derived that describe the behaviour of average waste mixture, not single waste materials. In the previous instalments of the sanitary landfill model, specific material degradabilities \(D_0\) were derived for temperate climates. So how can the climatic dependencies derived above for average waste be used to adjust also specific material degradabilities \(D_0\) to different climates?

A few boundary conditions can be stated:
- The adjusted degradabilities \(D'\) must remain within the bracket of 0% o 100%, to fulfil mass conservation.
- It makes sense to assume that the ranking of adjusted degradabilities remains unchanged also in dry or cold climates. A material \(U\), which degrades better than a material \(V\) in a temperate climate, will also very likely degrade better in a more arid or more boreal climate.
- If \(L_0 = L_0'\), i.e. for a temperate climate, all \(D'\) must be \(D_0\).
- If \(L_0'\) is zero, i.e. no methanogenic decay at all, all \(D'\) must be zero.

A function which reflects all these boundary conditions is the following exponential approach:
Eq. 2.3  \[ D' = 1 - e^{\left[ \alpha \cdot \ln(1 - D_0) \right]} \]

where,
- \( D' \)  Climate-adjusted degradability within 100 years in landfill [-]
- \( D_0 \)  Original degradability in temperate climate within 100 years in landfill [-]
- \( \alpha \)  Factor to consistently adjust degradability to a certain climate [-]

This converts any collection of \( D_0 \) values into a corresponding set of \( D' \) values, which fulfil all of the above boundary conditions. The parameter \( \alpha \) adjusts the degradabilities consistently. For \( \alpha = 1 \) all degradabilities are their original \( D_0 \) value for temperate climate. For \( \alpha = 0 \) all degradabilities become zero. The model only operates with values of \( \alpha \) between zero and one to depict the reduced decay amounts in dry and cold climates. Mathematically though, \( \alpha \) can be larger than one. For ever increasing values of \( \alpha \) all degradabilities approach 100% and therefore heed the condition that degradabilities must not become larger than 100%. The effect of \( \alpha \) is shown in the next figure.

Fig. 2.5  Effect of the factor \( \alpha \) on adjusted degradabilities \( D' \).

How can the correct alpha value applicable to a certain climate be obtained? The methanogenic decay amounts \( L_0 \) can be converted into overall degradabilities \( D_0 \) as used in the LCA landfill model by considering how much carbon has been converted to methane and CO\(_2\) and how much carbon is available. Municipal solid waste contains about 33 w% carbon. Landfill gas contains about 50% CO\(_2\) therefore if all carbon in municipal solid waste were decaying, one could expect a methane yield of 220 kg CH\(_4\) per ton of waste. With this information the \( L_0' \) values derived above can be converted to a value for \( \alpha \).

Eq. 2.4  \[ \alpha = \frac{\ln(1 - L_0'/L_0\text{max})}{\ln(1 - L_0/t /L_0\text{max})} \]

where,
- \( L_0' \)  Adjusted methanogenic decay amount for specified climate [kg CH\(_4\)/t waste]
- \( L_0t \)  Methanogenic decay amount in temperate climate, constant 60 kg CH\(_4\)/t waste
- \( L_0\text{max} \)  Maximal methanogenic decay amount possible, constant 220 kg CH\(_4\)/t waste
From the two adjusted L0' values (Eq. 2.1 and Eq. 2.2) two alpha values can be derived: \( \alpha_p \) and \( \alpha_t \) (precipitation-adjusted and temperature-adjusted). The model shall heed either impediment to decay amount. This is achieved simply by multiplication:

\[
\text{Eq. 2.5} \quad \alpha = \alpha_p \cdot \alpha_t
\]

The resulting \( \alpha \) parameter can then be used to adjust any of the default temperate waste material degradabilities to a specific climate, using Eq. 2.3.

### 2.2.4 Short-term emissions

The calculation of waste-specific emissions based on a waste's degradability was presented in (Doka 2003-III:48ff). The same model is applied here, but with the climate-adjusted degradabilities derived above. Essentially, emissions are derived from the degraded matter, but heeding re-precipitation within the landfill body. This means that although matter might be mineralised and released from the original waste material as such, it might still not be emitted but retained within the landfill. This was modelled by deriving a so-called release factor \( r_e \), which can moderate the degraded masses (Doka 2003-III:118). This re-precipitation is basically driven by the chemical environment in a landfill. For the present model the assumption was made that the behaviour depicted by the release factor is characteristic for mixed municipal waste deposits and is not fundamentally influenced by climate. Therefore the emitted mass of an element \( e \) present in a waste can be expressed with the following.

\[
\text{Eq. 2.6} \quad \frac{E_e}{m_e} = D' \times r_e
\]

where,

- \( E_e \) Short-term emission of the element \( e \) [kg e/kg waste]
- \( m_e \) Concentration of element \( e \) in waste fraction, [kg e/kg waste]
- \( D' \) Climate-adjusted decomposition rate of waste, [kg/kg in 100a] (see Eq. 2.3)
- \( r_e \) Release factor for element \( e \), [kg/kg]

In the model, this emitted amount is split up into emissions to air or to leachate, using a factor \%gas\(_e\) (Doka 2003-III:117). For complex wastes with dissimilar degradation behaviour of their constituting waste fractions, a corresponding weighted mixture of this formula is employed.

