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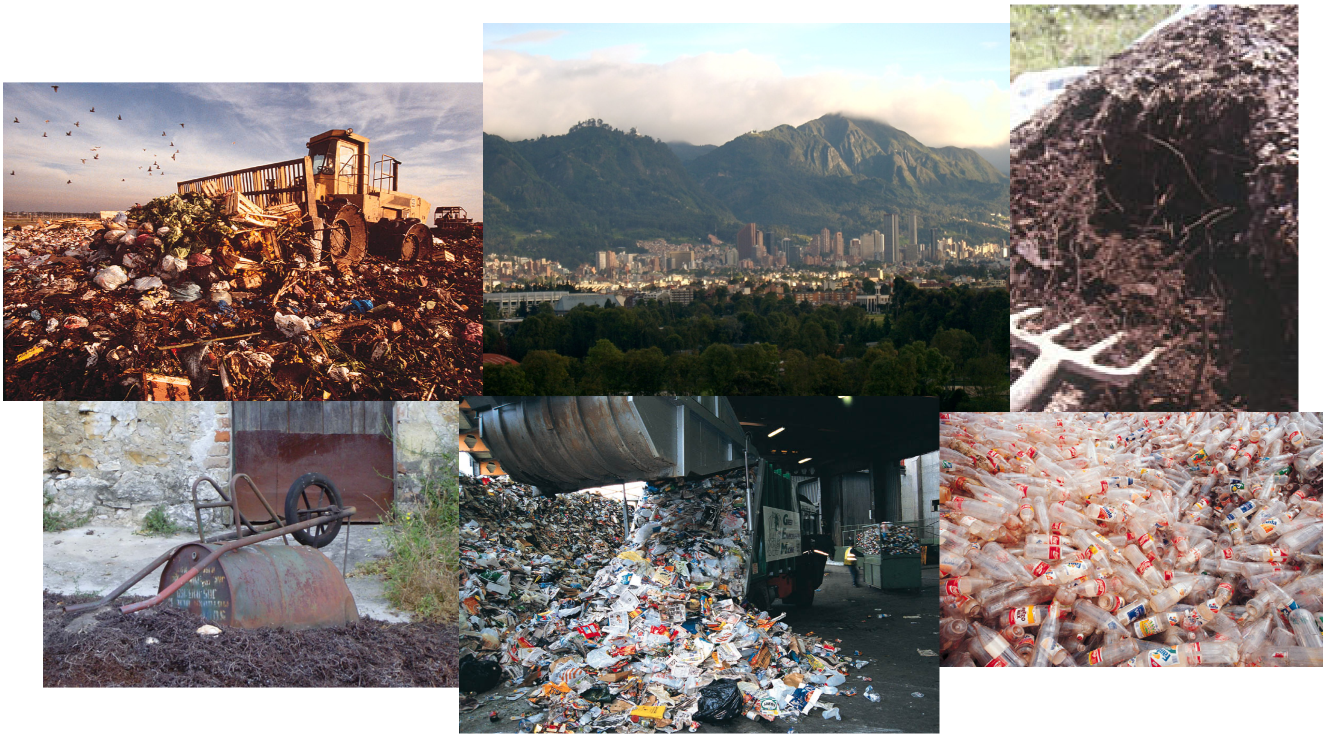
Environmental Assessment of Municipal Waste Management Scenarios: Part II – Detailed Life Cycle Assessments



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Environmental Assessment of Municipal Waste Management Scenarios:

Part II – Detailed Life Cycle Assessments

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European Commission Joint Research Centre

