Life Cycle Based Cost-Benefit Assessment of Waste Management Options

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Presentation for ISWA Annual Congress 2006, 2006.10.1-5., Copenhagen

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EXECUTIVE SUMMARY

In a study for the EU commission (DG-JRC), a new method for monetarisation of life cycle impacts has been developed and applied to different waste treatment scenarios, using the most recent generic data sources for the different technologies.

The objectives of the study were:

- To provide life cycle assessment data, results and strategy recommendations for municipal waste management in the new EU member states, with Malta and Krakow as specific cases.
- To provide background insights for the development of a common European reference system for life cycle assessment (LCA).

The study applies a mainstream “bottom-up” life cycle assessment method, supplemented by data for background processes from “top-down” input-output matrices, based on national accounting statistics combined with national emission statistics.

In terms of impact assessment, the study applies both a mainstream “midpoint category method” with 14 midpoint impact categories, and an “endpoint” or “damage category method” where the midpoint category results are further modelled to a single damage category and expressed in monetary units (Euro), thus allowing full monetarisation of all environmental externalities.

Separate comparisons are made of the different treatment options for each individual waste fraction. These comparisons show that both from an economic and an environmental view-point, the preferable technologies are:

- Maximum source-separation (80-85% recovery rates from kerb-side collection),
- Composting of wet biodegradable waste with biogasification of the forced leachate,
- Maximum materials recycling,
- Incineration of the residual waste (with energy recovery).

The following strategic recommendations for waste management are made:

- Initiatives are required to overcome any structural, financial, technical and psychological barriers for increased recycling of separately collected waste fractions.
- Government intervention is necessary to ensure recycling of waste fractions with a high heating value, since on a pure economic cost basis incineration appears preferable for these fractions, while recycling is preferable when the environmental externalities are taken into account.
- Long-term forecasts should be made of the future waste amounts under increasing rates of recycling and composting, to avoid overinvestment in incineration capacity and the consequent technological lock-in.
- Government intervention could most efficiently be made at the EU level, due to the low importance of geographical conditions, and the disperse nature of waste management impacts/benefits.

For a common European reference system for LCA, we identified the following issues to be addressed:

- In the mainstream “bottom-up” LCA method, the experience based cut-off rules lead to data gaps of a size that causes concern. For waste collection and the upstream inputs to waste incineration we found previously undetected data gaps of 76% and 62% of the total environmental impacts, respectively, by comparing to the more complete input-output data.
- In attempting to complete the data from the mainstream method with data from input-output based NAMEA matrices, we encountered problems in obtaining data at an adequately disaggregated level for material recycling.
- For situations of co-production, the preferred ISO procedure (system expansion) is not yet consistently applied in standard, commercial LCA databases. This may lead to inconsistencies and errors when using such databases to provide background data.

INTRODUCTION

The following waste material fractions are analysed separately:

- Wet biodegradable wastes,
- Paper and cardboard wastes, subdivided in Cardboard wastes, Newsprint wastes and Other paper wastes,
- Plastics wastes, subdivided in Polyethylene wastes, and Other plastics wastes,
- Glass wastes,
- Iron and steel wastes,
- Aluminium wastes,
- Other wastes (Residual).

The analysed product systems include collection from the household and all subsequent unit processes. The studied waste treatment technologies are:

- uncontrolled landfill
- directive compliant landfill,
- directive compliant incineration with energy recovery, 100% SNCR, and semidry flue gas scrubbing
- home incineration,
- central composting with energy recovery,
- central composting without energy recovery,
- home composting,
- material recycling.

The study intends to model modern, best available technology (BAT), but also the current waste management infrastructure was modelled for comparison. More precisely, modern technology is generally defined as Directive compliant, and taking into account the information provided in the BREF notes.

The study was intended to be specific to the two study areas, i.e. Malta and Krakow. However, since only a limited amount of specific data was available for the two areas, the study relies mostly on generic data. This implies that the data and results are also applicable to other European cities with similar population densities.

