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LCIA OF IMPACTS ON HUMAN HEALTH AND ECOSYSTEMS • METHODOLOGY

Gone…but not away—addressing the problem of long-term impacts from landfills in LCA

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Abstract

Background, aim and scope Land filling of materials with content of toxic metals or highly persistent organic compounds has posed a problem for life cycle assessment (LCA) practitioners for many years. The slow release from the landfill entails a dilution in time, which is dramatic compared to other emissions occurring in the life cycle, and with its focus on the emitted mass, LCA is poorly equipped to handle this difference. As a consequence, the long-term emissions from landfills occurring over thousands of years are often disregarded, which is unacceptable to many stakeholders considering the quantities of toxic substances that can be present. On the other hand, inclusion of all future emissions (over thousands of years) in the inventories potentially dominates all other impacts from the product system. The paper aims to present a pragmatic approach to address this dilemma.

Materials and methods Two new impact categories are introduced representing the stored ecotoxicity and stored human toxicity of the contaminants remaining in the landfill after a 'foreseeable' time period of 100 years. The impact scores are calculated using the normal characterisation factors for the ecotoxicity and human toxicity impact categories, and they represent the toxicity potentials of what remains in the landfill after 100 years (hence the term 'stored' (eco)toxicity). Normalisation references are developed for the stored toxicity categories based on Danish figures to support comparison with indicator scores for the conventional environmental impact categories. In contrast to the scores for the conventional impact categories, it is uncertain to what extent the stored toxicity scores represent emissions, which will occur at all. Guidance is given on how to reflect this uncertainty in the weighting and interpretation of the scores.

Results and discussion In landfills and road constructions used to deposit residuals from incinerators, less than 1% of the content of metals is leached within the first 100 years. The stored toxicity scores are therefore much higher than the conventional impact scores that represent the actual emissions. Several examples are given illustrating the use and potential significance of the stored toxicity categories.

Conclusions and perspectives The methodology to calculate stored human and ecotoxicity is a simple and pragmatic approach to address LCA's problem of treating the slow long-term emissions at very low concentrations appropriately. The problem resides in the inventory analysis and the impact assessment, and the methodology circumvents the problem by converting it into a weighting and interpretation issue accommodating the value-based discussion of how to weight potential effects in the far future.

Keywords Ecotoxicity · Human toxicity · Landfill · Life cycle impact assessment (LCIA) · Long-term impacts · Stored toxicity

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1 Background, aim and scope

The impact assessment of emissions of toxic substances from landfills, e.g. metals from flue gas cleaning products or polychlorinated biphenyls from old electronics, is controversial in life cycle assessment (LCA). These substances typically leach very slowly from the landfill, and often, emissions to the environment will not occur before the collection of leachate ceases in the future. The slow leaching means that leachate concentrations are low—often far below predicted thresholds of effects in the surrounding environment. On the other hand, the total amounts leaving the landfill may be considerable in the very long time perspective, and this gives problems in the life cycle impact assessment (LCIA) where impacts are normally aggregated over time. One gram of a toxic chemical emitted to water thus has the same impact score regardless whether it is emitted as a pulse (in seconds) or slowly over thousands of years. LCIA focuses on the emitted mass, not on the concentration, and this disregard of the temporal course of the emissions (the different “dilution in time”) introduces a strong bias between landfill processes, emitting over centuries or millennia, and all the other processes in a product life cycle, which typically emit over seconds to hours or days.

Estimation of emissions from landfills has been the topic of many papers and several workshops over the last decade (e.g. Doka and Hischier 2005; Finnveden and Huppes 1995; Finnveden and Nielsen 1999; Hellweg et al. 2001; Hellweg and Frischknecht 2004; Sundqvist et al. 1997.) No common approach has emerged on how to address long-term emissions from landfills, and in many LCA cases, the topic is not addressed at all (e.g. Rieradevall et al. 1997; Diamond et al. 1999; Ménard et al. 2004; Morselli et al. 2005; Durucan et al. 2006; Emery et al. 2007). When the topic is addressed, the landfill emissions have hitherto been included in the LCI as if they occurred from any other process in the life cycle following one of three approaches:

1. Estimating the emissions in the foreseeable future (e.g. aerobic phase or 100 years) and ignoring whatever is emitted thereafter (Finnveden 1999; Nielsen and Hauschild 1998)
2. Modelling the entire emission from the landfill using geochemical modelling to determine the kinetics (Hellweg 2000)
3. Modelling the emission from the landfill until the substance concentrations in the leachate reach the level of the substance background concentration in pore water or ground water (Birgisdóttir 2005)

Each of these approaches has inherent problems. The first approach will often be unacceptable to important stakeholders, and it is not in accordance with a precautionary approach to decision making, since only a very small fraction of the toxic substances in the landfill may be emitted in the foreseeable time horizon, and a large potential for toxic releases remaining in the landfill escapes the assessment (e.g. Finnveden and Nielsen 1999).

