ecoinvent: Services

Waste Treatment and Assessment of Long-Term Emissions

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Abstract

Goal, Scope and Background. The disposal phase of a product's life cycle in LCA is often neglected or based on coarse indicators like 'kilogram waste'. The goal of report No. 13 of the ecoinvent project (Doka 2003) is to create detailed Life Cycle Inventories of waste disposal processes. The purpose of this paper is to give an overview of the models behind the waste disposal inventories in ecoinvent, to present exemplary results and to discuss the assessment of long-term emissions. This paper does not present a particular LCA study. Inventories are compiled for many different materials and various disposal technologies. Considered disposal technologies are municipal incineration and different landfill types, including sanitary landfills, hazardous waste incineration, waste deposits in deep salt mines, surface spreading of sludges, municipal wastewater treatment, and building dismantling. The inventoried technologies are largely based on Swiss plants. Inventories can be used for assessment of the disposal of common, generic waste materials like paper, plastics, packaging etc. Inventories are also used within the ecoinvent database itself to inventory the disposal of specific wastes generated during the production phase. Inventories relate as far as possible to the specific chemical composition of the waste material (waste-specific burdens). Certain expenditures are not related to the waste composition and are inventoried with average values (process-specific burdens).

Methods. The disposal models are based on previous work, partly used in earlier versions of ecoinvent/ETH LCI data. Important improvements were the extension of the number of considered chemical elements to 41 throughout all disposal models and new landfill models based on field data. New inventories are compiled for waste deposits in deep salt mines and building material disposal. Along with the ecoinvent data and the reports, also Excel-based software tools were created, which allow ecoinvent members to calculate waste disposal inventories from arbitrary waste compositions. The modelling of long-term emissions from landfills is a crucial part in any waste disposal process. In ecoinvent long-term emissions are defined as emissions occurring 100 years after present. They are reported in separate emission categories. The landfill inventories include long-term emissions with a time horizon of 60'000 years after present.

Results and Discussion. As in earlier studies, the landfills prove to be generally relevant disposal processes, as also incineration and wastewater treatment processes produce landfilled wastes. Heavy metals tend to concentrate in landfills and are washed out to a varying degree over time. Long-term emissions usually represent an important burden from landfills. Comparisons between burdens from production of materials and the burdens from their disposal show that disposal has a certain relevance.

Conclusion. The disposal phase should by default be included in LCA studies. The use of a material not only necessitates its production, but also requires its disposal. The created inventories and user tools facilitate heeding the disposal phase with a similar level of detail as production processes. The risk of LCA-based decisions shifting burdens from the production or use phase to the disposal phase because of data gaps can therefore be diminished.

Recommendation and Perspective. Future improvements should include the modelling of metal ore refining waste (tailings) which is currently neglected in ecoinvent, but is likely to be relevant for metals production. The disposal technologies considered here are those of developed Western countries. Disposal in other parts of the World can differ distinctly, for logistic, climatic and economic reasons. The cross-examination of landfill models to LCIA soil fate models could be advantageous. Currently only chemical elements, like copper, zinc, nitrogen etc. are heeded by the disposal models. A possible extension could be the modelling of the behaviour of chemical compounds, like dioxins or other hydrocarbons.

Keywords: Bottom ash landfill; building dismantling; ecoinvent; hazardous waste incineration; municipal incineration; municipal wastewater treatment; residual material landfill; sanitary landfill; subsurface waste deposits; Switzerland; waste disposal inventories; waste-specific inventories

1 Motivation

Waste disposal is a somewhat neglected part in life cycle inventories. For production and use phases elaborate data gathering is performed, but the disposal phase receives often comparatively inferior treatment. For example, mere waste masses are summed up into a single figure, or masses of a few waste types are inventoried (see e.g. Boustead 1999). Such parameters pose an unnecessary complication in the valuation step of LCA. Seeing that waste disposal processes are indeed man-made technosphere processes that emit pollutants like any other technosphere process, there seems no need to use extraordinary inventory parameters like 'waste mass' for disposal processes.

A comparatively better way to inventory waste disposal is to use *average* disposal emissions. For example for waste incineration the air emissions, water emissions and auxiliary material inputs during average operation can be inventoried per kilogram of waste treated. However, waste incinerators treat very dissimilar waste materials like paper, plastic, kitchen waste, glass, or cans. These different materials will cause very different emissions when incinerated. So, using *average* incinerator data for all these materials is by comparison as accurate as if the burden from the production of any industrial material (metal, plastic, paper etc.) were inventoried using the emissions of one average industry mix.