LIFE CYCLE INVENTORY

The analysed waste composition was that of the two study areas, i.e. Malta and Krakow. The substance composition of each fraction was taken from the Ecoinvent Tool (Doka 2003) and verified against the AWAST study results (Fehringer et al. 2004) and supplemented with data from here when important differences were found.

It is assumed that all households will have kerb-side collection of both residual wastes and source-separated waste fractions, since it is unlikely that neither directive compliant nor economically optimal recycling rates can be achieved by bring collection alone (Tucker & Speirs 2002). This implies that bring systems are not be seen as an alternative to kerb-side collection, but as a complementary element of a multi-faceted collection system that optimises recycling through offering the households different suitable ways to dispose their wastes. We have therefore used a fixed design for bring collection, based on 1 collection point per 1,000 inhabitants. It is assumed that under these conditions, bring collection does not involve more transport work than kerb-side collection, i.e. private transport is not increased, since drop-off is done on the way to other errands, and capacity utilization in waste collection is equal for kerb-side and bring systems.

Data for collection systems were mainly taken from the AWAST study (Kranert et al. 2004), including data on collection costs. The process and emission data are taken from the Ecoinvent database (www.ecoinvent.ch).

For wet biodegradable wastes, we have used a range of 0 – 30 EUR/Mg for the net additional cost of separate collection, based on the assertion by Ricci (2003) that an optimised collection system for wet biodegradable wastes will result in so low amounts of putrescent materials in the residual wastes that collection frequencies for these wastes may be reduced, even to the extent that the increased collection costs for wet biodegradable wastes are completely offset by cost savings in the collection of residual wastes. Data for the precise size of these costs savings are still limited, but we find it safe to say that the net additional costs for separate collection of wet biodegradable wastes will not exceed the additional cost of separate collection of other materials, i.e. 30 EUR/Mg.

For the different waste treatment options, we applied transfer coefficients (linking the substance composition of each waste fraction to emissions in the different output compartments) from Ecoinvent (Doka 2003), after a thorough validation. For other emissions, as well as upstream processes, i.e. inputs to the waste treatment and recycling processes, data are also taken from the Ecoinvent database, with a few modifications based on AWAST data (for composting see below). For dioxin, 50% of the Directive emission limit value are applied. The electricity generation is calculated from the lower heating value of the wastes, using a gross efficiency of 25%, based on AWAST (Bozec 2004). Because of the importance of emissions from electricity production, all supplies of electricity in the Ecoinvent data, except supplies for primary aluminium production, have been changed to the long-term marginal electricity for central Europe (Weidema 2003), which has been modelled with data for German average coal fired technology as proxy for modern coal fired power. The costs of the waste treatment processes are taken from AWAST (Bozec 2004) and the value of recycled materials from www.letsrecycle.com, verified against other sources.

The best available technology for composting is regarded as the one that results in the largest energy utilization, since this can replace other more polluting energy sources. We have therefore based our technology description on a composting process where the acid hydrolysis takes place in a closed reactor with collection of the forced leachate, which is transferred to an anaerobic digestion phase for biogas production. The biogas is used for electricity (and heat) production, while the hydrolysed waste is composted, first in the reactor under ventilation with a biofilter on the outgoing air and later in open windrow composting. The process is described in Kjellberg et al. (2005). The composting plant is modelled with Ecoinvent data (Nemecek et al. 2004) using the same transfer coefficients as for landfill, assuming an 80% degradation of the wet biodegradable wastes, except for some more recent emission data from Ødegård et al. (2005) and DEFRA (2000). CO_2 emissions are calculated as the residual carbon from the carbon balance, i.e. the amount of carbon in the waste (100%) minus the carbon emitted as methane or carbon monoxide and minus the 20% carbon in the final compost.