The second approach provides somewhat meaningless results for an LCA. It considers a time horizon of 10^4–10^5 years and we cannot give any reasonable estimates of the management and the conditions of the landfill during this time. Normally, the quantities of metals emitted over such long periods of time will be considerable, and when they are assessed together with the other emissions that actually occur in the product life cycle, they will often completely dominate the impact assessment, which is in strong contrast with the importance that is normally assigned to land filling of metals. To avoid such a bias, the modelling of the entire emissions is often combined with a discounting of future impacts—implying that emissions are less problematic if they occur in the distant future. Hellweg et al. (2003) discuss the issue of discounting in more details.

The third approach considers the landfill to become part of the ecosphere when the leachate concentrations reach the background level in the environment surrounding the landfill, and this approach only includes emissions occurring up to that point. Depending on the background levels, this may occur within a few centuries for some of the metals, but this approach requires knowledge of the leaching kinetics, which is presently not available for many of the concerned metals and persistent compounds in the landfill, particularly under potentially changing redox- and pH conditions in a distant future.

As a fourth approach, a distinction between emissions occurring in different time frames was proposed in the US LCI (Camobreco et al. 1999) in which three separate time frames are defined: 20 years (the active period of the landfill), 100 years (roughly the life span of a given generation), and 500 years (presented as corresponding to indefinite time reference, where emissions reach their theoretical yield). However, a much longer time frame is necessary for long-term emissions of persistent substances (e.g. Hellweg 2000) and as was recommended by the 22nd Discussion Forum on LCA (Hellweg and Frischknecht 2004), the Ecoinvent LCI database has recently applied a distinction between short- and long-term emissions (Doka and Hischier 2005). The long-term emissions (occurring after 100 years and up to 60,000 years) are reported as a separate emission category distinguished from the short-term emissions, but it remains unclear how impact assessment is proposed differentiated between these categories of emissions. In the Impact 2002 + User Guide, Humbert and co-workers recommend the same distinction between short- and long-term emissions and propose that their impact scores are calculated using the same factors and presented separately in the results to allow evaluating in the interpretation whether they potentially form a problem for future generations.

According to Doka and Hischier (2005), long-term emissions are only relevant for disposal processes and uranium production. The following processes can be specified:

- Landfills for waste (household waste, non-combustible waste and some industrial waste)
- Landfills for flue gas cleaning products and slag and ashes (from coal power plants and waste incineration)
- Road systems and other uses of residual products from incineration processes (e.g. additive to concrete)
- Landfill of waste from mining activities

Long-term emissions are only relevant for the very persistent substances in waste and residual products, i.e. the metals and some highly persistent organic compounds, since all non-persistent organic substances are expected to be degraded in a landfill within the first 100 years.

The uncertainty in predicting the emissions occurring after the foreseeable future is very large. Many materials and substances, which remain in the landfill longer than the foreseeable time horizon, are generally considered either inert or very persistent. This is the case for polyvinyl chloride (PVC) and most other polymers, glass, ceramics, metals, slag and ashes, tar, impregnated wood and many other materials. However, in the very long time perspective, most of these materials will eventually decompose to an extent where hazardous components will be made available for transport and leaching out of the landfill and into the surrounding environment. The decomposition of highly persistent substances and materials in the landfill is influenced by the physical and chemical conditions in the landfill, and these will change with time under influence of different parameters, which are discussed extensively in Hansen et al. 2004. In the very long time perspective (centuries or millennia), two parameters are found to be decisive for the future emissions of persistent pollutants from the landfill (Hansen et al. 2004):

- The future management of landfills by human society
- Geological processes occurring at the landfill site—processes like coastal erosion, glaciers, or earth quakes

The uncertainties in these two parameters are so large in the long time perspective that it is meaningless to apply some average situation and try to model long-term emissions for this since the variation in emissions from a worst to a best case will be very large. Instead, we propose to circumvent the modelling uncertainties for the long time emissions without omitting the potential impacts they may have from the LCA, by creating a new impact ‚category which we call ‘stored toxicity’.

