Assessing Long-term Effects of Municipal Solid Waste

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Abstract

Waste disposal models in product life-cycle assessment studies (LCA) estimate the environmental burdens from the disposal phase of a product. This data *complements* the burdens from other life-cycle phases; manufacture and use. Waste disposal models were created for the Swiss life-cycle inventory (LCI) database ecoinvent (Frischknecht et al. 2003a). These models heed the composition of chemical elements and other characteristics of waste materials, i.e. they aim to be applicable to *specific products and materials in disposal*, not only the average mix of municipal waste.

Burdens from landfills are important within the disposal process chain. Also waste incineration indirectly relies on landfills for the disposal of incineration residues. The emission potential of landfills is significant, if a comprehensive time horizon is applied. Modelling such long time spans is a challenge. Many factors concerning the future of a landfill body are uncertain. Since LCA makes *coarse estimates* of burdens *likely* to occur, and not guaranteed forecasts, similar levels of uncertainty exist in almost every part of LCA. Estimating landfill behaviour is not substantially different. Based on current landfill research and field data, the landfill models created for ecoinvent make a broad, informed estimate of the possible emissions of Swiss landfills including the far future. The inclusion of such long-term emissions in LCA has been discussed controversially in the past. Full discounting of future burdens has been proposed, in cases owing to concepts in economy. In general, LCA has the goal of pointing out least burdening options. The effort to make LCA systems very broad and comprehensive helps to avoid the risk of burden shifting. Discounting future burdens in LCA studies introduces the risk of *burden shifting into the future*. This clashes clearly with ideas of sustainability and intergenerational equity, for which LCA is employed as a tool of perception. Limiting our perception is not going to be helpful. However, landfill models in LCA are coarse and their significance should not be overstated. Like other elements in LCA, they are preliminary. They can and should be improved.

Introduction

Life-cycle assessment studies (LCA) estimate the environmental burdens attributable to a specific product or process. The goal of such studies is to point out least-burdening options, or to optimise one single product. It has long been recognised that LCA studies need to be comprehensive in scope in order to avoid burden-shifting:

- LCA needs to look at **all life-cycle phases of a product** (manufacture, use, disposal) in order to avoid burden shifting from one phase to another. I.e. not a single process is assessed, but the whole *chain of processes* linked to that processes either by necessity or by consequence.
- LCA needs to assess the contributions to **various environmental effects** (e.g. global warming, summer smog, ecotoxic pollutants, resource use) in order to avoid burden shifting from one issue to another.
- LCA needs to assess burdens **irrespective of their location or time of occurrence**, in order to avoid burden shifting from one region to another (e.g. from Western Europe to China) or from one period to another (e.g. from present to the future).
- LCA needs to assess burdens **irrespective of emission concentration** or of acceptable thresholds, in order to avoid burden shifting from a few 'above-threshold' polluters to a dispersed multitude of 'below-threshold' polluters generating the same net pollutant flow.

This is of course a huge undertaking for any product and involves assessments of a large number of processes across most branches of economy. LCA is therefore often forced to simplify procedures by using estimates, average or generic data of processes. **LCA has a *wide focus,* but *shallow depth.*** Instead of assessing each single manufacturer and technology involved in the making of a certain product, industry averages, proxy or generic data can be used to give an estimate of the involved burdens. The available knowledge on manufacturing and use processes is quite large and many experts, like product designers, engineers, maintenance personnel, have extensive knowledge of their product. LCA experts, often engineers themselves, usually have a large reservoir of knowledge to draw from to assess the life cycle phases of manufacture and use; only sometimes hindered by corporate confidentiality. The life cycle phases of *manufacture* and *use* are therefore usually well covered in LCA studies.

In comparison, information about the *disposal* of a particular product is less widespread. In the past, burdens from disposal processes were often not assessed at all, but were concealed behind a sum indicator like 'kilogram waste to disposal'[1]. Thus, a whole life cycle phase was assessed with a single figure. This is as coarse like assessing the *manufacture* of *any* material with one single sum indicator 'one kilogram material from factory', i.e. assuming the manufacture

[1] This kind of assessment makes every waste material uniform. E.g. the burdens from the disposal of one kilogram of electronic equipment will be judged identical to those from one kilogram of kitchen waste, when their content of pollutants and actual disposal technologies and burdens will be very dissimilar.

burdens for *any* material to be uniform. Even today, some renowned LCA studies still use these very coarse, mass-based sum indicators to assess waste disposal, e.g. Boustead (1999).