The climate adjustment influence the short-term transfer coefficients as they express the emitted fraction over the initially present fraction, i.e.

\[
\text{Eq. 2.7} \quad \frac{SSTK_e}{m_e} = \frac{E_e}{m_e} = D' \times r_e
\]

where,

- \( SSTK_e \) Waste-specific short-term transfer coefficient for element \( e \) [kg/kg e]

### 2.2.5 Long-term emissions

In (Doka 2003:51) the long-term transfer coefficients for a specific waste were presented. The long-term behaviour, as expressed by long-term transfer coefficients is formulated as follows:
LCI model of unsanitary landfill

2. Influence of climate on landfill degradation

\[ \text{Eq. 2.8} \]

\[ LTTK_e = STTK_e + (TK_e - STTK_e) \times \left( \frac{\phi LTTK_e - \phi STTK_e}{TK_e - \phi STTK_e} \right) \]

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]
- \( TK_e \) Maximal long-term transfer coefficient of the element \( e \), [kg/kg element]
- \( \phi LTTK_e \) Long-term transfer coefficient of the element \( e \) for average waste, [kg/kg element]
- \( \phi STTK_e \) Short-term transfer coefficient of the element \( e \) for average waste, sum of gas+leachate [kg/kg element]

The long-term behaviour is in part dependent on the short-term behaviour of the specific waste (STTK\(_e\)). This means that any climate-adjustments made to degradabilities will also be inherited by the long-term behaviour of this specific waste. This is reasonable: If climate slows down degradation in the short-term, it is very likely that this impediment is also continued. But the long-term behaviour is also depending on the behaviour of the average landfill (last bracket in Eq. 2.8). This expresses basically that long-term behaviour of material not-emitted in the short term from a specific waste will be proportional to the behaviour of material not-emitted in the short term in average municipal solid waste (Doka 2003-III:51 and Fig 6.1).

The latter is derived from the working point model for the average landfill, which for this project is also made dependent on climate.

**The working point behaviour of the average landfill body.**

As we have seen, to judge the long-term emission behaviour of a specific waste, we need information about the long-term behaviour of the average landfill as a whole.

For the short-term behaviour the same climate dependencies for degradabilities as derived above apply, i.e. temperature and infiltration rates can influence degradabilities.

After the short-term the leachate concentrations and therefore emission rates are assumed to be modified to a new level. This is expressed with the reduction factor \( x_s \) (Doka 2003-III:50). This "settling" of the landfill to a less active regime is again mostly dependent on the chemistry of the landfill body. For this reason the reduction factors \( x_s \) derived for Swiss municipal waste landfills are also used in the present model.

At this new level of emission rate, the landfill continues to leach pollutants, until the next change in chemistry occurs: the carbonate buffer can be washed out leading to a drop in pH. At this point leachate concentrations change abruptly for several pollutants: for most cationic heavy metals they increase, for oxianions they decrease, but for easy soluble ones like potassium or chlorine they remain constant. This dynamic is expressed with another parameter \( x_e \) (Doka 2003-III:51). The point in time when the carbonate buffer will be washed out is determined site-specifically. The end of the carbonate phase is expected when all calcium carbonate in the landfill is leached. This time can be calculated based on the initially available calcium in the landfill, the available infiltration water and the calcium concentration in leachate:

\[ \text{Eq. 2.9} \]

\[ t_e = \frac{m_{Ca}}{c_{Ca,L} \times V_{eff}} \] corrected to \( t_e \leq 60'000 \text{ years} \]

where,
- \( t_e \) Time in years when the pH buffer is washed out [yr]
- \( m_{Ca} \) Initial concentration of calcium in average landfilled waste [kg/kg]
- \( c_{Ca,L} \) Concentration of calcium in average leachate [kg/L]
- \( V_{eff} \) Effective annual leachate volume per kilogram of waste [L/kg.yr]
The effective annual leachate volume \( V_{\text{eff}} \) is calculated in the same way as in Eq. 5.20 of (Doka 2003-III:32) with some location-specific adaptations:

- Annual water infiltration rate \( I \)
- Overall landfill height \( h \)
- Permafrost conditions reduce available liquid water. The ratio of \((L0'/L0,\) derived from Eq. 2.2 on page 8, is multiplied with the infiltration rate \( I \) to express this reduction.
- Residence time of preferentally flown water \( T_p \). This parameter is given in years and depends on landfill height, i.e. the duration becomes shorter, if landfill height is smaller. The given number for \( T_p \) is therefore linearly adjusted using the landfill height, i.e. \( T_p \cdot 15/h \), where 15 is the generic landfill height using in the landfill model in (Doka 2003-III:31), to which the original \( T_p \) figure of 0.17 years refers to.

The new formula to calculate the effective leachate volume therefore becomes the following expression.

\[
V_{\text{eff}} = \frac{1 - w\%}{h \cdot \delta \cdot \frac{T_p \cdot \frac{h}{15} \cdot w\%}{I \cdot \alpha_i \cdot v\%}}
\]

Eq. 2.1

where,
- \( w\% \): Share of preferential flow in leachate output (22%)
- \( h \): Landfill height (m)
- \( \delta \): Average waste density (kg/m\(^3\))
- \( I \): Rain infiltration rate (mm/m\(^2\)a)
- \( \alpha_i \): Factor expressing the effect of permafrost conditions (-), cf. chapters 2.2.2 & 2.2.3
- \( T_p \): Residence time of preferentially flown water (0.17a)
- \( v\% \): Water content in average waste (25 w-%)

Therefore the difference in weathering speed of a certain location will influence the point in time when buffer washout is completed, which in turn influences long-term transfer coefficients of the landfill. The end of carbonate phase time \( t_e \) is then used in the calculation of long-term coefficients in the same manner as the earlier model (Eq. 6.9 + 6.10 in Doka 2003-III:50) \(^5\) with the difference that \( t_e \) is now not a constant derived for an average Swiss municipal landfill, but variable according to local conditions.