2 Methodology

The methodology for assessment of stored toxicity was developed for the Danish EPA as part of a project with the main goal to collect data and establish a methodology for calculating the impacts of treatment and land filling of waste in LCA (Hansen et al. 2004). Initially, a distinction between short-term emissions (<100 years, ‘foreseeable future’) and long-term emissions (in principle indefinite time horizon) was introduced, as proposed by Nielsen and Hauschild (1998) and also recommended in Hellweg and Frischknecht (2004). For the short-term emissions, inventories were established for generic treatment practices and operations, e.g. incineration, transport, establishing and operation of landfills, etc. Emissions were modelled for land filling of slag, ashes and flue gas cleaning products from electricity production and from incineration of waste, partially based on empirical data for leaching of metals from residual products (Hjelmar 1996). As a consequence, the environmental impacts from these emissions could now be included in the ordinary life cycle impact assessment, especially in the toxicity impact categories. Due to incompleteness of data, it was only possible to include a limited number of metals in the short-term emission modelling. For incineration, these were As, Cd, Cr, Cu, Ni, Pb and Zn, and for power production, they were Cd, Cr, As, Mo, Se, V, Mg, Pb and Zn.

3 Inventory

Considering the difficulties and uncertainties involved in gathering data and modelling even the short-term emissions (<100 years), it was considered futile to attempt to model the long-term emissions. Therefore, all residual substances in the landfill after 100 years (for the metals typically more than 99%) are considered potential emissions contributing to the stored toxicity categories:

- Stored ecotoxicity
- Stored human toxicity

4 Characterisation

For each of these impact categories, the characterisation applies the ordinary characterisation factors for ecotoxicity and human toxicity of the substances in question as also suggested by Humbert et al. (2005), but in contrast to the sensitivity analysis proposed by Humbert et al. (2005), it is proposed in this study to treat the long-term emissions in these two new impact categories. The impact scores for the stored toxicity categories represent the impacts that may happen on a long term if all remaining toxicity in the landfill is released. In this study, a coastal landfill will probably represent a different situation from an inland landfill both in terms of the environmental compartments, which become affected, and the nature of the geological processes that may cause the future releases. Again, this is not something we can know much about, and it was therefore decided to assume a division of the emissions between 50% going to water and 50% going to surface soil to allow the scores to reflect the toxicity potentials in both of the major environmental media. The use of the same characterisation factors as for the conventional ecotoxicity and human toxicity impact categories is due to the reasoning that uncertainties about the conditions under which the long-term emissions will be released and act are so large that current characterisation factors are as good an estimate as any to represent future conditions.

5 Normalisation

For the use of the new impact categories together with the existing impact categories of an LCIA method, there is a need for normalisation references in order to allow comparison to other impacts from the product system. The inventory for a set of Danish-based normalisation references was established based on mass flow analyses of substances in all major waste streams containing significant amounts of persistent hazardous substances, which were landfilled in DK in the year 1994 (the same reference year as applied for all other impact categories in EDIP97—see Stranddorf et al. 2005). Wastes included were slag and ashes from waste incinerators and coal-fired power plants, impregnated wood, tar and polluted soil among others (Hansen et al. 2004). The stored ecotoxicity of the inventoried substances was determined applying the environment-dependent interatomic potential (EDIP) characterisation factors for chronic aquatic and terrestrial ecotoxicity as shown in Table 1. Finally, the normalisation references were expressed as person equivalents by dividing by the number of inhabitants in Denmark in 1994 in accordance with the EDIP methodology (see Hauschild and Wenzel 1998).

The stored ecotoxicity normalisation references are dominated by copper (various sources) and polycyclic aromatic hydrocarbon (PAH) from creosote (preservation of wood). For human toxic impacts, a similar approach gave normalisation references for exposure via water and soil as shown in the Electronic supplementary material (Table 2).