Disposal emissions should be calculated in a way which is as detailed as the production and use phase. This means to heed the *waste-specific* burdens generated by a specific waste composition. With data from substance flow analysis in disposal processes, it has become feasible to model the fate of single chemical elements in disposal processes an hence to calculate waste-specific disposal inventories. Such inventories were first calculated in Frischknecht et al. (1994), Zimmermann et al. (1996) and Sundqvist et al. (1997). The aim of this part of the ecoinvent project is to create waste-specific life cycle inventories for different waste compositions, based on the Swiss disposal processes (Doka 2003).

2 Scope

The technologies inventoried in ecoinvent are based on plants in Switzerland and Germany. Switzerland banned the landfilling of burnable wastes in 2000. The landfilling of burnable wastes in sanitary landfills is inventoried nonetheless, since such practices are common in Europe. Switzerland exports some of its hazardous waste to subsurface deposits located in old German salt mines. Inventories for subsurface deposits are based on the Herfa-Neurode deposit. Wastewater treatment is only applicable to municipal sewage, not to sewage dominated by industrial processes. Recycling processes are generally not considered in these inventories. However, for building material disposal, a sorting plant is inventoried which can be seen as a prerequisite for recycling. Production processes with recyclate inputs heed recycling processes in addition to primary material input; e.g. paper recycling is inventoried in the pulp and paper production inventories of ecoinvent. The treatment and disposal of radioactive wastes is inventoried in the nuclear power production chain. Along with the ecoinvent data and the reports, also Excel-based *software tools* were created which allow the ecoinvent member to calculate waste disposal inventories from arbitrary waste compositions. Only waste compositions given as chemical elements like copper, zinc, nitrogen, carbon etc. are modelled. The fate of individual chemical substances (e.g. PCB, hexachlorobenzene) is not modelled[1].

Disposal processes deliver the service of waste treatment. The reference flow for waste disposal datasets is 'kilogram waste'; except for wastewater treatment where 'cubic-metres wastewater' is used. The waste composition is defined as concentrations of 41 chemical elements[2]. Additional parameters like water content, heating values, burnability, and overall degradability in sanitary landfill are used to characterise the waste.

[1] In some models such compounds are heeded as a generic, waste-independent emission. For wastewater treatment the generic fate of 'compounds' such as COD, BOD, DOC, TOC, NH_4, NO_3, PO_4, SO_4, is heeded.

[2] The heeded 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.

3 Disposal Models

Starting with a given waste composition, the disposal models need to establish what the behaviour of the waste input is during the disposal process. To describe the disposal behaviour, so called transfer coefficients (TK) are used (Eq. 1). A transfer coefficient describes what fraction of a pollutant inputted into the disposal process will be emitted through a certain output route, e.g. emissions to air. Pollutants are assessed only as chemical elements. The fate of chemical compounds (e.g. dioxins, PAH etc.) cannot be modelled without individual decomposition data.

$$Out_{X,n} = In_X \cdot TK_{X,n} \tag{1}$$

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

Waste compositions are taken from literature, manufacturers, or theoretical considerations (Doka 2003). Transfer coefficients are derived from average operation. For incinerators and sanitary landfills the transfer coefficients are additionally adapted to waste materials heeding waste characteristics like burnability and degradability. The output routes n of Eq. (1) can be emissions to air or to water, or the generation of secondary waste materials like bottom ash in incineration. In order to assess the *complete* burden of a disposal technology, the generated secondary waste streams need to be inventoried as well. For example, incinerator bottom ash is landfilled. The concept of waste-specific inventories requires that all secondary waste streams differ according to the original waste input; i.e. bottom ash from incineration of PVC will be different in quantity and quality than bottom ash from incineration of paper[3]. Certain expenditures are not related to the waste composition and are inventoried with average values, e.g. emissions of carbon monoxide or dioxin. These are not waste-specific, but *process-specific exchanges*.

3.1 System boundaries and data structure

Disposal inventories contain the burdens from the treatment of the original waste, but also from all generated subsequent waste steams. For example, incineration inventories include the burdens from the landfilling of resulting incineration residues (**Fig. 1**, box D). Similarly, in a sanitary landfill leachate is collected and treated in a wastewater treatment plant, which produces a sludge, which in turn is incinerated. All these downstream burdens are included in the inventory of sanitary landfilling. For disposal processes, cumulative data includes all *downstream* burdens. In contrast, cumulative data of *production processes* includes all *upstream* burdens (**Fig. 1**, box P). The combination of production inventories and disposal in-

[3] These examples of waste-specific bottom ashes must be thought of as imaginary portions of the average bottom ash, which are attributable to certain co-incinerated waste fractions. It is not suggested that waste materials are incinerated separately.