With this all components are in place to calculate the location-specific and waste-specific emission from a landfill.

### 2.3 Weathering in wet and dry climates

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

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\(^5\) See the erratum to Eq 6.12 of (Doka 2003) in this report page 15.
climates, evaporation at the surface is so strong that an *upward* pressure gradient results, which draws groundwater through the landfill to the surface. On the surface the water evaporates and leaves behind solid, but brittle precipitates from the dissolved elements, called evaporites or efflorescent salts. Evaporites are prone to surface erosion by wind. Wind erosion is a relevant emission pathway in arid climates.

**Fig. 2.6** Conceptual differences in emissions from landfill sites in wet climate and in dry climate.

**Fig. 2.7** As an illustration: evaporites forming on tailings from copper mining at Bahia de Ite, Peru. Photo taken from (Diaby et al. 2006)

For the present landfill model following assumptions are made:

- Net annual infiltration is considered to decide if a landfill site has reversed leachate flow.
- Net annual infiltration is calculated from the mean annual precipitation (MAP) [mm/year] minus the actual annual evapotranspiration (ETa) [mm/year]
- With a negative net annual infiltration, the site is considered to have reversed leachate dynamics ("reversed dry site")
- On such reversed dry sites no leachate collection and treatment can be performed
- On reversed dry sites leachate solids accumulate on top of the site as evaporites
- Of the accumulated evaporites 10% are considered to be eroded by wind and inventoried as emission to air of each chemical element. This percentage can be altered. 10% was already used in the ecoinvent database for uranium tailings in dry locations like Namibia (Doka 2008).
- Of the windblown evaporites the sum mass is also inventoried as particulate matter emission (PM). A conservative profile of 20% PM$_{2.5}$, 30% PM$_{2.5-10}$, and 50% PM$_{>10}$ is assumed (Doka 2008).
- Of the accumulated evaporites 90% are inventoried as emission to industrial soil.

The modelling leads to following emission characteristics of controlled or uncontrolled landfills in wet or dry climates:

<table>
<thead>
<tr>
<th>Wet climate (positive net infiltration)</th>
<th>Controlled landfill: Sanitary landfills (s)</th>
<th>Uncontrolled landfill: Unsanitary landfills (u) and Open dumps (d)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>LFG: possibly captured</td>
<td>LFG: direct emission to air</td>
</tr>
<tr>
<td></td>
<td>ST leachate: treated in WWT and emitted to river</td>
<td>ST leachate: no WWT, emitted to groundwater</td>
</tr>
<tr>
<td></td>
<td>LT leachate: emitted to groundwater</td>
<td>LT leachate: emitted to groundwater</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Dry climate (negative net infiltration)</th>
<th>Controlled landfill: Sanitary landfills (s)</th>
<th>Uncontrolled landfill: Unsanitary landfills (u) and Open dumps (d)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>LFG: possibly captured</td>
<td>LFG: direct emission to air</td>
</tr>
<tr>
<td></td>
<td>ST leachate: no WWT, emitted to soil and air</td>
<td>ST leachate: no WWT, emitted to soil and air</td>
</tr>
<tr>
<td></td>
<td>LT leachate: emitted to soil and air</td>
<td>LT leachate: emitted to soil and air</td>
</tr>
</tbody>
</table>

Abbreviations

- LFG Landfill Gas
- WWT wastewater treatment for leachate
- ST short term (0-100 years)
- LT long-term (101-60'000 years)
- s, u, o Landfill management types, see chapter 5.4 'Landfill management' on page 22.

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

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6 This is in accordance to methodological choices in the ecoinvent database, where particulate emissions to air can be inventoried as PM figures (where mass and size is the dominant aspect) and additionally as toxic compounds contained in the particulates (where toxicity is the relevant aspect).
2.4 Erratum for calculation of average transfer coefficients

The sanitary landfill model of (Doka 2003-III) included a calculation of elemental transfer coefficients for the average landfill behaviour, i.e. the typical behaviour of chemical elements for the average landfill. These transfer coefficients are instrumental in deriving the waste-specific transfer coefficients valid for a specific material with a specific degradability.

For the calculation of the average transfer coefficients, three different stages of the landfill are discerned:

(a) Short term phase 0-100 years after waste placement, dominated by the methanogenic phase
(b) Carbonate phase, from 100 years until the carbonate buffer is washed out (pH drops)
(c) Post-carbonate phase, remaining time after 4500 years until 60'000 years.

Transfer coefficients were calculated for the three points in time:

\[ \text{STTK} = \text{Short-term transfer coefficients for } 0 - 100 \text{ years} \]
\[ \text{TK}(t_e) = \text{Transfer coefficients until pH drop, for } 0 - t_e \text{ years (index } e \text{ stands for "end of carbonate")} \]
\[ \text{TK}(t_g) = \text{Cumulated long-term transfer coefficients, for } 0 - 60'000 \text{ years (index } g \text{ stands for "glacial period starts")} \]

The equation 6.12 in Doka (2003-III:51) gives the cumulated long-term transfer coefficient for elements following a linear leaching dynamic. An error was detected in these equations. The corrected equation is given below. The corresponding equation for elements following a exponential leaching dynamic is correct (Eq. 6.11 in Doka 2003-III)

\[ TK(t_g) = TK(t_e) + \left( \frac{\text{STTK}}{100a} \right) x_s \cdot x_e \cdot (t_g - t_e) \]

Corrected to \( TK(t_g) \leq TK \).