In addition to the above-described composting technology, also central composting without energy recovery has been modelled. Here, the data from Nemecek et al. (2004) is used for methane emissions (3.5 kg per Mg wet biodegradable waste), the rest of the carbon being emitted as CO_2 (except what remains in the compost). Home-composting is modelled as an intermediate between aerobic and anaerobic digestion.

IMPACT ASSESSMENT METHOD

We have selected a combination of characterisation models from two of the most recent life cycle impact assessment methods, the IMPACT2002+ v. 2.1 and the EDIP2003 methods. A criterion for selecting the IMPACT 2002+ or EDIP 2003 characterisation models has been their ability to provide specific site-dependent characterisation factors for emissions from processes that are geographically specified in the inventories (i.e. processes identified as located in Malta and Poland, respectively). The main criteria for choosing a *specific* characterisation model has been completeness in coverage, both in terms of how much of the impact chain is covered by the model, and in terms of substances included.

Fourteen midpoint categories are included (in alphabetical order):
- Acidification
- Ecotoxicity
- Eutrophication
- Global warming
- Human toxicity
- Injuries
- Ionizing radiation
- Mineral extraction
- Nature occupation
- Non-renewable energy
- Ozone layer depletion
- Photochemical ozone impacts on vegetation
- Respiratory inorganics
- Respiratory organics (photochemical ozone impacts on humans)

Three damage categories are defined:

- Ecosystem impacts
- Human well-being
- Economic production.

The starting points for the damage assessment are the category indicator results from the midpoint impact assessment. In principle, the starting point could also be the inventory result, thus circumventing the midpoints. However, it is seen as an advantage that consistent results can be obtained at both midpoint and damage level by combining the two categorisation models.

The three damage categories are later aggregated, first by expressing ecosystem impacts in terms of human well-being, and then by introducing a conversion factor between human well-being and monetary measurement units, thus allowing aggregation of all three damage indicators in a single impact category “human production and consumption efficiency”, measured in the monetary unit EUR_{2003}. Thus, the endpoint impact assessment method is a way to determine the economic externalities of the waste treatment options. In general, future impacts are not discounted.

Ecosystem impacts are expressed in terms of human well-being by applying a temporary proxy value derived from the protection target expressed in the Convention on Biological Diversity. This proxy value is 21 biodiversity-adjusted hectare-years (BAhy) per quality-adjusted human life year (QALY), and is verified by comparing to current environmental protection expenditures.

Having aggregated all physical impacts (externalities) into one unit (QALY), allows us to determine an equivalence between gross economic production and QALYs. The monetary value of a QALY has an upper limit defined by the budget constraint, i.e. the fact that the *average* annual income is the maximum that an *average* person can pay for an additional life year. Since a QALY by definition is a life-year lived at full well-being, the budget constraint can be determined as the *potential* average annual income at full well-being, which is equal to the potential annual economic production per capita. We determine the potential annual economic production per capita to be 74000 EUR_{2003} with an uncertainty estimate of 62000 to 84000 EUR_{2003}. This monetary value of a QALY corresponds well to the 74,627 EUR willingness-to-pay estimate of the ExternE project. Differences to other estimates can be explained by inherent biases in their valuation approaches.

IMPACT ASSESSMENT RESULTS

Comparing treatment options for the wet biodegradable fractions

Figure 9.6.1 shows the environmental impacts for different treatment options for the wet biodegradable fractions. This shows an advantage of composting *with energy recovery* over the other options. The main reason for this is the better energy utilisation, and thus the lower net emission of greenhouse gases. However, due to uncertainty on the energy conversion efficiencies, the difference to incineration is only significant at the 90% confidence level, i.e. there is a 5% chance that incineration has lower environmental impacts.