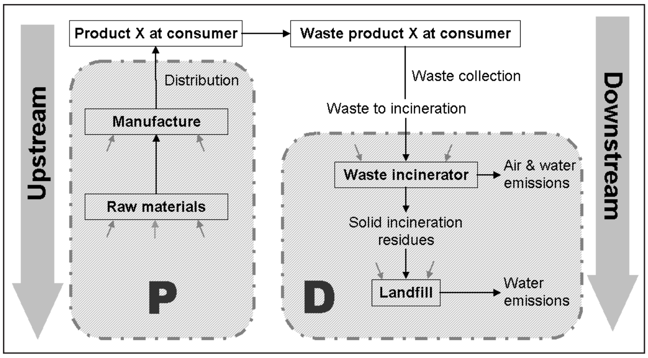

Fig. 1: Schematic data structure and system boundaries for disposal processes (box D on the right), as opposed to production processes (box P on the left). Lines with arrows indicate mass flows

ventories (plus use phase data) then results in the complete life cycle inventory of the material. Of course, disposal inventories also contain some upstream burdens, like manufacture of auxiliary materials and plant infrastructure.

Waste collection is *not* included in ecoinvent disposal inventories (**Fig. 1**, boundary of box D). The two exceptions to this are building demolition inventories where the building dismantling and subsequent transport and disposal are included, as well as wastewater treatment where transport in sewers is included. For municipal solid waste collection an inventory per ton-kilometre is provided as a separate dataset in the ecoinvent database.

In incineration and sanitary landfills the energy produced from the waste is calculated waste-specifically based on the given lower heating value. High calorific wastes are able to produce their own process energy, or in the case of high-temperature hazardous waste incineration reduce the amount of complementary fuel input. Any net energy generation is treated as a co-product of waste incineration. The used default allocation is 0% on energy production and 100% on waste incineration service, based on proceeds. With default allocation, energy from waste incinerators is virtually burden-free, which corresponds well with the notion that the consumption and generation of such energy is environmentally favourable. This advantage is attributed to the *consumer* of waste incinerator energy and not to the waste creator. The data format of the ecoinvent database (EcoSpold) allows the user to alter these default allocation factors.

3.2 Overview of the different disposal inventories

The database ecoinvent data v 1.1 contains 283 different disposal inventories. **Table 1** gives an overview of the different disposal technologies and the processes heeded in the respective inventories.

Table 1: Overview of the different disposal technologies in ecoinvent and heeded processes in these inventories

Disposal technology	Heeded processes in ecoinvent (Doka 2003)
Inert material landfill (IMF, 'cleanfill')	Land use and infrastructure. Process energy. Leachate emissions are considered to be negligible.
Residual material landfill [a] (RMLF)	Land use and infrastructure. Process energy. Specific projected leachate emissions over 60'000 years.
Bottom ash landfill [a] (for MSWI bottom ash only, 'slag compartment')	Land use and infrastructure. Process energy. Specific projected leachate emissions over 60'000 years.
Sanitary landfill [a] for municipal solid waste (MSWLF)	Emissions via landfill gas. Land use and infrastructure. Process energy and partial utilisation of landfill gas as fuel. Specific projected leachate emissions over 60'000 years heeding the degradability of waste materials. Treatment of leachate over the first 100 years in a municipal wastewater treatment plant including all further downstream burdens (see there).
Municipal solid waste incineration [a] (MWSI)	Air emissions from grate incineration after purification (electrostatic precipitation, flue gas scrubber, DeNOx). Water emissions after scrubber liquid treatment. Land use and infrastructure. Process energy and partial incineration heat utilisation. Auxiliary materials and water consumption. Transport and landfilling of solid residues (bottom ash to bottom ash landfill; flue gas ashes, precipitator ashes and scrubber sludge solidified with cement to residual material landfill) including all further downstream burdens (see there).
Hazardous waste incineration [a] (HWI)	Air emissions from enforced rotary kiln incineration after purification (flue gas scrubber, DeNOx), water emissions after scrubber liquid treatment, land use, infrastructure, process energy and partial incineration with heat utilisation. Fuel oil, auxiliary materials and water consumption. Transport and landfilling of solid residues (scrubber sludge solidified with cement to residual material landfill) including all further downstream burdens (see there).
Municipal wastewater treatment [a] (WWT)	Transport of wastewater in sewers. Infrastructure sewers. Overload discharge. Three stage treatment of wastewater. Water emissions in treated water. Digestion of wastewater treatment sludge including metal volatilisation. Land use and infrastructure for five different plant size classes. Process energy and incineration of digester gas with internal energy utilisation. Auxiliary materials consumption. Transport and disposal of digester sludge as a mix of direct surface spreading (landfarming) or incineration in a municipal solid waste incinerator including all further downstream burdens (see there). Landfarming considers local fate factors for nitrogen and phosphorus heeding air emissions and plant uptake.
Building material disposal	See also Althaus et al. (2004) in this issue. Dismantling of building materials (process energy, dust emissions) and disposal in three scenarios: A. Direct recycling (cut-off), B. sorting plant (transport, sorting process energy, land use and infrastructure, disposal of non-recyclable parts (fine fraction) in sanitary landfill), C. Direct disposal in inert material landfill (for mineral materials) or municipal solid waste incinerator (for burnable materials) including all further downstream burdens.
Surface spreading	Direct spreading on industrial soil. Complete and direct emission to soil. Energy consumption, infrastructure and emissions of spreading process.
Subsurface waste deposits	Deposition in former potassium salt mines in Germany. Packaging of waste. Infrastructure. Process energy. No direct emissions from the deposited waste are inventoried.