This error affects only the calculation of cumulated long-term transfer coefficients \( TK(t_g) \). The other sets of transfer coefficients STTK and are \( TK(t_e) \) correct. The error was made only in calculating the linear leaching dynamics of the third phase (post-carbonate).

The correction can lead to larger average long-term transfer coefficients \( TK(t_g) \). The consequence of the error correction is however limited. Transfer-coefficients are restricted by a maximal value of 100% they cannot exceed, and in the uncorrected model already many \( TK(t_g) \) were at 100%. Notable exceptions are zinc, which before had a long-term transfer coefficient of 82.6% and after the correction has 100%. Also carbon, which leads only to TOC emissions, is now completely emitted in the long-term, while before its long-term transfer coefficient was 38%. For phosphorus, the transfer coefficient is elevated from 2.18% to 8.14%. These transfer-coefficients are only representing average landfill behaviour, i.e. the typical behaviour of chemical elements for the average landfill, and not the behaviour of a specific waste.
3 Landfill management types

Apart from the influence of climate on landfill decay, the management of the landfill can play a significant role in the environmental effects of a landfill material. Within this study three different classes of landfills are distinguished:

- Sanitary landfill
- Unsanitary landfill
- Open, uncontrolled dump

The **sanitary landfill** is a highly managed and maintained landfill and is inventoried as in (Doka 2003) which includes a bottom liner, waste compaction, landfill gas capture, soil covers and managed renaturation after operation. Leachate treatment is performed when climate allows it (Cf. chapter 2.3 'Weathering in wet and dry climates' on page 12).

The **unsanitary landfill** has only waste compaction and daily soil covers, but no bottom liner, no leachate treatment, and no landfill gas capture. This description is based on (Alfayez 2011) for the majority of landfills in Jordan. Only spontaneous renaturation after operation is assumed. Many other types of sub-managed landfills are imaginable down to open dumps with no management, but this description is used as an intermediate between sanitary landfills and open dumps. The waste compaction and waste cover represents some limited form of management and affects the methane correction factor (see chapter 5.7 'Methane correction factor MCF' on page 22).

The **open dump** is a totally uncontrolled dump and has no waste compaction, no soil covers, no bottom liner, leachate treatment, and no landfill gas capture. Also here spontaneous renaturation after operation is assumed.
Tab. 3.1  Managements characteristics, or lack thereof, in the three different landfill types

<table>
<thead>
<tr>
<th></th>
<th>Sanitary landfill</th>
<th>Unsanitary landfill</th>
<th>Open dump</th>
</tr>
</thead>
<tbody>
<tr>
<td>Excavation pit</td>
<td>√</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Bottom liner</td>
<td>√</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Waste compaction</td>
<td>√</td>
<td>√</td>
<td>–</td>
</tr>
<tr>
<td>LFG collection tubes</td>
<td>√</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Leachate treatment</td>
<td>√†</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Soil cover</td>
<td>√</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>LFG capture</td>
<td>√</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Methane correction factor MCF</td>
<td>1.0</td>
<td>1.0</td>
<td>&lt; 1</td>
</tr>
</tbody>
</table>

† Leachate treatment only in sufficiently wet climates with downward leachate flow (see chapter 2.3 ‘Weathering in wet and dry climates’ on page 12)

Fig. 3.1  Example of an unsanitary landfill near Deir Alla, Jordan (Selmo 2010). Visible is the applied earth cover on the right side, and the seeping, uncollected leachate on the left and insert.

3.1  Sanitary landfill operation phase

3.1.1  Landfill gas pumps

During the short-term, landfill gas can be captured from the landfill. To this purpose landfill gas pumps are employed. An average constant, process-specific electricity demand was attributed to each kilogram of waste (0.00135 kWh/kg, cf. Doka 2003-III:62).

For the present model this electricity demand is made waste-specific. As allocand the mass of carbon in captured landfill gas is used. For the average Swiss landfill, for which this figure was derived,
features 0.02894 kilogram of carbon in captured landfill gas.\(^7\) Relating these two figures result in an electricity demand of 0.0466 kWh for each kilogram of carbon in captured landfill gas. So multiplying this figure with the carbon in captured landfill gas from a specific waste results in a waste-specific figure for pumping energy. Naturally, if no landfill gas is captured this amount is zero and no electricity demand is required.

### 3.2 Energy production

Depending on the chosen parameters, in the sanitary landfill model the captured landfill gas can be utilised for energy production. This was already modelled in (Doka 2003-III), but no multifunctionality was invoked and all burden was allocated to the disposal function of the landfill. Any resulting energy would be free of any burden (Doka 2003-III:63).

In ecoinvent v3 any allocation is managed on a database systems level and not within single inventories. It is therefore straightforward to inventory the resulting net energy production as system outputs.

Gross energy production is on one hand dependent on the energy available in methane in captured landfill gas. The user-definable model parameter %capt determines the fraction captured.\(^8\) And the waste specific modelling and degradabilities determine how much carbon in waste will be transferred to landfill gas (a share of 56% methane carbon is used). On the other hand the user-definable conversion efficiencies determine to what extent landfill gas energy is utilised and to which products (electricity and/or heat).