Table 1 Normalisation references for stored ecotoxicity based on an inventory for Denmark 1994, applying the EDIP97 factors for aquatic and terrestrial ecotoxicity for the stored substances, expressing the impacts as compartment volumes contaminated to the predicted no effect concentration of the substance (PNEC)

Substance or substance group	Source	Amount land filled (ton/year)	Stored ecotoxicity score			
			Water		Soil	
			m^3 water	%	m^3 soil	%
Nickel	Various (MFA)	955	3.2×10^{11}	0.5	3.3×10^{8}	12
Mercury	Various (MFA)	0.6	2.4×10^{9}	0.0	3.2×10^{7}	1.2
Cadmium	Various (MFA)	19	1.1×10^{12}	1.9	2.0×10^{8}	7.5
Lead	Various (MFA)	1,550	1.6×10^{12}	2.6	7.8×10^{7}	2.9
Arsenic	Various (MFA)	64	6.0×10^{10}	0.1	1.1×10^{8}	4.0
Arsenic	Wood preservation	16	1.5×10^{10}	0.0	2.6×10^{7}	1.0
Copper	Various (MFA)	5,600	3.5×10^{13}	59	5.6×10^{8}	21
Chromium	Various (MFA)	3,250	1.1×10^{12}	1.8	1.6×10^{8}	6.0
Chromium	Wood preservation	20	6.7×10^{9}	0.0	1.0×10^{6}	0.0
PAH (benz(a)pyrene-TEQ)	Contaminated soil	0.14	8.2×10^{10}	0.1	4.6×10^{6}	0.2
PAH (benz(a)pyrene-TEQ)	Car tires	0.03	1.6×10^{10}	0.0	8.7×10^{5}	0.0
PAH (benz(a)pyrene-TEQ)	Bio ashes	0.001	4.3×10^{8}	0.0	2.4×10^{4}	0.0
PAH (benz(a)pyrene-TEQ)	Creosote in wood	32.5	1.9×10^{13}	33	1.1×10^{9}	40
PAH (benz(a)pyrene-TEQ)	Asphalt	0.46	2.8×10^{11}	0.5	1.5×10^{7}	0.6
Dioxin (I-TEQ)	Slag/ashes	0.00007	9.2×10^{10}	0.2	4.9×10^{6}	0.2
Total			5.9×10^{13}	100	2.6×10^{9}	100
Inhabitants DK, 1994 (5.166×10^{6})						
Normalisation reference (person equivalent, m^3/person/year)			1.1×10^{7}		5.1×10^{2}	

The normalisation reference is expressed as a person equivalent (annual impact from an average person). Inventory figures documented in Hansen et al. 2004

6 Weighting and interpretation

Compared to the traditional environmental impacts characterised in LCIA, the stored toxicity impacts are considerably more uncertain. In the first place, it is unknown how large a fraction of the stored substances will ever be released to the environment—the stored toxicity potential thus represents the worst case where everything is released. Secondly, the temporal course of the emissions still remains unknown. They may occur suddenly as a consequence of some geological event, but a large fraction of the release is likely to occur through gradual leaching over thousands of years at very low concentrations, which may not be able to cause any effects in exposed individuals or ecosystems. This should be taken into account in the interpretation of the results, and the weight assigned to these new impact categories relative to the traditional impact categories should reflect this. Applying the ethical archetypes derived by Hofstetter (1998) from Cultural Theory and applied in the EI99 methodology (Goedkoop and Spriensma 2000), the following importance might be assigned to the stored toxicity impacts by three different ethical profiles in the interpretation of the results:

- Individualist: A very low importance and a weight close to zero due to the high uncertainty of the release and conservative nature of the stored toxicity potentials and due to the very long time span, which will allow us to prepare and find a solution if impacts should occur. The use of positive discounting, which will typically be favoured from an individualist perspective will also reduce the significance of the future impacts.
- Egalitarian: A certain importance. The egalitarian may accept the conservative estimate of the stored toxicity, applying a precautionary approach with an aim for intergenerational equity, particularly for the contributions from very toxic metals or persistent organic pollutants.
- Hierarchical: The hierarchical tends to show more faith in our ability to model and regulate the emissions and would probably prefer best estimates to the conservative estimates inherent in the stored toxicity potentials. On the other hand, a certain importance (between the individualist and the egalitarian) will probably be given to the stored toxicity in order to avoid that the LCIA methodology favours uncritical land filling of persistent toxic substances.