[a] For these technologies Excel calculation tools are provided to members of ecoinvent for own calculations.

3.3 Landfill models

Waste landfills are massively heterogeneous systems with complex chemical and hydrological processes which are not fully understood. While landfill operation is limited to e.g. 30 years, the emissions from the landfill continue for centuries or millennia. How can these future emissions be inventoried for LCA? Obviously, any future process cannot be *measured* today. The future emissions of landfills have to be *modelled* and the landfill models are always based on certain assumptions.

In earlier LCI landfill models, thermodynamical calculations were used to simulate long-term landfill behaviour (e.g. Hellweg 2001, Sundqvist et al. 1997). This concept is sound[4], however currently thwarted by landfill complexity. Thermodynamics is unable to model the vast phase complexities encountered in landfills. Solubility determining phases for different elements are often still unknown and known phases fail to predict measured concentrations. Alternative landfill models were based on availability tests. 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 long-term emissions (Zimmermann et al. 1996, Frischknecht et al. 1996). This concept is deemed inaccurate by recent findings in landfill research. Given enough time *all phases* can be altered by geochemical weathering processes and turned into available phases. 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).

Within the ecoinvent project new landfill models are developed. Three landfill types are distinguished: 1) sanitary landfills, 2) bottom ash landfills, 3) residual material landfills. Data on field measurements of leachate concentrations and landfill contents for each landfill type from a literature survey was compiled (Doka 2003). The relation of leachate concentration vs. landfill content represents the *currently observed landfill behaviour.* Adaptive projections into the future are made, which heed key parameters like annual water flow and pH development. This is a top-down approach. The model is calibrated to Swiss climate for precipitation and evapotranspiration rates and also heeds preferential flow of water transport through the landfill body. Development of pH 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[5]. 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* mobility with acidic pH (e.g. Cr, Mo, As). The model is chiefly based on original concepts by the author described in detail in Doka (2003) and in part uses adaptations of earlier models such as Belevi & Baccini (1989).

[4] At least for inorganic landfills with no or little biological activity.

[5] 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.

In the sanitary landfill model the degradability of waste fractions is used to calculate emissions to landfill gas during the methane phase. New data was used to derive degradability of wastes. An addition to the sanitary landfill model is the so called release factor, which heeds the re-precipitation of degraded material *within* the landfill. Re-precipitation delays emissions into the future. Different release factors are calculated for each chemical element and are calibrated according to field measurements of actually occurring landfill emissions.

3.4 Inventory of long-term emissions

Landfills emit pollutants for very long time spans. In LCA emissions are generally integrated over time and presented as one emission flux per functional unit. The long time spans involved in landfill emissions has prompted a coarse time resolution for emission categories, which was already introduced with the 1996 version of the ETH/Ecoinvent database[6]. In the ecoinvent database emissions are also divided into *short-term* and *long-term* emissions, denoted ST and LT respectively. Short-term emissions occur between now and the arbitrarily chosen limit of 100 years after present; long-term emissions occur more than 100 years after present. Short-term and long-term emissions are reported in separate emission sub-categories. The only sources of long-term emissions in the ecoinvent database are landfilling processes and uranium production[7].

In the ecoinvent database, each inventory figure is given as *a mean value*, but also an *uncertainty range* is inventoried (Frischknecht et al. 2004). For long-term landfill emissions following concept was applied. For the worst case or upper boundary value of the long-term emissions a total release from the landfill was assumed[8]. This expresses the potential complete weathering of the landfill contents, as suggested by landfill research. For the mean value of the long-term emissions the releases of 60'000 years after present are integrated. Why 60'000 years? 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. Landfills will be eroded and their contents redistributed in the landscape. 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 uncertainty of the transfer coefficients regarding the future landfill development is combined with the uncertainty in the composition of the waste materials to result in the total uncertainty of the emission.

Long-term releases from landfills are expected to be emitted into the soil *below* the landfill, not to the surface. Man-

[6] See Annex F, page 2, of Frischknecht et al. (1996)

[7] Processing residues from uranium ores release radioactive radon 222Rn for a very long time. In ecoinvent 2000 these emissions are inventoried over 80'000 years.