From any electricity production the electricity needed for pumping landfill gas is subtracted. I.e. it is assumed that – if possible – landfill gas utilisation provides the energy of landfill gas pumping. The latter is also calculated in a waste-specific manner, cf. chapter 3.1.1 'Landfill gas pumps').

So net energy production is fully flexible in this model, from specific waste composition, specific degradability and its dependence on climate, to site technological parameters like capture rate and conversion efficiencies, all can be provided by the user and deliver waste-specific net energy outputs.

### 3.3 Infrastructure and land use

No infrastructure is inventoried for unsanitary landfills or open dumps. It is assumed that these types of deposits are established on unprepared ground, i.e. that no liners, excavations or gas collection pipes are employed.

For unsanitary landfills and open dumps only some land occupation is inventoried. The CORINE land use category code 132 ('dump site') is applied. Land occupation is based on the given landfill height \(h\) and the given operation phase duration \(T_o\). Additionally it is assumed that the area will be left to spontaneously restore to a natural area, but that over a period of 10 years the area will still be classified as dump site, leading to a longer occupation duration.

---

\(^7\) From 0.33 kg carbon in average waste, times a short-term release to air of 16.338% (based on fraction's degradabilities) times 53% overall rate of LFG capture.

\(^8\) See chapter 5.5 'Landfill gas capture and flaring' on page 22
Eq. 3.1 \[ A = \frac{(T_o + 10\alpha)}{h \times \delta} \]

where,
- \( A \) Land occupation per kilogram of waste \([\text{m}^2 \cdot \text{a} / \text{kg waste}]\)
- \( T_o \) Operation phase duration, user-defined \([\text{years}]\)
- \( h \) Landfill height, user-defined \([\text{m}]\)
- \( \delta \) Waste density, 1000 kg/m\(^3\)

4 **Required climate data**

Dependencies on climate effects have been introduced into the present municipal landfill model. Several parameters are required to calculate climate-specific inventories. These new parameters are compiled in the workbook 'Central Repository.xls' in sheet 'site'. Only white cells shall be changed by the user.

Following climate data is used:
- Mean annual precipitation (MAP) [mm/year]
- Mean annual temperature (MAT) [°C]
- **Actual** annual evapotranspiration (ETa) [mm/year]

Values for these parameters can be derived from the following charts. But also other data sources can be used. But please note that the actual evapotranspiration ETa is required, not the potential evapotranspiration (PET). PET figures are theoretical values of total evapotranspiration, if water availability is not limited, while actual evapotranspiration ETa is limited by the actually present water. ETa is low to zero in hot and arid regions, while the PET in such regions is large.

![Fig. 4.1 Mean annual precipitation (MAP) [mm/year] (from www.worldgrids.org)](image-url)
Fig. 4.2  Mean annual temperature (MAT) [°C] (from www.worldgrids.org)

Fig. 4.3  Mean *actual* annual evapotranspiration (ETa) [mm/year] (Mu et al. 2011). An interactive, zomable online map for this data is available at http://www.arcgis.com/apps/OnePane/main/index.html?appid=b1a0c03f04994a36b93271b0c39e6c0f (Feb 2017)
4.1 Wind-eroded fraction

Only applied for landfill sites with negative net infiltration, i.e. dry or arid climates. Defines how much of the leachate evaporites, which formed on the landfill surface will be suspended to air and inventoried as air emissions. Default value is 10%.

5 Other required data

Several other parameters are used for the landfill model.

5.1 Geography code

The user can select a geographic region, which will be incorporated in the Ecospolo inventory.

5.2 Landfill height

The average height in meters to which the waste is ultimately accumulated in the landfill. This parameter will have influence on infrastructure in controlled landfills, land use and the available leachate volume per kilogram waste.

Any positive number in meters is accepted, even zero. A landfill height of zero represents completely aerobic conditions i.e. very shallow or continually mixed dumps, cf. chapter 5.7 'Methane correction factor MCF' on page 22. As a nominal height of zero would lead to infinite land use and unrealistic leachate volumes, any landfill height below 0.1 is corrected to 0.1 m = 10 centimeters to calculate land use.

It can be difficult to obtain typical generic data for landfill heights, especially for informal dumps. One source for India quotes typical landfill heights of 5 to 10 meters, which includes landfills ranging for unengineered open dumps to unsanitary landfills with cover and bottom liner only (Singh et al. 2007:Tab.1). The same study lists four existing, but anonymised unsanitary landfills in India with following characteristics. Neither of the landfills have bottom liners nor final covers.

| Tab. 5.1 Details for 4 unsanitary landfills in India from (Singh et al. 2007:Tab.3) |
|---------------------------------|--------|--------|--------|--------|
| Area ha | A (Delhi) | 30 | B (Delhi) | 16.2 | C (Chennai) | 20 | D (Chennai) | 55 |
| Height m | | 16 | 13 | 3 | 3 |
| Operation phase yr | 22 | 13 | 18 | 18 |
| Bottom liner | none | none | none | none |
| Leachate collection | none | none | none | none |
| Final cover | none | none | none | none |

5.3 Landfill operation phase duration

Duration in years during which the landfill is being actively filled. This will be used for land use calculations.
The operation phase should be equivalent to the total landfill capacity in kg divided by the annual waste deposition rate in the landfill in kg/year. The capacity in turn is connected to the occupied landfill area and the average landfill height.

5.4 Landfill management

Choice between sanitary landfill (s), unsanitary landfills (u) and open dump (o).

In the unsanitary landfills and open dumps all leachate is emitted directly to groundwater (or to soil/air in arid climates). See also Tab. 3.1 on page 17.