Institute for Environment and Sustainability

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List of acronyms

BAT – Best Available Technology

BREF – BAT reference document

CF – Characterisation Factor

CFC – Chlorofluorocarbons

DALY – Disability Adjusted Life Years

DK – Denmark

EC – European Commission

EPA – Environmental Protection Agency

EU – European Union

FGC – Flue Gas Condensation

GDP – Gross Domestic Product

GEP – Gross Economic Product

GHG – Greenhouse Gases

GHP – Gross Household Production

HDF – Human Damage Factor

HDPE – High Density Polyethylene

IPCC – Intergovernmental Panel on Climate Change

ISO – International Standard Organisation

JRC, IES - Joint Research Centre, Institute for Environment and Sustainability

LCA – Life Cycle Assessment

LCIA – Life Cycle Impact Assessment

Mg – Mega-gram or metric tonne

MSW – Municipal Solid Waste

NAMEA – National Accounting Matrix Including Environmental Accounts

PAF – Potentially Affected Fraction of species

PAH – Polycyclic Aromatic Hydrocarbons

PDF – Potentially Disappeared Fraction of species

PE – Polyethylene

PET – Polyethyleneterephthalate

PM – Particulate Matters

PP – Polypropylene

ppb – parts per billion

ppm – parts per million

PS – Polystyrene

PU – Polyurethane

PVC – Polyvinylchloride

QALY – Quality Adjusted Life Years

SCR – Selective Catalytic Reduction

SNCR – Selective Non-Catalytic Reduction

TEG – TriEthyleneGlicol

TPM – Total Particulate Matter

UES – Area of Unprotected Ecosystem

UK – United Kingdom

US – United States

VOC – Volatile Organic Compound

YLD – Years of Life lived with Disability

YLL – Years of Life Lost

1 Foreword

The European Commission's Strategy on the Prevention and Recycling of Waste outlines why life cycle thinking is essential in the move towards more sustainable consumption and production. The importance of life cycle thinking is further highlighted in the Commission's complimentary Strategy on the Sustainable Use of Natural Resources, in its Integrated Product Policy, as well as in the proposed revisions to the European Waste Framework Directive and the up-coming Sustainable Consumption and Production Action Plan.

All the stages associated with a product's life cycle, from the extraction of raw materials, manufacture, use, recycling operations, as well as the ultimate disposal of waste contribute to pressures on the environment and the consumption of resources. Differences amongst product options can occur at different stages in each life cycle, as well as between different impact categories. Over their life cycles, products, both goods and services, contribute to climate change, stratospheric ozone depletion, photooxidant formation (smog), eutrophication, acidification, carcinogenic effects, the depletion of resources including land use, and noise, among others. To consider the full life cycle of products, hence quantify the impacts, support which product option is preferable, and identify where improvements might be made, requires life cycle thinking.

Life cycle assessment (LCA) is a widely used and internationally standardized (ISO14040 ff)[1] methodology that helps to quantitatively support life cycle thinking. LCA compliments many regulatory- and more site- or process-oriented risk and impact assessments. In the context of waste management, the focus of this report, questions include whether it is better to e.g. incinerate plastics, paper, and biodegradable wastes to generate heat and electricity, or whether it is preferable to e.g. recycle and compost. Answering these and similar questions requires consideration of the emissions and resources consumed that are associated with, for example, the upstream activities of providing virgin materials versus recycling them, or the burdens attributable to different fuels that may be replaced by energy generated from waste.

In 2004, following its international workshop and conference on life cycle assessment and waste management[2], the Institute for Environment and Sustainability (IES) of the European Commission's Joint Research Centre (JRC) launched a series of regional pilot case studies[3] in collaboration with

[1] ISO 14040:2006 "Environmental Management – Life Cycle Assessment – Principles and framework" and ISO 14044:2006 "Environmental management – Life Cycle Assessment – Requirements and guidelines".

[2] http://viso.jrc.it/iwmlca

[3] Koneczny K., Dragusanu V., Bersani R., Pennington D.W. Environmental Assessment of Municipal Waste Management Scenarios: Part I – Data collection and preliminary environmental assessments for life cycle thinking pilot studies, European Commission, JRC-IES, 2007.

representatives of the European Union's new member states, acceding countries, and associated countries. The representatives selected, and provided, statistical data for nine waste management regions[4]. The life cycle assessments took into account the situation around 2003 in each region and example management scenarios that achieve Directive compliance and beyond. The assessments focused on some of the key emissions, wastes generated, and resources consumed. These initial assessments helped demonstrate the many trade-offs, and benefits, that are associated with different waste management options.