As a practical approach to the interpretation of the stored toxicity scores in relation to the traditional toxicity scores, the following guidance is given based on our experience with the use of the stored toxicity potentials using the EDIP methodology (Hansen et al. 2004):

- When stored human toxicity or ecotoxicity scores are of the same order of magnitude (i.e. less than a factor 10 higher) as the persistent toxicity scores for the other emissions from the product system, they should not be given any weight in the interpretation of the results.
- When the stored toxicity scores are between one and two orders of magnitude higher than the traditional persistent toxicity score, they should be included in the interpretation with a weight similar to that assigned to the traditional toxicity scores.
- When the stored toxicity scores are more than two orders of magnitude higher than the traditional persistent toxicity score, they should be given a high weight in the interpretation, and the toxicity scores of the short term emissions may be ignored.

Weighting factors based on distance to politically set reduction targets (Wenzel et al. 1997) have been derived for the stored toxicity categories to be used together with similar factors for the traditional impact categories (Hansen et al. 2004).

7 Results

As noted earlier (also by e.g. Finnveden and Nielsen 1999), the emissions of metals and persistent organic compounds, which occur from a landfill in a short term of 100 years, are almost negligible compared to those that may occur in the long term. In a specific study where we compared different end of life treatments for PVC, we only had data to specifically trace 9 metals and calculate transfer factors for these to be used in the short-term emission modeling for the landfill. However, the first 100 years (for a typical Danish landfill corresponding to an L/S[1] of 2) accounted for a leaching of only 0.00006% to 0.2% of the total amount of heavy metals present in the landfilled waste. It was shown that even in an application of bottom ash for road building, where the leaching is significantly higher due to the thinner layer of residuals (100 years leaching corresponds to an L/S of 13.5), still less than 1% of the total amount leached within 100 years for all heavy metals (Birgisdóttir et al. 2007). When the emissions during the first 100 years are assessed together with the other toxic emissions from the product system, they are nevertheless still noticeable in their contribution to the total toxic impact from the system.

To calculate the stored ecotoxicity and stored human toxicity potentials from the pollutants remaining in the

[1] L/S is the liquid to solid ratio—the ratio between the weight of percolated water and the weight of the residual product. The smaller the precipitation and the thicker the landfilled layer of material, the lower the L/S at any given time. In leaching tests, the naturally occurring leaching is simulated through accelerated percolation of water through the residual product reaching high L/S ratios much quicker than in the real world.

landfill, the study applied the standard EDIP97 characterisation factors for ecotoxicity and human toxicity in soil and water, assuming a final 50:50 partition in soil and water as described above. Following normalisation of the stored toxicity scores, they are summed to give one stored ecotoxicity potential and one stored human toxicity potential.

Examples of results are found in Fig. 1, which shows the environmental profile of disposal of 1 kg PVC with a content of lead- and zinc-based stabilisers and DEHP as plasticiser. The material is incinerated, and the residuals from the flue gas cleaning are landfilled. The assessment is performed using the EDIP97 method with additional modelling of stored toxicity potentials. The most obvious aspect is the rather dominating 'stored toxicity' impact scores, which, for stored ecotoxicity, is as large as the largest of the traditional impact scores (global warming and acidification) and for stored human toxicity even a factor 7 higher. In the persistent toxicity score (chronic human and ecotoxicity from all emissions occurring before 100 years), more than 60% is caused by the modelled *short-term* emissions from the landfill of the residual products. A comparison of stored human toxicity and persistent toxicity shows a difference of a factor 80, and applying the interpretation guidance given above, particularly the stored human toxicity should thus be given a weight similar to that given to the toxic impacts from the short-term emissions from the life cycle.

A reason for the high stored toxicity potentials that can be observed may also be that the normalisation references based on the Danish situation are too small. For the other impact categories, the EDIP normalisation references are based on European impacts, and the person equivalent represents an average European. Development of European normalisation references would uncover this, but it is foreseeable that landfilling of toxic substances is lower in Denmark than in many other European countries.

Figure 2 shows a case where the inclusion of stored toxicity does not have a strong effect. The figure presents the results for production of 1-kWh electricity to the grid using the Danish average grid mix (mainly coal-based). In this case, the residual products from the flue gas cleaning of the coal-fired power plants are not landfilled but mainly used for filling in harbours, which causes a relatively rapid leaching of their content of persistent substances. This is seen in a persistent toxicity score, which is more prominent than in the case shown in Fig. 1, whereas the stored toxicity is rather modest, reflecting that little is left of the persistent substances after 100 years.