The chosen waste management type will affect the name given to the process, e.g. "treatment of municipal solid waste, open dump".

5.5 Landfill gas capture and flaring

%capt: Percentage of capture of generated landfill gas as an average over all the methanogenic phase. Due to seepage, trailing off of gas production even in well designed landfills the capture rates are usually not above 60% (though of course in single years higher rates can be achieved). Non-captured LFG is directly emitted to air. Default value is 53%.

%flare: Percentage of flaring of the captured landfill gas. Default value is 34% in Switzerland. Gas captured, but not flared is assumed to be utilised in a gas motor for energy production.

The fate of landfill gas is therefore calculated in the following way:

\[ \text{landfill gas directly emitted to air} = (1 - \text{%capt}) \]
\[ \text{landfill gas flared, but not utilised} = \text{%capt} \times \text{%flare} \]
\[ \text{landfill gas utilised} = \text{%capt} \times (1 - \text{%flare}) \]

Please note that the capture rate defined here must be coherent with the chosen scope of the dataset. For instance in a treatment dataset is supposed to represent a national average, the capture rate must be representative and include also all landfills which do not capture any landfill gas at all. For instance if 80% of waste is disposed in landfills without any capture, and 20% is disposed in landfills with a capture rate of 50%, then the correct national average for \%capt is 10% (= 80% · 0% + 20% · 50%). The rate of flaring \%flare however relates only to those landfills that actually capture landfill gas, not all landfills.

5.6 Landfill gas utilisation efficiencies

Conversion efficiencies in landfill gas utilisation for heat and electricity describe the technology of landfill gas utilisation. They relate only to the heating value of landfill gas actually converted in utilisation, not to landfill gas not utilised. Gross efficiencies are requested here.

5.7 Methane correction factor MCF

In shallow and/or uncompacted waste dumps the methane in landfill gas can become oxidised to CO\(_2\) due to exposure to aerobic conditions. The IPCC landfill model (IPCC 2006-3) has a methane correction factor to account for this effect with the following values. A value of 1 means there is no
further methane oxidation before release, a value of 0.5 means that half of the methane is oxidised to CO₂ before release. At an MCF of 0.7 a share of 30% of the methane in landfill gas will be oxidised. For managed landfills with compacted and covered waste an MCF of 1 will always be used.

**Tab. 5.2** Values for the methane correction factor MCF according to (IPCC 2006-3).

<table>
<thead>
<tr>
<th>Site type</th>
<th>IPCC Methane correction factor MCF</th>
</tr>
</thead>
<tbody>
<tr>
<td>Managed – anaerobic</td>
<td>1.0</td>
</tr>
<tr>
<td>Managed – semi-aerobic</td>
<td>0.5</td>
</tr>
<tr>
<td>Unmanaged – deep (&gt; 5 m waste) and /or high water table</td>
<td>0.8</td>
</tr>
<tr>
<td>Unmanaged – shallow (&lt; 5 m waste)</td>
<td>0.4</td>
</tr>
<tr>
<td>Uncategorised solid waste disposal sites</td>
<td>0.6</td>
</tr>
</tbody>
</table>

These values are taken as a basis to derive height-dependent formula to calculate the MCF in the model for unmanaged dumps. Following formula is used. At a landfill height of zero, no methane emissions are resulting, as no anaerobic conditions will occur. With increasing height of the dump, anaerobic conditions in the landfill body are more likely, but a part at the surface of the uncovered dump will still be able to have enough oxygen for methane oxidation. The MCF therefore never reaches a value of 1 for any height, only asymptotically approaches it.

\[
MCF = \begin{cases} 
  h > 6\text{m} : MCF = 1 - \frac{2.0604}{h} \\
  h < 6\text{m} : MCF = 0.26833 \times \sqrt{h} 
\end{cases}
\]

where,
- **MCF** Methane correction factor [-]
- **h** Landfill height, [m]

**Eq. 5.1**

![Fig. 5.1 Methane Correction Factor MCF in dependence of dump height](image)

The MCF and share of methane in emitted landfill gas is therefore determined by the landfill height specified by the user. It is however optionally possible to enter an override value for the MCF, which is then used with priority.
Giving a height of zero - or an MCF of zero - represents a completely aerobic dump, where no methanogenic conditions occur and any carbon air emissions occur as CO₂. This can be representative for very small, private dumps. As the landfill height is used for other calculations as well (leachate volume, land occupation), very small landfill heights will be corrected to avoid errors, cf. chapter 5.2 'Landfill height' on page 21.

5.8 Waste composition

Waste definitions are given in the central 'hub' workbook "Central Repository 2017.xls" in sheet 'waste input'.

Calculating waste-specific inventories is the core purpose of the landfill model. For unsanitary landfills in developing nations waste scavengers can pick out recyclables from the waste, also animals can pick out food items. For the purposes of this model, the landfilled composition is assumed to remain on location and lead to emissions. This means that the defined waste composition shall reflect the composition after recyclables have been removed.

6 Properties of exchanges

In Ecospold2 each exchange can have several properties describing the exchange in more detail, like dry mass, water content, fossil carbon content etc. The tool adjusts also some properties of exchanges. For instance an emission of NMVOC or DOC from a biomass waste material should receive content properties reflecting the fact that this emission will contain no fossil carbon. On the other hand wastes with fossil carbon should lead to emissions with their biogenic carbon properties set to zero.