This report, based on a study carried out on behalf of the JRC by 2.-0 LCA Consultants[5], considers in further detail the waste management options for the island nation of Malta and the central European city of Krakow, Poland. The life cycle assessments use more robust data relative to the first demonstration studies, consider the potential for use of cutting-edge methodologies, and take into account waste management costs.

The resultant life cycle impact indicators provide a basis to compare the emissions and resources consumed attributable to each waste management option in terms of their contributions to e.g. different environment and human health burdens. One of the methods furthermore highlights how some of the trade-offs between environment, health, and the waste management costs might be partially considered in a single life cycle based cost-benefit framework, as a support to other decision-making information. At the same time, work is still ongoing in the European Platform on Life Cycle Assessment to provide a European Reference Life Cycle Data System (ELCD) and supporting Technical Guidance Documents[6]. The approaches and data presented in this report are therefore of an exploratory/demonstration nature and were conducted from a research perspective.

Life cycle thinking and related methodologies, such as life cycle assessment, are now playing an ever-increasing role in supporting the decisions of consumers, suppliers, business, and governments. These detailed life cycle assessments for Malta and Krakow helped more comprehensively quantify some of the environmental advantages of compliance with EU Directives for municipal waste management, particularly in the context of climate change. The assessments equally quantified some of the likely benefits, and trade-offs, at different scales of public administration; local, national, European, and global. Further reductions in waste management costs, at the same time reducing environmental burdens, can be achieved by going beyond just compliance.

Dr. David W. Pennington

4 Balancing Waste Management with Economic Growth, Press Release, European Commission, Joint Research Centre, 29 June, 2006 (see also http://viso.ei.jrc.it/lca-waste)
5 *European Commission study, contract no. 380827 F1ED*
6 http://lca.jrc.ec.europa.eu

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2 Summary

This report presents the research study results of life cycle assessments for municipal waste management, focusing on two of the JRC's pilot study[7] areas, Malta and Krakow (Poland). These detailed research studies use more robust assessment data relative to the previous demonstration studies, consider the potential to use various approaches for conducting LCAs, apply cutting-edge life cycle impact assessment methodologies, and take into account the costs of different waste management options.

2.1 Strategic recommendations for waste management

The results of this study are considered adequately clear to support the following recommendations for waste management:

- Initiatives are required to overcome any financial, technical and psychological barriers for increased recycling of separately collected waste fractions.
- Government intervention may be necessary to ensure recycling also of some of the waste fractions with a high heating value, since on a purely economic basis incineration appears to be preferable for these fractions, while recycling is preferable when the environmental burdens are taken into account.
- Long-term forecasts should be made of the future waste amounts and types under increasing rates of recycling and composting, to avoid over-investment in capacity and consequent technological lock-in.
- Government waste management guidance might most efficiently be made at the EU level, due to what appears to be the low importance of geographic variations and the disperse nature of impacts/benefits of the regional/global scale when considering a life cycle perspective. However, this will not replace the additional need to consider variations from a local impact perspective in relation to choosing the location of facilities as well as other local variations in the life cycle studies such as the local need for heat produced or compost, meeting national legal requirements, etc.

2.2 Study area and scope

The main results of these life cycle studies are presented relative to one metric tonne of municipal solid waste from private households, including waste from commercial operations when this is collected together using the same infrastructure

[7] Further details can be found in: Koneczny K., Dragusanu V., Bersani R., Pennington D.W. Environmental Assessment of Municipal Waste Management Scenarios: Part I – Data collection and preliminary environmental assessments for life cycle thinking pilot studies, European Commission, JRC-IES, 2007.

as the household waste. While various scenarios are taken into account, these are not exhaustive and many factors may influence the validity of the results.

The main differences between the composition of the waste of the two study areas is that Malta has a much higher fraction of wet biodegradable waste. Depending on the management option, biodegradable waste can be an important contributor to climate change.