[8] The only known exception to this is chromium which partly forms the mineral chromite ($FeCr_2O_4$), which is thermodynamically exceptionally stable and is not accessible by geochemical weathering. The maximal leachable amount of chromium is therefore estimated to be 25%, based on Huber et al. (1996).

made barriers like landfill base seals and clay layers are considered to be functional only over a few decades (Baccini et al. 1992, Frühwirth et al. 2000). In soil, the pollutants are reversibly adsorbed to the soil matrix, but ultimately are discharged to the groundwater. The time lag for emissions to reach the groundwater was calculated to be on average 1000 years. This is negligible compared to the inventoried time frame of 60'000 years. Landfill long-term emissions are therefore inventoried as emissions to groundwater.

The long-term stability of underground deposits in salt mines was investigated. It is a common fate for salt mines after closure to be flooded with water entering from layers above. This increases the mobilisation of pollutants. There is concrete evidence that mining shafts compromise the mechanical stability of geological layers. It can be concluded that the risk of a future pollutants release from underground deposits is non-zero. It was decided however not to heed any direct emissions from underground deposits in the inventory, since a) scarce risks are not included in ecoinvent LCI and b) no emissions occur today.

3.5 Impact assessment of long-term emissions

Until now impact assessment methods have usually not specified how to deal with emissions occurring in the far future.[9] In the past long-term emissions have been valuated just like short-term emissions. Only Eco-indicator'99 provides a particular weighting set for the individualistic archetype, who only considers short-term impacts and neglects any long-term impacts. A lot of arguments can be made regarding the inclusion or exclusion of long-term impacts from LCIA results. An overview is given on pages 5ff. of Frischknecht et al. (2003); see also Hellweg (2003), Finnveden et al. (1999). No consensus amongst the ecoinvent administrators was achieved. The ecoinvent administrators decided in summer 2003 by majority vote to apply the same characterisation factors for long-term emissions as for short-term emissions, unless an LCIA method provides applicable recommendations. Two of the most important arguments for this course of action were that the holistic approach of LCA contradicts the cutting-off of emissions caused by the assessed system and the notion that default exclusion of long-term impacts is contrary to intergenerational equity expressed by key ethical values and sustainability concepts. Two of the most important arguments against this decision were the lack of knowledge of the future in terms of environmental fate and manageability of pollutants and the lack of common acceptance of such a long-term perspective in decision making. The administrator group agreed that further research and discussion on this question is necessary. Since summer 2003, the discussion on assessment of long-term emissions has been continued in the 22nd LCA discussion forum.[10]

Consequently, in ecoinvent LCIA results *the impacts from long-term emissions are included by default*, like in preceding versions. The only exception is the individualistic perspective of Eco-indicator'99 (I, I) where only the impact of short-term emissions is assessed. The applied characterisation factors of LCIA methods are based on simulations of *current* environmental conditions. Applying characterisation factors to short-term and long-term emissions alike, presupposes that the *future environmental conditions* are identical to the current environmental conditions. Obviously, this is a flawed presumption, as background concentrations and also the environments pollutant buffering capacity can change over time. For a more accurate impact assessment of long-term emissions characterisation factors based on a *future* environment would be needed. Such future factors are currently not available. Instead the available factors are also applied to long-term emissions as a proxy. A similar approximation is common in LCA for geographical differences: Characterisation factors derived for European countries are routinely applied to emissions occurring in all regions of the world. The resulting *geographic mismatch* of characterisation factors is commonly accepted by LCA practitioners. The *temporal mismatch* of applying current characterisation factors to long-term emissions is quite lower than these geographic mismatches. I.e. the possible *future state* of e.g. the Swiss environment (temporal variety) is well within the range of the various environmental states *currently* encountered on the planet (geographical variety). Until more detailed characterisation factors are available, these mismatches and the resulting inaccuracies seem acceptable.

4 Results and Discussion

4.1 Is disposal relevant in the Life Cycle?

Disposal is part of a product's life cycle and should be included in LCA. But how relevant is the disposal phase compared to the production phase? For building materials a comparison of the production phase versus disposal phase is given in this issue (Althaus et al. 2004). Results for polyvinylchloride PVC, polyethylene PE, and paper with disposal in municipal incinerators are exemplified with life cycle results with impact assessment according to Eco-indicator'99 (H,A) shown in **Fig. 2**. Disposal has a certain importance in the life cycle burdens, while production usually remains the dominant phase.

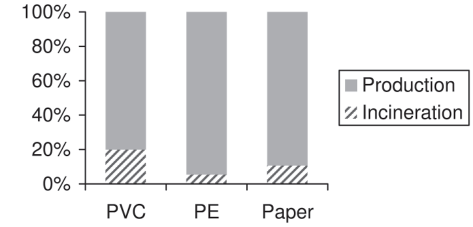

Fig. 2: Comparison of Eco-indicator'99 (H,A) impact scores from production and waste incineration of three materials. (H,A) refers to the default Hierarchist perspective in Eco-indicator'99 with an average weighting (Goedkoop et al. 1999)

[9] This is perceived as a limitation of LCIA as explicitely stated in clause 8 of ISO 14042.