Disposal models in LCA

To close the data gap regarding disposal in LCA studies, disposal models were created since the mid-1990ies. First efforts were made by Göran Finnveden and Jan-Olov Sundqvist in Sweden (Finnveden et al. 1995, Sundqvist et al. 1997) as well as Peter Zimmermann and Gabor Doka in Switzerland (Zimmermann et al. 1996). The initial focus was on common municipal disposal technologies like landfills, incineration as well as treatment of municipal waste water.

A life-cycle inventory (LCI) of disposal processes should indicate burdens as *waste-specifically* as possible. If a waste material contains no cadmium, then no cadmium emissions from waste shall be inventoried. The disposal models must take into account the *specific composition* of the disposed waste material, as well as other material properties that influence disposal burdens. Apart from these direct burdens, also indirect expenditures should be assessed, e.g. auxiliary material use e.g. for flue gas scrubbers, transports, facility infrastructure. Many disposal processes generate *secondary waste streams*. For example, the incineration of a paper product will generate solid ashes, which in turn will have to be disposed; usually in landfills. A disposal process LCI should be able to a) heed amounts and qualities of secondary waste materials in waste-specific manner; and b) include the burdens generated further downstream in the process chain by the disposal of this secondary waste, sometimes leading to yet further waste streams.

Waste disposal models in the LCI database ecoinvent

The ecoinvent waste disposal models are described in extensive reports[2]. The inventoried disposal technologies include municipal incinerators, hazardous waste incinerators, municipal sanitary landfills[3], bottom ash landfills, residual material landfills, inert material landfills, building dismantling and sorting plants, municipal wastewater treatment plants, subsurface waste deposits (old salt mines) and surface spreading (landfarming). In those inventories, waste-specifity is achieved by heeding the *elemental composition* of each inventoried waste material. Additional information like burnability or degradability is used for particular disposal technologies. The waste disposal models calculate the burdens that originate from the treatment of the waste materials. The behaviour of each chemical element within the disposal process is described using so-called *transfer coefficients*. A transfer coefficient describes what fraction of a pollutant

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[2] The ecoinvent report No. 13, only available to ecoinvent members, describes all models in detail (Doka 2003). An excerpt of the report describing the landfill models is available for free from http://www.texma.org/LCA-Forum/Documentation/documentation.html (Doka 2004). A synoptic article appeared in International Journal of LCA (Doka et al. 2005).

[3] Direct landfilling of untreated municipal solid waste is prohibited in Switzerland since 2000. The models for sanitary landfills are kept in ecoinvent, since this disposal technology is still common in Europe.

inputted into the disposal process will be emitted through a certain output route. Transfer coefficients thus determine the generated emissions or secondary waste flows, c.f. Eq. (1).

$Out_{X,n} = In_X \cdot TK_{X,n}$ *Eq. (1) with*

X = *index for chemical element*
n = *index for output route*
$Out_{X,n}$ = *Output of chemical element X to output route n*
In_X = *Input of chemical element X to disposal process*
$TK_{X,n}$ = *Transfer coefficient for element X to output route n*

Transfer coefficients are derived from average operation data, sometimes adapted to waste characteristics. Since mass conservation applies to chemical elements, the models are conceptually quite straightforward. Emissions of non-conservative, degradable compounds like dioxins or hexachlorobenzene are not inventoried in a waste-specific manner, but sometimes included as a generic *process-specific* emission. Indirect expenditures for auxiliary material use is inventoried as waste-specifically as possible, e.g. an incinerated chlorine-rich waste will lead to larger burdens from acid buffering in flue gas scrubbing than a low chlorine waste.

A closer look at ecoinvent landfill models

Landfills constitute the concluding disposal process for many disposal process chains, i.e. a lot of the ultimate waste streams end up in landfills. Landfills can be important contributors to total disposal burdens of a waste material, cf. Fig. 1. The concepts used to derive the landfill models in the ecoinvent database shall be outlined here, focussing on the less complex cases of inorganic landfills; bottom ash landfills and residual material landfills.