Five scenarios were analysed for each study area – the baseline scenario and four alternatives accounting for the compliance requirements of the Landfill Directive 1999/31/EC and of the Packaging Directive 2004/12/EC:

(A) Baseline waste management infrastructure (circa 2003, i.e. before joining the EU)

(B) Incineration-based scenario with increased recycling (Directive compliant)

(C) Composting-based scenario with increased recycling (Directive compliant)

(D) Economic optimum scenario, in which the waste treatment options with lowest cost for each waste fraction are combined, while ensuring that EU Directive requirements are fulfilled.

(E) “Societal optimum” scenario, in which the waste treatment options are combined in a way that Directive requirements are fulfilled, while minimising the direct costs of waste management plus the costs of the environmental burdens (“externalities”).

For the composting and the incineration-based scenarios (B and C), recycling was set to the ultimate percentage requirements of the packaging waste Directive. This was applied not only to the packaging wastes but to the entire municipal solid waste stream. Targets were also set inline with the Landfill Directive’s ultimate requirements for the diversion of biodegradable waste. Except for the baseline scenario (A), the studies sought to take into account modern, best available technology (BAT).

2.3 Results

In general, differences in the impact assessment results among the scenarios are determined primarily by the recycling rates, due to displaced production or avoidance of primary materials, and secondly by the degree of energy recovery and associated benefits. Climate change is often a key consideration.

All four proposed scenarios (B to E) have significantly lower overall environmental impacts than the baseline scenario A. This shows that the move to be compliant to the Landfill and Packaging Directives and the application of BAT considerably improves the environmental performance of the waste management. Further benefits can be achieved through more intensive options.

The results for Malta, which has a higher bio-degradable waste fraction, are generally quite similar to the results for Krakow. For most impact categories there is an ordering of the scenarios from best to worst: E > D > B > C > A. The societal optimum (E) is, per definition, typically the best. Generally, the more complete management of the waste offered by the economic optimum (D) also makes this attractive. The baseline (A) was the least intensive management option, with common reliance on uncontrolled landfill. The order of preference of the incineration (B) – and the composting-based scenarios (C) was dependent on e.g. the composition and the fraction of the overall waste stream treated, hence what goes “unutilized” to landfill, and the impact category considered.

When considering only the biodegradable waste fraction, combined biodigestion and composting performs better than incineration in the context of climate change. The reason is that biodigestion and composting result in this study in a better energy utilisation of wet biodegradable wastes than incineration. The home composting option turns out to be problematic, particularly if there is anaerobic digestion. Comparing treatment options for the plastics, paper, paperboard fractions suggests lower environmental impact for recycling with incineration as the second-best option. Recycling is the best option for glass. Recycling is similarly the option with lowest net environmental impact for the iron and steel fractions, although the recycling process contributes more to climate change, eco-toxicological burdens and injuries than the displaced virgin steel production. For aluminium, recycling is by far the option with lowest net environmental impact.

There are both large potential economic and environmental advantages in a strategy that avoids landfilling of untreated municipal wastes. For Malta, the societal optimum scenario (E) has significantly smaller overall environmental impact than the other scenarios, while for Krakow the only statistically significant improvement was compared to the bio-digestion/composting-based scenario (C).

Some scenarios were not found to be statistically significantly different. When climate change was a key issue, this was mainly due to the uncertainty associated with the energy efficiency of the incineration, composting, and recycling, i.e. how much CO_2 emissions are avoided by these waste management options. The fact that the uncertainty is dominated by the CO_2-emissions holds promise for reducing the uncertainty through the use of more specific information, especially on the energy conversion efficiencies of the different waste treatment technologies.

From purely a cost perspective, depending on the costs of separate collection, incineration can provide more income than recycling for waste fractions with a very high heating value, such as polyethylene. However, when external costs are included (i.e. if environmental costs are internalised), recycling comes out with the lowest costs to society.

2.4 Comparison to previous studies

In general, the conclusions here concur with those of previous studies, but are more clearly in favour of recycling in combination with biodegradation with energy recovery plus composting. This is mainly due to differences in data and assumptions used.

The studies assumed 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 separate collection and high collection rates are important parameters for the economic advantage of the recycling option. The studies also applied more recent environmental and cost data that are representative of the best available technologies. Especially for the composting option, this is important for the results.