[10] On May 7th 2004. See the website http://www.texma.org/LCA-Forum/lca-forum.html for further information and documentation.

4.2 Contribution analysis

How are the burdens of disposal generated? The dominance of contributions is exemplified for municipal incineration of polyethylene, paper and average municipal solid waste, MSW, with impact assessment according to Eco-indicator'99 (H,A) in **Fig. 3**. Air emissions from the incinerator and long-term emissions from the landfilling of incineration residues are dominant contributions to the overall score for incineration.

For the wastes in Fig. 3 the short-term emissions to air are dominated by CO_2 emissions, while other common air pollutants like NO_x, SO_x, NMVOC or methane play a minor role. This reflects the extensive flue gas treatment performed in the incinerator. The burden from long-term emissions is determined by two factors 1) 'How much of the waste is transferred to the landfilled incineration remains?' and 2) 'What pollutants are contained in these remains?'. A significant part of average MSW is inert in combustion and will be transferred directly to the bottom ash (14 w%). But also burnable, low-ash wastes can yield bottom ash, but also fly ash and scrubber sludge from the flue gas treatment. Per kilogram of average MSW 190 grams of bottom ash are produced and 14 grams of other solid residues. These quantities also contain oxygen added from oxidation processes. By comparison, incineration of 1 kilogram paper produces 80 + 12 grams of solid remains, 1 kilogram PE produces 19 + 6 grams of solid remains. A large part of the long-term burdens of MSW are determined by copper emissions. Average MSW contains 1200 ppm copper and 1090 ppm are released in landfills. For paper and PE also other elements like cadmium and arsenic become important. Burdens from auxiliary materials reflect the expenditures in flue gas purification steps and cement for the solidification of flue gas treatment residues. Of these, solidification cement is the most important contributor followed by consumption of sodium hydroxide, used to neutralise acidic flue gases. The burden from auxiliary materials usually increases with the mass of fly ashes and scrubber residues generated. The exact amounts are determined by the requirements for treatment, which in turn are determined waste-specifically by the composition of the ash and residue. Transport also reflects the amounts of generated unburnable residues, as it is used for transport of slag and residues to the landfills and the transport of auxiliary materials. The contribution 'proc.' combines the process-specific burdens from MSWI and landfills. It is essentially dominated by the process-specific burdens from MSWI. All these statements are based on the three examples presented here and valuation with Eco-indicator'99 (H,A). For other wastes with other compositions or other LCIA methods different inventory parts can become dominant.

4.3 Uncertainty analysis

Within the ecoinvent database first efforts were made to consistently quantify uncertainties in LCI (Frischknecht et al. 2004). Each inventory exchange is recorded along with a range indicating the uncertainty of the inventoried mean value. The uncertainties of cumulative inventories (inventory results) are calculated with Monte-Carlo analysis, publicly available in version 1.1 of the database. Monte-Carlo analysis generates a mean value and an upper and lower boundary value for each LCI exchange. From these three figures it is possible to estimate the influence of exchange uncertainty on the resulting impact assessment result. This is exemplified in **Fig. 4** for the incineration of average municipal solid waste.

The mean inventory values for the incineration of average municipal solid waste are weighted with the according characterisation factors from the LCIA method Eco'indicator'99 (H,A). Ranking then establishes the dominance of each emis-

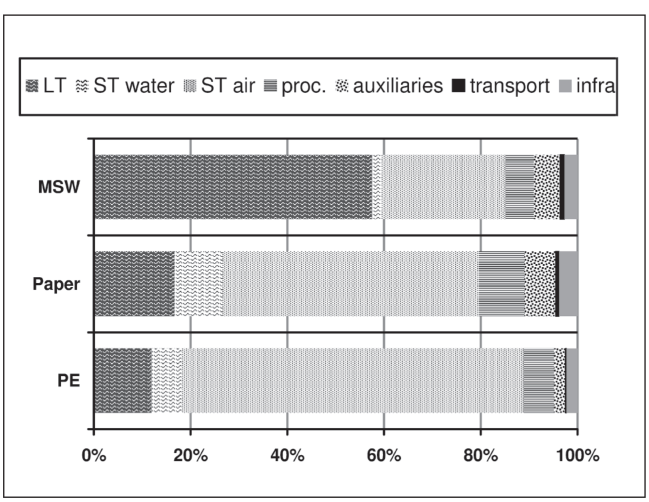

Fig. 3: Contributions to Eco-indicator'99 (H, A) impact scores for incineration of three exemplary materials. LT: long-term emissions from landfilling of incineration residues; ST water: short-term emissions to water from the incinerator and the landfills; ST air: short-term emissions to air from the incinerator; proc.: emissions and burdens which are independent of waste composition; auxiliaries: auxiliary material inputs for flue gas treatment and solidification cement; transport: transport services between incinerator and landfills and for auxiliary materials; infra: Infrastructure for incinerator and landfills