Birgisdóttir and co-workers study the use of residual products from waste incineration as construction material in road building, substituting the use of gravel. Their results emphasise the importance of including the potential long-term emission with stored ecotoxicity scores, which are 2–3 orders of magnitude higher than the traditional environmental impacts considered (Birgisdóttir et al. 2007).

8 Discussion

As illustrated by the results shown above and as emphasised in earlier work (e.g. Hellweg and Frischknecht 2004), there is a need to distinguish between short- and long-term emissions in LCA. This temporal differentiation is neces-

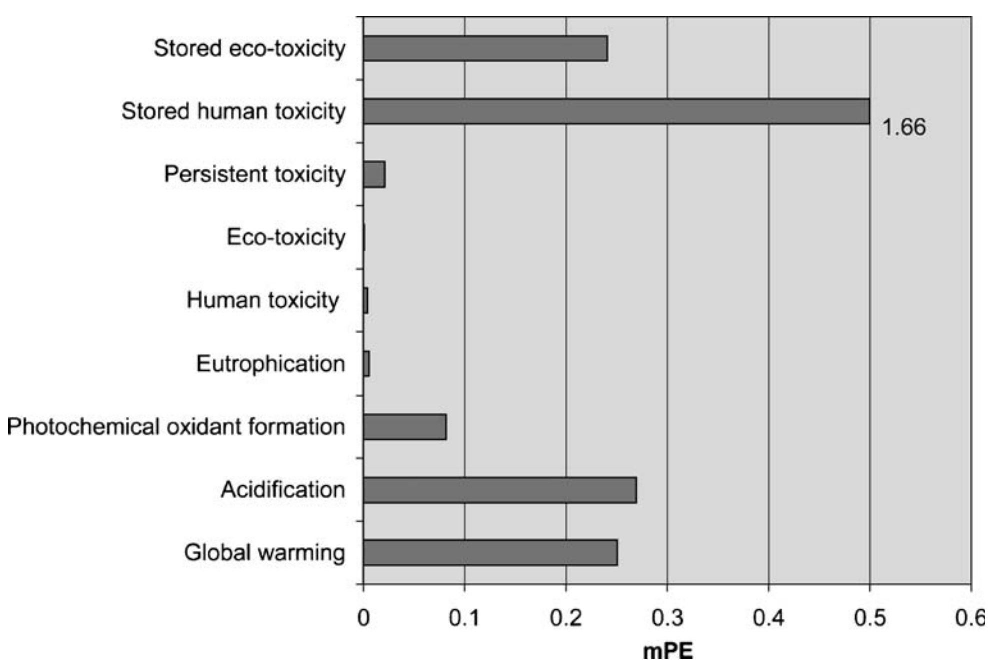

Fig. 1 The normalised environmental impacts from incineration of PVC (1 kg) with stabilisers (0.2% lead and 0.1% zinc) and subsequent landfilling of residues. The unit is milli-person equivalents