2.5 Overall Limitations

The scope of this study was limited to alternative waste management options (landfilling, incineration, composting and material recycling), considering wastes already generated. In many cases, in accordance with the principles of the waste hierarchy, the prevention of waste generation through more sustainable consumption and production can prove a more cost-efficient and environmentally sound management strategy than waste treatment. This study does not investigate reuse as an alternative to material recycling. The environmental merits of reuse systems are very dependent on local transport distances, and the associated costs may be often decisive.

The scenarios applied in this study, as well as the associated emissions and results, are not actual predictions of future situations, as these can be influenced by changes including in waste composition (which was kept constant in this study).

This study has been based on specifically described current best available technologies. Other - both current and future technologies - may have different performances to those described in this study. There are also likely data gaps in the emissions and resource consumption inventory and in the impact assessment. Nevertheless, these studies are based on current state-of-the-art information and practice. Preliminary approaches were adopted to highlight uncertainties associated with available data, suggesting the overall conclusions and main findings are likely to remain robust.

2.6 Inventory methodology

A life cycle inventory consists of the direct and indirect emissions, and the resources consumed, in a product's life cycle, ´from cradle to grave' – including related raw material extractions, energy acquisition, materials production, manufacturing, use, recycling, ultimate disposal, etc. The inventory is based on a model of the mass and energy balances for the different waste management options. This accounts, for example, for the life cycle emissions associated with virgin materials that are displaced by recycled materials.

Two inventory methodology variants were applied here:

1) A "conventional" ISO-based life cycle assessment method; i.e. a model based on linking of individual unit processes, using process specific data for the whole life cycle with expertise and cut-off rules to determine which processes to include.

2) A hybrid life cycle assessment method; i.e. a model of the linked processes of method 1), but for the foreground system only (i.e. the waste management technologies), with the addition of background processes (i.e. of consumed diesel fuel, electricity and others as well as substituted primary materials) from environmentally extended economic input-output matrices. This is based on national accounting statistics combined with national emission statistics (known overall as NAMEA matrices). This hybrid approach has the aim of providing a more complete model.

Due to limitations in the available data, however, the hybrid method could not be applied consistently throughout the analysed systems and the results were therefore limited. Furthermore, the underlying limitations of the methodology and data, hence the relative robustness of the results, still require critical review. This was not within the scope of this research study.

Some of the theoretical and practical advantages and disadvantages of the two modelling approaches are discussed by e.g. Lenzen (2001).

2.7 Impact assessment methodology

After compilation, tabulation, and preliminary analysis of the life cycle inventory, it is necessary to calculate, as well as to interpret, indicators of the pressures/impacts that are associated with emissions to the natural environment and the consumption of resources. For this purpose, life cycle impact assessment provides indicators for the interpretation of the inventory data in terms of their contribution to different impact categories or environmental burdens. The indicator results facilitate the evaluation and comparison of the options (here for waste management) in terms of climate change, cancer effects, land use, etc.

The scope of the assessment is, with some exceptions, limited to the consideration of contributions to impacts at the regional and global scales. The overall indicator results reflect the sum of contributions to each impact category, summed over time and space. These regional and global insights compliment information from e.g. more detailed site and temporal specific assessments, which may be conducted for example in the context of legislative requirements for emissions from a specific facility.

Two impact assessment methods were applied in this study:

- A method, where the analysed systems are compared at the level of so-called midpoint impact indicators and categories, e.g. with one indicator for each environmental category such as CO_2-equivalents for contributions to climate change.
- An “endpoint” or “damage category method”, where the midpoint category results are further modelled to single damage categories, such as Quality Adjusted Life Years for human health, and then, complimentary in this study, weighted across human health, ecological effects and resource consumption to facilitate overall cross-comparison in terms of monetary units (Euro).