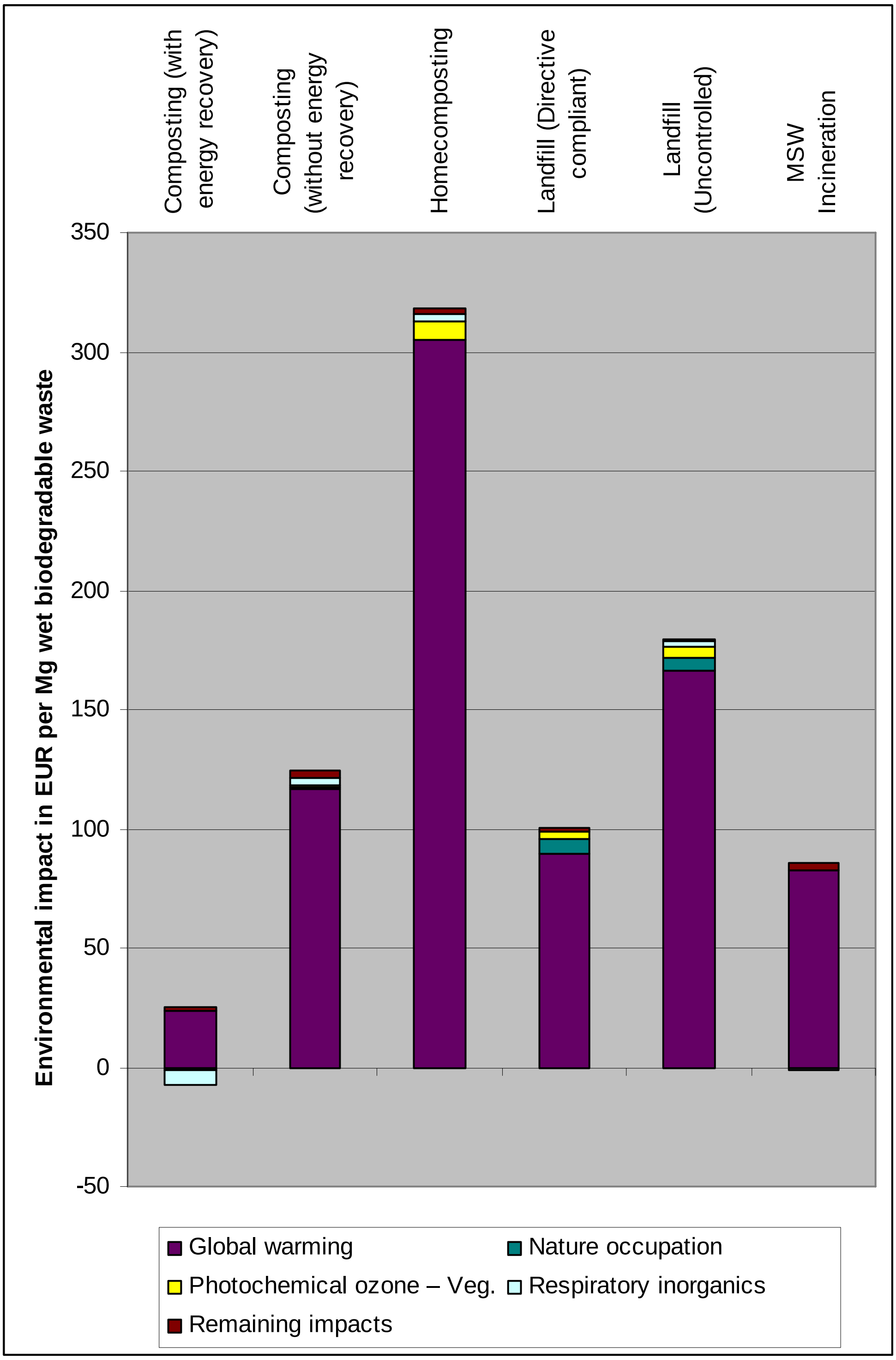

Figure 9.6.1. Environmental impacts for different treatment options for the wet biodegradable fractions of municipal solid waste. Waste collection not included. To obtain the total value for a column, the negative values should be subtracted from the positive.