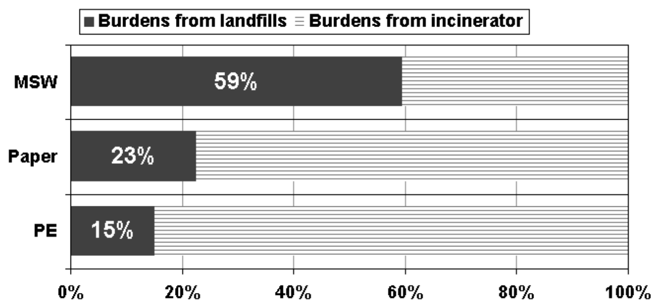

Fig. 1 Contribution from landfill burdens to the total burdens from disposal of municipal solid waste, paper, or polyethylene. 100% are the total disposal burdens expressed with the impact assessment method Eco-indicator'99 (H,A).

The direct emissions from landfilling of a specific waste are derived from Eq. 1, i.e. the emissions to leachate ($Out_{X,n}$, with n = leachate) are calculated from a known waste composition (In_X) using the transfer coefficients ($TK_{X,n}$) for each chemical element[4]. The landfill model

[4] Not all the elements of the periodic table are modelled. The focus is on 41 common and/or toxic elements. These elements are O, H, C, S, N, P, B, Cl, Br, F, I, Ag, As, Ba, Cd, Co, Cr, Cu, Hg, Mn, Mo, Ni, Pb, Sb, Se, Sn, V, Zn, Be, Sc, Sr, Ti, Tl, W, Si, Fe, Ca, Al, K, Mg, and Na.

essentially is set by these transfer coefficients, which describe the *average behaviour of chemical elements* in the landfill.

Several problems arise in determining transfer coefficients for landfills. Landfills are massively heterogeneous systems with complex chemical and hydrological processes, which are far from being understood in detail by landfill research. While landfill operation is limited to e.g. 30 years, the leachate emissions continue far into the future. The landfill model needs to make an estimate about emissions which have not occurred yet. Clearly, any future process cannot be measured today. Therefore, landfill models always have to be derived from certain assumptions.

First approaches in LCA studies to derive landfill transfer coefficients used thermodynamical calculations to simulate long-term landfill behaviour (e.g. Sundqvist et al. 1997, Hellweg 2000). This concept is sound[5], but obstructed by landfill complexity. Thermodynamics is currently unable to model the vast mineral phase complexities encountered in landfills. Solubility determining phases for different elements are often still unknown and models of the known phases fail to predict actually measured field concentrations. Another approach is based on availability tests (e.g. Zimmermann et al. 1996, Frischknecht et al. 1996). Here 24h-laboratory tests are used to determine the leachable amount from a waste material. It is then assumed that this leachable amount is equal or proportional to the future long-term emissions and that the remainder is an inert, unavailable part. This concept is deemed inaccurate by recent findings in landfill research. Given enough time, all phases can be altered by geochemical weathering processes, turned into available phases and emitted. Long-term geochemical weathering cannot be simulated accurately in abridged laboratory set-ups. There is no fundamental stop to leaching in real landfills (see e.g. Sabbas et al. 1998).

To circumvent these problems the landfill models in ecoinvent (Doka 2003) are based on *field measurements* of average leachate concentrations and of average landfill content for each chemical element. The relation between the current leachate concentration and current landfill content quantifies the currently *observed* mobility of each chemical element. This is the starting-point for the landfill model. From there, *future* mobility is estimated heeding key parameters like water flow and pH development. The models are *adapted projections* of the currently observed landfill behaviour. The development of pH is a chief parameter. It is determined by washout of acid buffering carbonate minerals (Huber et al. 1996). For each landfill type a typical duration of the carbonate buffer phase is calculated[6]. Total carbonate washout drops the pH to acidic values, resulting in increased emissions of many heavy metals (e.g. Zn, Pb, Cd). Some elements are unaffected by pH (e.g. Na, Cl), while *oxianion-forming elements* show decreasing

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5 At least for inorganic landfills with no or little biological activity.

6 For sanitary landfills a carbonate phase of 4300 years is calculated, for bottom ash landfills 23'000 years, and for residual material landfills 660'000 years.

mobility with acidic pH (e.g. Cr, Mo, As). Water flow through the landfill is determined by precipitation rates, evapotranspiration rates, preferential flow through the landfill body. All these parameters are heeded in a generic manner representing a Swiss average situation, but not data specific to a single landfill site.