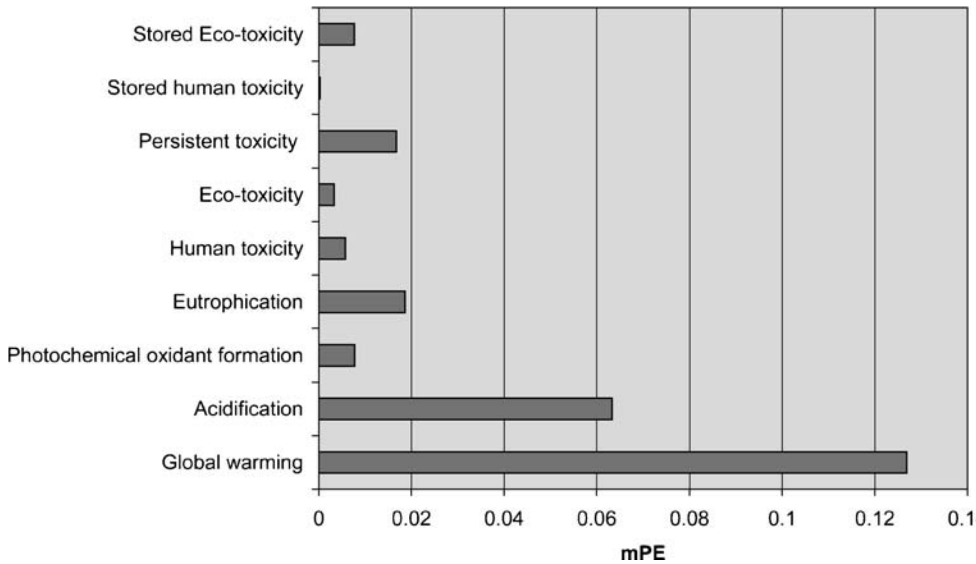

Fig. 2 The normalised environmental impacts of electricity production (1 kWh Danish average grid mix). The unit is milli-person equivalents

sary for scientific reasons, i.e. the higher uncertainties related to actual impacts of long-term emissions compared to short-term emissions. Furthermore, specifically for landfills, there is much evidence that more than 99% of the hazardous substances still remain in the landfill after 100 years.

Basically, there is an inventory problem of quantifying the long-term emissions. In which quantities and forms will the substances in a landfill enter the environment? Such information is a prerequisite for performing the impact assessment. In addition to this lack of knowledge, the impact assessment struggles with its own problems of modelling fate and exposure of the substances in a relevant form and particularly representing the impacts of very slow releases of persistent substances at very low concentrations.

The presented approach for assessment of the long-term impacts of landfilled persistent substances in LCIA proposes to solve the problems related to the meaningful inventory and characterisation modelling of the long-term emissions of persistent substances from landfills by converting it into a weighting and interpretation problem. By developing normalisation references based on inventories of human and ecotoxicity of the annually stored amounts of persistent pollutants (expected to remain stored after 100 years), in the same way as normalisation references are calculated for the other impact categories, the often vast human and ecotoxicity scores associated with persistent pollutants like toxic metals in landfills are scaled down to something that, in principle, is comparable to the impact scores for the conventional human and ecotoxicity categories. Thereby, the problems that may be associated with the presence of these compounds in the landfills are neither forgotten nor grossly overestimated in the assessment. Instead, they are brought forward to the weighting and interpretation steps of the assessment in a form, which allows them to be considered in a conscious way in accordance with the priorities of the main stakeholders of the study.

In summary, the stored toxicity potential approach involves a number of assumptions:

- Human toxic and ecotoxic impacts from long-term emissions (>100 years) of persistent pollutants (in particular metals) should be treated separately from the impacts from short-term emissions (<100 years) due to the large differences in the temporal course of the emissions and the uncertainties about future conditions determining the form of the emission.
- The stored toxicity categories apply the same characterisation factors as the conventional human and ecotoxicity categories as a proxy and assume a distribution 50:50 between water and soil.
- The normalisation of the impact scores applies normalisation references based on inventories for the annual accumulation of persistent pollutants in landfills and the like.

The proposed approach is made operational for the EDIP97 LCIA methodology in this study, but the concept is directly adaptable to other LCIA methodologies, since we use the existing characterisation factors for human and ecotoxicity and just calculate the relevant normalisation references using the inventory of landfilled persistent compounds shown for Denmark in 1994 in Table 1. Currently, the inventories behind the normalisation references are only available for Denmark, but inventories for European normalisation factors might be developed in the same way.

The stored toxicity impact categories have been applied in the impact assessment of the Easewaste software for modelling of solid waste management systems (Christensen et al. 2007).

9 Conclusions and perspectives

The proposed framework/methodology introduces a simple way of handling impacts from long-term emissions of metals and persistent organic compounds from landfills. Acknowledging the high uncertainties related to modelling of release from landfills, the entire amounts of persistent toxicants remaining in the landfill after the foreseeable time horizon of 100 years are included, and a default partitioning representing an equal split between water and soil is assumed. This represents a crude estimate of the potential impacts, but it also ensures that they are taken into account, and the treatment in a separate impact category allows the proper weight to be given to them in the weighting and interpretation of the LCA. The alternative today is either to include them in the other emission-related impact categories (leading to a too strong focus on them) or to leave them out of the assessment by disregarding emissions occurring after 100 years. Landfills and the potential impact of these is an issue of high concern for both public and politicians, and we see the stored toxicity approach as an interim solution, which may convert the problem into a weighting and interpretation problem until a satisfactory solution may be developed for the modelling problems in the inventory and characterisation of these impacts.

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