The homecomposting option turns out to be problematic, due to the assumption of partly anaerobic digestion, i.e. that some homecomposting takes place with insufficient aeration, and thus emits methane, a potent greenhouse gas. Combined with a relatively high degree of decomposition in composting (80%) compared to landfill (27% decomposition), the resulting global warming dominates the result. In the directive compliant landfill, a larger amount of the methane is captured and combusted.

The result for incineration is also dominated by the global warming impact resulting from the combustion of the wet biodegradable wastes, a more complete breakdown and thus a larger release of CO_2 than from composting, which is only partly offset by the recovered energy. Nevertheless, incineration is the best option after composting with energy recovery, although the difference to directive compliant landfilling is not statistically significant.

When including direct economic costs in the calculations, composting with energy recovery still comes out the best solution, in spite of the higher costs of collection and treatment. Graphs that illustrate this, as well as similar graphs for other waste fractions can be found in the project report (Wedema et al. 2006). For all waste fractions, where recycling is an option, this is the preferable option. For “Other wastes” and the residual that is not separately collected, incineration is the option with the lowest environmental impacts.

Comparison of the Krakow and Malta results

When analysing waste fraction per waste fraction, the results from Krakow and Malta do not differ significantly, in spite of the site-dependent impact assessment applied. The results are thus independent of geographical conditions. However, differences in waste composition may still lead to large differences, when analysing the entire municipal waste. For example, due to the much larger fraction of wet biodegradable wastes in Malta, Malta waste treatment scenarios generally have much more contribution to global warming and less displaced emissions than the same scenarios in Krakow.

COMPARISON OF BOTTOM-UP AND TOP-DOWN DATA

The data sources and results mentioned above are for traditional “bottom-up” LCA data. In addition, we have applied input-output (IO) data from Denmark (Weidema et al. 2005) for comparison, revealing data gaps of 76% and 62% of the total environmental impacts for waste collection and incineration, respectively.

For example, the traditional “bottom-up” data for waste collection only cover the maintenance and diesel consumption of the collection vehicle, while the IO-data cover all inputs to the waste collection service, such as electricity, machinery, chemicals, air and ship transport, radio and communication equipment, and wood products. Similar under-reporting in the traditional “bottom-up” LCA data is found for waste incineration.

Unfortunately, it was not possible to obtain adequate IO-data for material recycling, and to avoid bias, we applied the same data sources (i.e. traditional “bottom-up” LCA data) for all processes in our further assessment.