In the sanitary landfill model also air emissions occur. The degradability of waste fractions is used to calculate emissions to landfill gas during the methane phase. Also re-precipitation of degraded material within the landfill is heeded for each chemical element.

Uncertainty in landfill emissions

Uncertainty of inventoried data was addressed all throughout the ecoinvent database, using upper and lower boundary values to indicate an uncertainty range around the inventoried mean value. For landfill emissions the uncertainty range is determined by two factors: a) the uncertainty in waste composition In_X and b) the uncertainty of the transfer coefficient $TK_{X,n}$ (cf. Eq. 1).

Ad a): Uncertainty in waste composition was determined with a literature study on waste material compositions. From a large data sample on elemental compositions a relationship between a concentration and the uncertainty range of that concentration was derived. A lognormal distribution was assumed to be applicable for all uncertainty ranges. It turned out that elements in low concentrations tend to have larger relative uncertainties associated with them than elements in high concentrations[7]. This relationship is reasonable for every material and was used as a generic way to obtain uncertainty data from concentration data.

Ad b): The transfer coefficients describe the landfill behaviour in the far future. Any assertion about the future is uncertain, as many factors influencing that assertion are not known in detail. For the landfill models this can mean e.g. that precipitation or evaporation rates can change, rates of geochemical weathering can alter, earthquakes erosion or flooding events can occur, which all can change the conditions at a landfill site etc. It was beyond the scope of (Doka 2003) to make generic scenarios about such events and an abridged procedure was chosen. In accordance with current landfill research it was assumed that *in a worst case, all landfill contents will be emitted* into the environment (e.g. Sabbas et al. 1998, Johnson 2002). The *upper boundary value* of all transfer coefficients is therefore 100%[8]. The average (mean) value of transfer coefficients is derived from the landfill model without changes in hydrology, but with the projected changes in pH. The model is run to 60'000 years after present. Why 60'000 years?

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7 For example, the uncertainty range of a trace of 100 ppm might be 14 – 700 ppm (a factor 7), while the uncertainty range of a carbon content of 950'000 ppm might be 930'000 – 970'000 ppm (a factor 1.02).

8 Chromium partly constitutes the mineral chromite ($FeCr_2O_4$), which is thermodynamically exceptionally stable and is not accessible by geochemical weathering. The worst case (upper boundary) transfer coefficient of chromium is therefore estimated to be 25%, based on Huber et al. (1996).

After 60'000 years it is estimated that most of Switzerland will be covered by glaciers of the next ice age. Glaciers will thoroughly remodel the surface and ecosphere of Switzerland and environmental conditions will definitely become very different from today's conditions. The period of 60'000 years is proposed as a sort of 'ecological planning horizon'. After that period a distinct environmental interruption will occur. The figure of 60'000 years is a rough estimate based on historic ice age frequencies and does not anticipate current climate change predictions. The best case (lower boundary) value is predetermined by the average value, the upper boundary value and the chosen uncertainty distribution (lognormal)[9].

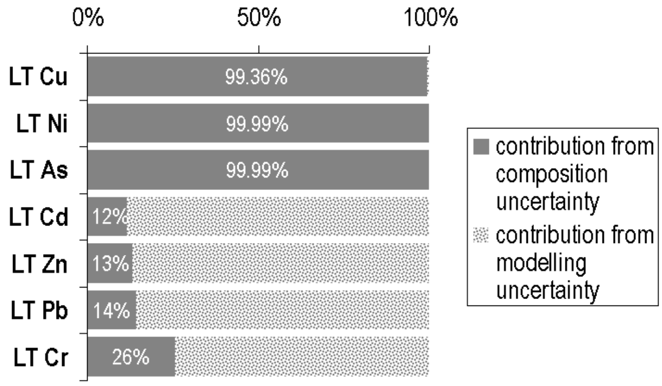

Fig. 2 Contributions to the total uncertainty of some important emissions from the disposal of municipal solid waste. Emission importance decreases top to bottom. LT refers to 'long-term emissions', explained in the next chapter.

Composition uncertainty and landfill behaviour uncertainty make up the uncertainty in the inventory[10]. Composition uncertainty is often dominant, as shown in Fig. 2.