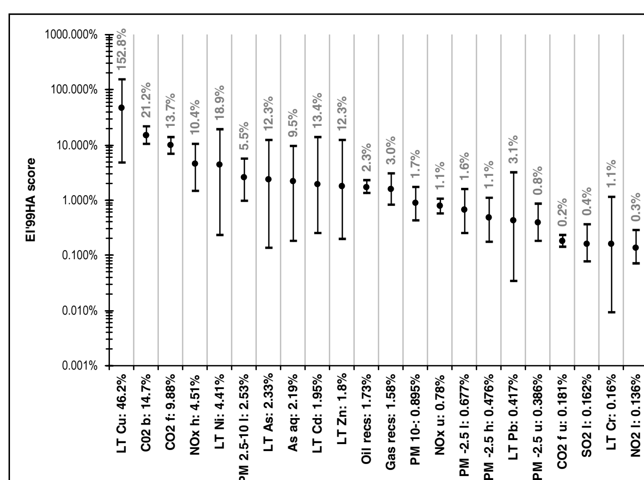

Fig. 4: Cumulative uncertainty in the 22 most relevant emissions for the incineration of average municipal solid waste MSW. 100% is the total mean impact score as measured by Eco'indicator'99 (H, A) for 1 kg MSW. Percentages in the x-axis labels indicate dominance of the *mean value*. Percentages within the chart area indicate the dominance of the *upper boundary value*. Abbreviations in the x-axis labels discern the ecoinvent subcategories of emissions, where LT: long-term emissions to groundwater; aq: to water; b: biogenic; f: fossil; h: high population density air emission; l: low population density air emission; u: unspecified air emission

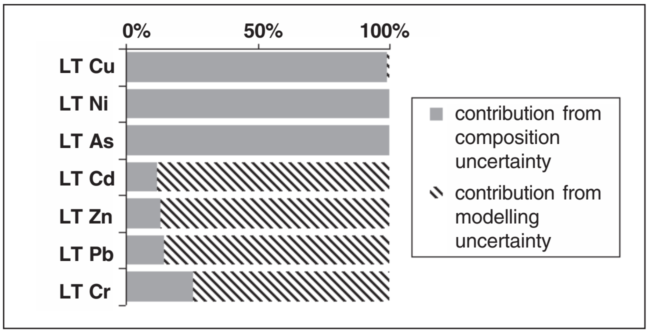

Fig. 5: Sources of uncertainty of the long-term emissions of Fig. 4. Contributions for the initial waste input uncertainty and the disposal model uncertainty

sion (left to right in Fig. 4; impact scores are given in percentages of the total mean score). Long-term copper emissions from the landfilling of incineration residues are the single most important emission in this example, which alone bears 46.2% of the total burden (LT Cu in Fig. 4). The upper and lower boundary values for each emission can be weighted too and displayed as ranges, which indicate the cumulative uncertainty of these emissions. The uncertainty ranges are considerable (note the logarithmic scale in Fig. 4). Uncertainty ranges are on average of one order of magnitude. Long-term emissions generally feature larger uncertainties than other emission types and presumably play an important role in the overall uncertainty for that process. But why are long-term emissions so uncertain? An *ad hoc* guess might be that the uncertainties in landfill development, i.e. uncertain transfer coefficients are to be blamed. Analysis shows that the uncertainty in transfer coefficients, is not the sole contribution to the cumulative uncertainty. As described by Eq. (1), an emission is a product of transfer coefficient and waste composition. The variation in the *composition* of the landfilled waste introduces additional uncertainties, especially for trace metals. Trace metals have low concentrations in waste and low concentrations naturally have a larger variability associated with them, which was heeded using a generic formula based on measurements (Doka 2003). The uncertainty in waste composition is more influential than the uncertainty of transfer coefficients for many important long-term emissions. The origin of uncertainty for the long-term emissions of Fig. 4 are shown in **Fig. 5**. For the long-term emissions of copper, nickel and arsenic the overall uncertainty is controlled by waste composition uncertainty. These three emissions alone generate over 50% of the total impact. The overall uncertainty for the incineration of MSW is likely to be influenced strongly by waste composition uncertainty.