Carbon content and share of biogenic (non-fossil) carbon are part of the waste fraction definition. The issue of carbon content properties is straightforward for single fractions with either 0% or 100% fossil carbon. But since the tool can also process mixtures of dissimilar fractions (complex wastes) correct properties must also given to the emissions in that case. This is addressed in the following.

6.1.1 Non-fossil carbon properties in emissions of landfill

In the tools of the current project any carbonaceous emissions, like NMVOC or DOC, receive adjusted properties for biogenic (non-fossil) and fossil carbon. Two different shares of biogenic carbon are derived: for short-term emissions and for long-term emissions.

For short-term emissions of the landfill the biogenic carbon share is derived from the carbon masses degraded in the short term (0-100a). It is the ratio of biogenic carbon degraded in the short term over the total carbon degraded in the short term.¹⁰

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¹⁰ As reported by vacationers in 2016, waste disposal by inhabitants of Madagascar can involve digging shallow pits with the volume of merely some liters near the house, filling them with waste, lightly covering them up and then moving to an adjacent patch.

¹⁰ Since there is a release factor of 100% for carbon in the model, i.e. any degraded carbon is also emitted, the release factor does not have to be heeded here.
\[
\%b_{ST} = \left( \frac{\sum_{\text{fraction } i} f_i \times m_{c,i} \times D_i \times \%b_i}{\sum_{\text{fraction } i} f_i \times m_{c,i} \times D_i} \right)
\]

Eq. 6.1

where,
\%b_{ST} Fraction of non-fossil, biogenic carbon in short-term emissions
f_i Share of fraction i in selected waste (mixture)
m_{c,i} Mass of all carbon in fraction i, [kg carbon/ kg wet waste]
D_i Degradability of fraction i
\%b_i Share of biogenic carbon in fraction i

The derived biogenic carbon ratio is also applied to any emissions of secondary wastes, e.g. incinerated sludges from any treatment of short-term leachate.

The ratio reflects the properties of the defined waste or waste mixture. For a waste containing only biogenic carbon, \%b_{ST} will be 100%. For a waste containing only fossil carbon, \%b_{ST} will be 0%. For a mixed waste containing biogenic and fossil carbon, the ratio will be close to 100%, as common fossil plastics have a usually only minor degradability. Some short-term carbon releases can also be fossil in origin, e.g. from leached plasticizers. The model also allows for correct speciation of carbon, e.g. from degradable polymers (D>>1%) which were manufactured from fossil carbon (%b=0%) like polycaprolactone PCL or polybutylene succinate PBS.

For long-term emissions a different biogenic carbon share of is applied. After the short-term phase much biogenic carbon remains in the landfill together with most fossil carbon. During the long-term phase (100-60'000a) carbon will be continued to be washed out. Long-term emissions will contain carbon of both origins, but in average operation the share of biogenic carbon will be reduced compared to the short-term, as the quickly degradable carbon will be mostly gone, and chiefly recalcitrant and slowly degradable biogenic carbon remains after the short-term. No information on the relative rates of decay of recalcitrant biogenic carbon vs recalcitrant fossil carbon are available, which would be necessary to pinpoint the biogenic carbon share during the long-term phase in non-trivial cases.\(^{11}\)

The ultimate end point of landfill development, which may or may not be reached during the modelled time frame of 60'000 years, is complete washout of all carbon. This endpoint allows the calculation of a biogenic carbon share from the carbon remaining after the short-term. This implicitly assumes that decay speeds of both types of carbon remain constant and therefore also the share of biogenic carbon in emissions.

\[
\%b_{LT} = \left( \frac{\sum_{\text{fraction } i} f_i \times m_{c,i} \times (1 - D_i) \times \%b_i}{\sum_{\text{fraction } i} f_i \times m_{c,i} \times (1 - D_i)} \right)
\]

Eq. 6.2

This formula is very similar to the previous one with the exception that the focus is not carbon degraded in the short-term, but carbon not degraded, signified by the use of (1-D\(_i\)). This is the carbon available to leaching in the long-term only. The share of biogenic carbon in long-term emissions will be applied to emissions stemming from long term leachate, e.g. TOC and DOC to groundwater, or – in arid sites – carbon in windblown particulates and carbon to soil.

The development of the ratio of biogenic carbon in emissions from a mixed carbon waste can be illustrated with the following scheme plotting remaining biogenic carbon versus remaining fossil

\(^{11}\) A waste possessing only one kind of carbon – fossil or biogenic – is a trivial case, as all emissions then have unmixed characteristics. Waste mixtures possessing fossil and biogenic carbon together are non-trivial.
carbon (not to scale). The model starts out with the deposited initial waste composition (1) with a defined content of biogenic and fossil carbon (point 1). After the short term a lot of biogenic carbon has been emitted and some fossil carbon as well. The steep green arrow represents emissions with a high share of biogenic carbon. After the short-term, waste still remains (point 2) but with an altered biogenic carbon share. The development in the long term can be characterised by two observations:

- The share of biogenic carbon will likely go down, as easily degradable carbon is already emitted and decay of fossil carbon matrixes will probably increase.
- With further emissions the development will approach the point of complete emission (point 3).

Point 3 might not actually be reached within the modelling timeframe of 60'000 years. This depends on climate characteristics. For the share of biogenic carbon in long-term emissions the straight line between point 2 and point 3 was employed. Point 2 is defined by the carbon remaining after the short-term.

For the trivial case of wastes with only one type of carbon, development moves simply along one of the two axes, and share of biogenic carbon is simply either 100% or 0%.