Inventory of long-term emissions

The emissions period of landfills is very long compared to other emission sources, e.g. fuel combustion processes which are virtually instantaneous. In order to be able to analyse these two types of emission, a distinction is made in the inventory. Emissions that occur in the period until 100 years after present are categorised as *short-term emissions*. Most emissions in the ecoinvent database are such short-term or ST emissions. Emissions that occur more than 100 years after present are categorised as *long-term emissions*. In ecoinvent long-term or LT emissions are generated in landfills (Doka 2003) or in sludge ponds of the refining of uranium (Dones et al. 2003), where an emission period of 80'000 years is modelled. This distinction leads to the use of *two* sets of transfer coefficients in each landfill model: a sort-term set to describe to

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9 The use of a triangular distribution, as well as an explicit, rather than implied lower boundary value, was prevented by compatibility issues in the implementation of triangular distributions in the ecoinvent software.

10 Leachate emissions are transported to the groundwater. The required timeframe for this process is small compared to 60'000 years, and leachate emissions are inventoried as groundwater emissions.

emissions until 100 years after present; and a long-term set to describe the emissions afterwards. The burdens from landfilling (cf. Fig. 1) are dominated by long-term emissions, while burdens from incineration are dominated by short-term emissions.

Impact assessment for long-term emissions

After the inventory stage of LCA, impacts of emissions are estimated using impact assessment methods (LCIA). Due to the broad focus of LCA, impact assessments are not site-specific, but generic. For example an LCIA method might indicate the *typical, average* damage of 1 kilogram cadmium emitted to water *anywhere within Western Europe*. LCIA methods are usually based on the current environmental situation in the referred region.

Currently, no LCIA methods exist that provide damage factors for *future* environments[11]. In a future environment, climatic conditions might be altered, which affect the fate of pollutants. Background concentrations of pollutants and/or population densities might be higher, leading to a larger damage per additional amount of pollutant in a more critically burdened environment. These future damage factors would be necessary to assess long-term emissions more accurately. Since no such damage factors exist yet, the available factors for the *present* environment are used as a proxy for the future environment[12]. It is common usage in LCA to fill data gaps with such proxy adaptations. Likewise only a few LCIA methods exist that assess damages in other regions than Western Europe. Since LCI records emissions regardless of location, inventoried emissions can occur anywhere on the globe, e.g. in China. A common solution to this problem is to assess emissions outside Europe with the same damage factors derived for the European situation as a proxy. As emphasised in the introduction, this is done in order to avoid burden shifting from one region to another. The resulting *geographical mismatch* of this proxy solution seems to be accepted.

Neglect of future emissions?

The common practice of weighing future emissions equal to present emissions has been examined in recent years. Hofstetter (1998) examined different value systems from cultural theory, called archetypes, and their relation to value choices made in LCA. With regard to temporal preferences, one archetype, the *Hierarchist*, chooses to weigh damages in the future equally to damages in the present. Another archetype, the *Egalitarian*, weighs damages to the future *higher* than present damages (Hofstetter 1998, p.56). A third archetype, the *Individualist*,

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[11] The valid LCA nomenclature for 'LCIA damage factors' is 'characterisation factors'. The more accessible synonym 'damage factor' is used here instead.

[12] This procedure has been applied in all major LCA software like Gabi, SimaPro, Umberto etc. as well as in both the former Ecoinvent database of 1996 (Frischknecht et al. 1996) and the current ecoinvent database of 2003 (Frischknecht et al. 2003a).

places his own well-being above all else and weighs damages to the future *lower* than present damages. Hofstetter also points out that there are other archetypes from cultural theory, which are not relevant in LCA, since they make no decisions or would not use LCA as a tool for their decisions; the *Fatalist* and the *Autonomist* (Hofstetter 1998, p.76). Hofstetter's concepts were adopted in the LCIA method Eco-indicator'99, resulting in three sets of damage factors (Goedkoop et al. 2000). The authors suggest the set based on the Hierarchist to be used as a default. The Individualist set of Eco-indicator'99 counts only short-term damages and completely neglects long-term damages. Curiously, the Egalitarian set does not weigh future emissions higher than present emissions, as suggested by Hofstetter (1998, p.56), but equally, like the Hierarchist.