4.4 Long-term emissions

In the 22nd LCA discussion forum at ETH Zurich the evaluation of long-term impacts was discussed amongst 30 LCA practitioners[11]. According to the results of the various discussion groups it was agreed that a) long-term impacts should be considered in LCA, and b) long-term emissions should be *inventoried separately* from short-term emissions. There was no consensus on whether short-term and long-term impacts should be weighted equally. Some prefer to weigh short-term emissions higher because it is closer to them. Consistent and approved forecasts should be used when considering future changing environments in LCI and LCIA. The majority of the forum attendants, however, felt no need to strive for a temporal resolution in generic LCI data that is more detailed than the differentiation made in ecoinvent and described in this paper. Time and money should rather be invested for a better geographic differentiation[12]. The issues of high uncertainty of long-term emissions and long-term impact assessment have been identified as areas for future research.

One might call LCI data of landfilling processes 'highly uncertain'. Uncertain data is however a common problem of LCA. In the ecoinvent database LCI uncertainties are *quantified* and it is thus possible to roughly illustrate the relative uncertainty of landfilling processes compared to other inventoried processes. Monte-Carlo analysis yields a mean, an upper and a lower boundary value for inventory figures (2.5 and 97.5 percentiles). For the examination at hand, these inventory figures are multiplied with the LCIA characterisation factors of two fully aggregating methods (Eco-indicator'99 (H,A) and Ecological Scarcity 1997), resulting in three LCIA score values for each dataset in the database[13]. Finally, the *relative dispersion* – the difference between the upper and the lower boundary value, divided by the mean value – as a compact expression for the overall uncertainty is calculated. This procedure yields indicative results for the typical uncertainty of those emissions, *which are relevant in the LCIA result*. The results show for all landfilling processes relative dispersions in the range of 0.9–4.8, while for non-disposal datasets relative dispersions in the range of 0.28–5.0 are obtained. Due to the interconnected nature of the database, the relative dispersions of other datasets are also influenced by landfilling processes. However, the contributions of long-term landfilling impacts to the LCIA results of other processes in the database is usually below 20% of the total score (Doka 2004). For this reason the contribution of landfilling processes to the relative dispersions for non-disposal processes is small.

Apparently, there is a *large overlap area* between 0.9 and 4.8, where landfilling processes have the same relative dispersions as those of other datasets of the database ecoinvent. About 70% of all production and supply datasets in this database have values which are well within the overlapping range. Thus, although landfilling processes have sometimes elevated uncertainties, they are – on the level of LCIA results – not seriously (i.e. not orders of magnitude) more uncertain than other processes within the database ecoinvent, These results, alas, do not heed the uncertainty in the characterisation fac-

[11] On May 7th 2004. See the web site http://www.texma.org/LCA-Forum/lca-forum.html for further information and documentation.

[12] This agrees well with the priorities expressed in the findings of a survey regarding the needs and requirements for LCIA, conducted by the UNEP/SETAC Life Cycle Initiative (Stewart & Goedkoop 2003). There the item 'Temporal differentiation is a key requirement of any LCIA model' got the lowest score, although one third of the respondents still gave scores between 3 and 4 (4 meaning 'complete agreement' and 0 'complete disagreement'). The UNEP/SETAC survey comprises 91 respondents from 28 countries.

[13] In ecoinvent 2000 Monte-Carlo analysis is currently not performed on LCIA results. Uncertainties for LCIA factors are not heeded here. The outlined method to calculate LCIA score uncertainties is not mathematically strict. The resulting values should however suffice for a relative comparison of process score uncertainties.

tors for *future* emissions, which are expected to be larger than the uncertainty in the characterisation factors for present emissions. The uncertainty of *characterisation factors* is currently not heeded in ecoinvent LCIA results.

5 Conclusions and Outlook

5.1 Conclusions

Waste disposal processes in LCA should heed the specific composition and characteristics of the waste material. Generic assessments 'per kilogram waste' for whole waste classes are too coarse. The models presented here allow the consistent modelling of waste disposal process chains. The resulting burdens from disposal have a certain, usually non-negligible relevance compared to material production. The disposal phase should therefore by default be included in LCA studies. The created inventories and user tools facilitate heeding the disposal phase with a similar level of detail as production processes. The risk of LCA-based decisions shifting burdens from the production or use phase to the disposal phase because of data gaps can therefore be diminished. A contribution analysis for long-term emissions from landfills is advisable.

5.2 Outlook

The presented disposal processes are based on average Swiss conditions. In the future, disposal processes for other regions could be inventoried, which can – in the case of landfills – be very dependent on climate. A type of disposal not heeded in the ecoinvent database is the landfilling of tailings. Tailings are mineral wastes from the refining of metal ores. Large masses of tailings are produced and they contain high levels of heavy metals. A tailings model for different world climate regions should be developed. Landfill models describe the behaviour of a landfill body, which is in some respects similar to the processes in soil. The cross-examination of landfill models to LCIA soil fate models could be advantageous. Currently only chemical elements, like copper, zinc, nitrogen etc. are heeded by the disposal models. A possible extension could be the modelling of the behaviour of chemical compounds, like dioxins or other hydrocarbons.

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