Fig. 6.1 Scheme of temporal development of biogenic and fossil carbon remaining in the landfill

7 Outlook

Models for waste- and location-specific inventories of uncontrolled waste dumping were presented here. The models must remain coarse, since they concern an unmanaged and uncontrolled process, or processes only partially under human control. The models are nevertheless useful in obtaining more accurate burdens of disposal in developing countries, which previously were approximated with the more elaborated treatment process of the developed regions.

One condition of the models continues to be that the deposited waste body remains on site, i.e. is not eroded mechanically, but merely affected by decomposition and leaching (Doka 2033-III:13). In an unengineered dump this assumption might be optimistic in the long run. Wind or wildlife can disperse waste items. This can lead to problems similar to littering (decentralised dumping of waste). Littering
can have a variety of effects, not all of which are environmental. Littering can be an eyesore and impede on aesthetic qualities of a landscape or dwelling area, which in turn can have an impact on touristic values of an area. These effects are not representing environmental burdens as recorded in LCA and therefore should be assessed with other methods than LCA. Littering can lead to costs for a municipality from increased collection efforts. The additional effort for collection (e.g. use of machinery, transport of personnel and waste) could be part of an environmental LCA of littering. Uncollected littered waste might also have direct toxic effects, for instance from cigarette butts, which could be captured with impact assessments, requiring a release model of compounds, as well as characterisation factors in LCIA. Toxic effects might also ensue from ingestion of waste items. Besides these more traditionally toxicological effects, littered waste might have physical effects on wildlife, e.g. animals choking on certain plastic items.

Fig. 7.1   Fox caught on a rusty drinks can. From http://homepage.eircom.net/~foxwatchireland/ (18 Jul 2017)

Unless littered waste accumulates to large heaps, single waste items are probably unlikely to affect local biodiversity directly. I.e. although some individuals of wildlife might suffer or even die from littered waste items, it is unlikely they will lead to a decrease in species richness locally. While for human health damages LCIA observes the burdens on individual humans, burdens on biodiversity are expressed in effects on the presence of the species as a whole, i.e. a decrease in overall species richness, and not the health state of an individual of any species. It is however imaginable to refine the merely twofold framework of species richness ("present/unaffected" vs. "not present/affected") currently used in LCIA into a more qualitative scale where gradual damages to individuals of wild species are depicted as an environmental burden too, not only the reversible disappearance of the whole species. This could be conceptually similar to the disability weight in counting health damages in human individuals, where DW=1 is death and DW=0 is perfect health.

Similar problems arise when considering marine dumping. Waste can get into the oceans by a variety of channels, like erosion of landfills near the shore, waste carried by rivers, waste littered on beaches, and waste dumped or lost from ships, and – rarely – from airplanes.

While biomass waste – like food waste, paper, cardboard, faeces – will usually disintegrate and be metabolised rather quickly, and mineral waste, like glass and metals, will likely sink to the sea floor and buried in sediments, oceanic plastic waste has recently received some attention. Due to its buoyancy and longevity, marine plastic invites a range of problems. Firstly there are physical effects. As with land-based littering, these effects depend on the physical shape of the plastic. Lost fishing nets can continue to "ghost fish" marine life. Ropes and strings can entangle animals. Plastic foils with holes can trap animals. As with land-based littering these effects could only be modelled if an extension to health damages to wildlife individuals were introduced.
Occasionally plastic waste might also have some marginal beneficial effects: Larger floating debris like for example a drinks carte or a bucket can provide a shelter volume for fish and living space for mussels, clams, sea anemones and seaweeds in a way the open ocean would not. A 15 m free-floating island of conglomerated debris of the 2011 tsunami in Japan consisting of buoys, ropes and aquafarming equipment has been found to form a small biotope for fish, barnacles etc. (Ghose 2014).

Plastic integrity might be compromised by exposure to UV light and mechanical forces. Some plastic might also sink to the sea floor. After breaking into smaller pieces, plastic might still pose a threat to a range of marine animals, who might swallow plastic pieces, mistaking it for food. When the animals are not being able to regurgitate plastic items, they accumulate in the animal, interfering with its ability to properly digest food.

Even in smaller form, plastic can pose a threat. Microplastic can absorb organic pollutants already in the seawater and thus increase the uptake and bioaccumulation of those pollutants in animals ingesting microplastics. This effect would be amenable to toxicity models used in LCIA. These models already feature standard bioaccumulation rates for pollutants. Microplastics increase these bioaccumulation rates. Assuming current emissions of a range of pollutants in seawater, the cumulated increase in exposure and impact could be attributed to the presence of microplastics. This would provide an avenue to derive compatible toxic impacts from microplastics, but it requires information on bioaccumulation increases and also a fate model and ultimate sink(s) of the plastic itself.

Marine microplastics originate not only from larger plastic pieces broken down in the sea, but can also be generated already on land, e.g. microplastics in certain cosmetic products for instance shower gels, soaps and skin exfoliating preparations (Gouin et al. 2015). Also synthetic textiles release microplastics, which during washing are transferred to water bodies. These microplastics released directly from their man-made application can be called primary microplastics. Wastewater treatment plants might reduce the flow of primary microplastics somewhat. Fate modelling of these particles
involves more stages than microplastics forming from larger debris in the open ocean, but some research on these steps has already been conducted.

These explorations illustrate that further aspects of waste treatment can still be explored and also the increasing shift of lesser-controlled processes into Impact Assessment procedures.

8 References

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

Alfayez 2011


Diaby et al. 2006


Doka 2003-II


http://www.doka.ch/13_II_WastelIncineration.pdf

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