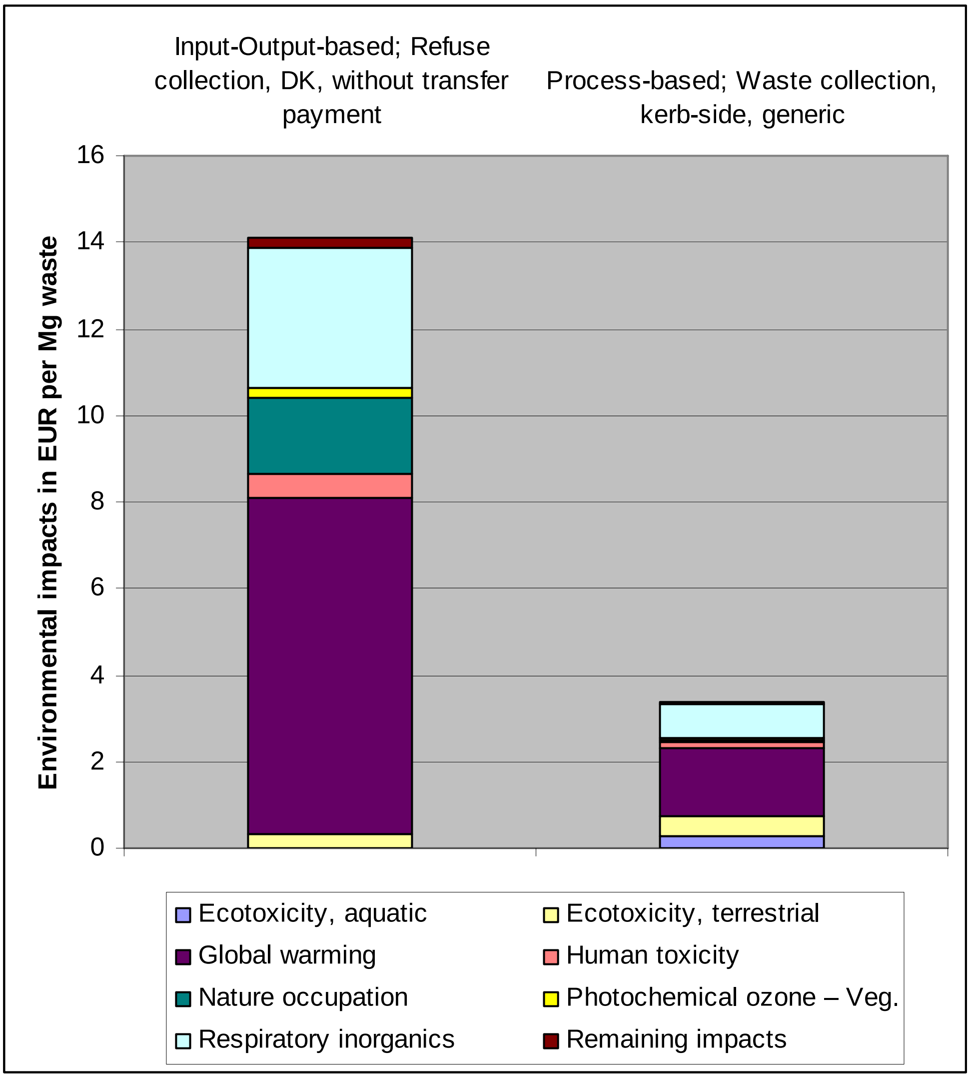

Figure 9.10.1. Comparison of endpoint results for waste collection, using input-output based data versus process-based data.

CONCLUSION

There appears to be large both economic and environmental advantages in a strategy that completely avoids landfilling of municipal wastes. For all separately collected waste fractions, recycling (including composting with energy recovery) is the waste treatment option with the lowest environmental impact, and for the remaining wastes ("Other wastes" and the residual that is not separately collected) incineration is the option with the lowest environmental impacts.

From a purely economic cost perspective, incineration provides more income than recycling for waste fractions with a very high heating value, such as PE and paper, depending on the costs of separate collection. However, when external costs are included (i.e. if environmental costs are internalised), recycling comes out with the lowest societal costs, even for these waste fractions.

The results are adequately clear to support general waste management decisions, and are not influenced significantly by local conditions or the impact assessment method applied (midpoint or endpoint).

The scope of the study is limited to alternative waste treatment options (landfilling, incineration, composting and material recycling). It is possible that in many cases, the *prevention of waste* *generation* is a more cost-efficient and environmentally sound management strategy than waste treatment.

Similarity and differences to previous studies

In general, our conclusions concur with those of previous studies, such as Villanueva et al. (2004), RDC-Environment & Pira International (2003), and Smith et al. (2001), Hogg et al. (s.a.), but are even more unambiguously in favour of recycling and composting with energy recovery. This is mainly due to differences in data and assumptions used.

We apply newer environmental and costing data, representative of the best available technology. Especially for the composting option, this is important for the results.

We assume low-cost, optimised collection systems, which can reach high collection rates by combining high levels of promotion with both kerb-side and bring collection options. Low costs of collection and high capture rates are important parameters for the economic advantage of the recycling option.

ACKNOWLEDGEMENTS

We would like to thank the teams of Prof. Olivier Jolliet and Michael Hauschild for their contributions to developing the impact assessment method for this project.

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