Hellweg (2000) and Hellweg et al. (2002) examined the motivations to neglect or discount future burdens. They find that discounting of future effects is frequent in everyday decisions for reasons of *pure time preference*[13]. Another reason for discounting is assuming increasing *productivity of capital*. Detrimental environmental effects in the future might be much cheaper to remediate than the same effects today, expressed in current money values. However, no financial mechanism exists that would allow remediation of future damages caused by the present generation, e.g. an intergenerational clean up fund. Also the severity of caused damage might increase in the future. For example, a future kidney transplant due to heavy metal poisoning might be perceived *more detrimental* than a kidney transplant today for technical reasons, or because future generations could demand a higher compensation for damage, if the real per-capita-income increases. Hellweg et al. (2002) conclude that discounting of future burdens because of pure time preference is unethical and clashes with sustainability principles. If LCA results are sensitive to discounting choices, then a thorough reasoning for these choices should be given.

Discounting of future burdens was discussed controversially within in ecoinvent database project (Frischknecht et al. 2003b) and again at the 22nd LCA discussion forum in Zurich[14]. Some of the arguments made by a minority supporting the possibility of discounting future burdens shall be presented and commented here. One argument states that our present problems could be eclipsed by future burdens. This is a statement of pure time preference ("Our problems *must be* more important than those of future generations.") and clashes with fundamental ethical values. Another pro-discounting argument is that future burdens are believed to be more uncertain than present burdens and therefore deserve less attention. With regard to the precautionary principle

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[13] I.e. a person will rather have a profit today than tomorrow. A profit today will therefore be more valuable than a profit tomorrow. By symmetry, costs today weigh more than the equal cost tomorrow. Environmental burdens can be understood as such costs, and future burdens are therefore less important than present burdens.

[14] On May 7th 2004. See the web site http://www.texma.org/LCA-Forum/lca-forum.html for further information and documentation. See also (Doka et al. 2005).

and the synoptic character of LCA, also uncertain burdens always have been included in LCA studies. It seems biased and *ad-hoc* to exclude some allegedly uncertain burdens, while at the same time other uncertain burdens are commonly included[15]. The issue of uncertainty shall be dealt with quantitative uncertainty analysis[16]. Another argument is that decision makers might not accept results that are influenced by far future burdens. LCA is a *perception tool* to help us grasp environmental burdens in a uniquely synoptic way. This unusual perspective is the *chief advantage* and *core purpose* of LCA, sometimes leading to surprises, as novel perspectives are inclined produce. This concept should not be lightly abandoned to comply with beforehand expectations of non-experts. Instead the purpose and concepts of LCA should be communicated clearly.

Conclusion

Landfills take up the waste materials of today and carry them into an unknown future. During that time, emissions will occur from the transformation of the landfill contents. Landfills are a means to avoid burdens today and move them far into the future. Those future burdens should be revealed by LCA, and coarse models to do that were outlined here. The mere shifting of burdens into the future cannot be regarded as a proper and sustainable solution of environmental problems. If an LCA study allows burden-shifting into the future to be presented and perceived as a good and sustainable solution of environmental problems, it has failed to fulfil the very goal of LCA: synopsis and avoidance of burden-shifting. Since the neglect to account for future burdens increases the risk of burden shifting into the future, this neglect cannot be part of LCA. **Many value systems and subjective choices are common and unavoidable in LCA. But value systems that consciously contradict the goals of LCA are not tolerable.** I would therefore propose that any value choices that lead to complete neglect of future burdens in LCA, for whatever reason, are not admissible; for example the value choices expressed by the Individualist archetype of cultural theory. Such choices are conceivable in other areas with dissimilar goals. If negligence of the future due to pure time preference is deemed an *unethical* behaviour (Hellweg et al. 2002), then LCA practitioners should not accept such choices, if they want to adhere to an ethical conduct; even if a commissioner of a study would be inclined to make such choices.

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[15] For example, the effect of summersmog pollutants is heavily influenced by the current weather conditions at the time of release, like sunshine intensity. A release of VOCs might have a full summersmog effect on a bright summer day, and virtually none on a rainy day. Nonetheless such very uncertain burdens are included in LCA studies unchallenged and on a regular basis.

[16] First steps to include quantitative uncertainty information were taken in the new ecoinvent database. A preliminary analysis showed that the datasets for landfill processes are not fundamentally more uncertain than the other datasets in ecoinvent (Doka 2005).

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