While not essential in decision making and controversial, the latter additional monetisation step is one method that facilitates further comparison or weighting across impact categories, such as human health and ecosystem impacts, as well as direct comparison of the external costs with e.g. the waste management costs. Nevertheless, caution is required with such methods in decision making to ensure e.g. qualitative considerations that cannot be expressed in costs are taken into account and that the uncertainties compared to the calculated direct costs of waste management are considered; for these reasons, as well as other issues such as putting explicit values on health and ecological impacts, the approach remains controversial. As with many of the approaches in this report, this study is therefore to be considered in the research context.

For the impact assessment and in the current absence of a European recommended approach (which is presently under development in the European Platform on Life Cycle Assessment – http://lca.jrc.ec.europa.eu), the study combined two of the most advanced LCA impact assessment methods, IMPACT 2002+ and EDIP 2003, and expanded on missing areas. These methods provide both midpoint and endpoint indicators.

A key criterion for choosing these models was completeness in coverage, both in terms of how much of the impact chain is covered by the model, and in terms of substances included (especially relevant for toxicity). Another criterion here, for research purposes, was the ability of the model to also provide site-dependent indicators for emissions from processes that are geographically specified in the inventories (i.e. processes identified as being located in Malta and Poland, respectively). The selection of these particular models is not an endorsement, nor are they necessarily the best available.

2.8 Methodological observations

These studies demonstrated potential advantages and disadvantages of using environmentally extended input-output matrixes as a background data to compliment ISO-based process foreground life-cycle inventory. The studies did not investigate some of the potential underlying limitations of using economic flow data to support life-cycle assessment, or vice-versa, which are topics of further investigations. The size of the problems identified suggests, however, a need for some improvements in both methods as well as the databases adopted.

In the "conventional" ISO process-based life cycle assessment method, so-called expertise and experience based cut-off rules are used to decide which processes in a life cycle are modelled and which are not. For practical reasons, these rules can result in leaving out e.g. parts of the machinery manufacture, legal services, etc., from the analysed systems depending also on the scope of the study and of the databases used.

It was not possible to complement the overall data from the mainstream method with environmentally extended input-output based data in this study as initially foreseen, as problems were encountered in the input-output approach in providing data at an adequately disaggregated level for material recycling. Material recycling poses a particular problem in input-output-tables, since the processing of primary and secondary raw materials take place in the same aggregated industries. Distinction between the important environmental differences of these processing routes was hence not possible.

For waste collection and the upstream inputs to waste incineration, the comparison of process-based LCI data with the environmentally extended input-output approach suggests that the former may result in the omission from the studies of 76% and 62% of the total environmental impacts, respectively. The extent of such omissions is not dependent on the methodology, per se, but depends on the quality and scope of the specific process data used, so applies only to the data adopted in this study. Furthermore, differences in data obtained by the two approaches are also attributable to underlying uncertainties of the methods and basic data. Further critical review is required.

The following recommendations are made:

- There is a need to critically examine the advantages and disadvantages of using environmentally extended input-output data to compliment process-based data.
- Methodological approaches for process-based LCA should be harmonised, to avoid inconsistencies if combining data from different databases and to ensure a minimum level of quality.

- Data from process-based LCI databases should be reviewed independently to ensure that the modelled system actually covers all relevant parts and that the cut-off rules were consistently applied in an appropriate way.

Combining the midpoint and the endpoint impact assessment methods in a consistent framework, as was recommended by various workshops focusing on this topic (e.g. Bare et al. 2000), facilitates the advantages of both methods, while eliminating their disadvantages. A straightforward approach was used in these studies, where constant factors are used to convert midpoint to endpoint results.

In the attempts to combine the two impact assessment methods Impact 2002+ and EDIP 2003 selected here, and expand on missing areas, some obstacles were found that require further elaboration. Recommendations and issues for further research include:

- The need for an impact characterisation model for emissions to groundwater.
- The characterisation models for e.g. metals and persistent organic chemicals in the context of toxicological effects may not adequately reflect irreversible binding and bioavailability over time in different environmental media.
- The endpoint characterisation models for ecotoxicity should be checked/calibrated to reflect the overall importance of ecotoxicity relative to other impacts on ecosystems
- There is a need to provide consistent endpoint indicators for ecotoxicological effects with those of other ecosystem impact categories.
- An endpoint characterisation model for aquatic eutrophication is missing.
- An endpoint characterisation model for tropospheric ozone impacts on vegetation is missing. This affects both the assessment of ecosystem impacts and impacts on agricultural crop production.
- A separate impact category for agricultural crop production should be created, which should include both the impact of ozone and the impacts of other ecotoxic substances on crop yields, the fertilisation effect of CO_2 and the different mineral nutrients in emissions, as well as soil losses through erosion. It could also include the non-fertiliser effect of adding compost to soil (e.g. reduced erosion, impacts on soil pathogens, improved soil workability and water retention capacity).
- A characterisation model for ecosystem impacts during relaxation after deforestation and climate impacts is missing.
- The lack of a site-dependent characterisation model for respiratory inorganics is seen as a potential shortcoming for the site-specific impact assessment.

- The available normalisation reference for Europe is from 1995. Its usefulness should be investigated and updates made, if warranted, on a continuous basis.
- The endpoint characterisation model for climate change should be updated, improved and better documented.
- A study should be performed to express the severity of ecosystem impacts in terms relative to human well-being, preferably in conjunction with a larger study to obtain consistent values for other issues including calibration to the values derived in the “Global burden of disease” study.
- As the endpoint method includes a number of additional assumptions that may be controversial, a wider scientific and stakeholder review procedure is needed to approach consensus on the procedures and values to use.

3 Scope

3.1 Geography

This research study addressed two of the JRC pilot study areas[8], i.e. Malta and Krakow. However, since the specific data available for the two areas were limited mainly to the waste compositions, the study relies mostly on generic data. This implies that the data and results may be equally applicable to other European cities and regions with similar population densities.

3.2 Technology

The study intends to model modern, best available technology (BAT), except for the 2003 baseline waste management infrastructure scenario (scenario A). More precisely, modern technology is generally defined as Directive compliant (referring to Directive 1999/31/EC on landfills, Directive 2000/76/EC on waste incineration, and Directive 2001/80/EC on the limitations of emissions of certain pollutants into the air from large combustion plants), and taking into account the information provided in the BREF notes WI and WT (JRC 2005 a & b). More specifications are provided in the inventory analysis (Chapter 5).

The waste management infrastructure scenarios are based on the ultimate political targets in the Directive 1999/31/EC on landfills and the packaging waste directive 94/62/EC, i.e. to limit landfilling of biodegradable waste to 35% of the biodegradable waste production in the reference year 1995, and to ensure recycling of minimum 55% of packaging wastes, with specific targets for glass, paper, metal and plastic packaging. However, here applied are the ultimate percentage requirements of the packaging waste directive not only to the packaging waste, but also to the rest of the waste that belongs to the corresponding waste type, e.g. the ultimate directive target of 60 % recycling for glass and metal packaging wastes is applied to all glass and paper in the municipal solid waste.

The size of the waste treatment plants is determined by the total amount of waste to be treated in the selected study areas, except for the high recycling scenario E where the amount of residual waste is so limited that joint incineration with other regions is foreseen.

For important processes that can be geographically identified (e.g. local, displaced energy production), data relevant for the local technology are applied.

[8] Koneczny K., Dragusanu V., Bersani R., Pennington D.W., , Environmental Assessment of Municipal Waste Management Scenarios: Part I – Data collection and preliminary environmental assessments for life cycle thinking pilot studies, European Commission, JRC-IES, 2007.

3.3 Time

No distinction is made in terms of the time of occurrence of emissions, as per common practice in LCA, with the exception of leachate from landfills. For leachate from landfills, emissions to groundwater before and after 100 years are separately modelled, in order to consider separately the importance of these emissions. This is in compliance with the requirements of the Strategic Environmental Assessment Directive[9], which foresees a distinction between short-term and long-term impacts.

In this study and reflecting common LCA practice, future impacts are not discounted, nor is age weighting applied to distinguish between affected age groups of the population, except for impacts on economic production (Chapter 6.2.4 and 6.2.6). A specific discussion on the importance of discounting for the results is provided in Chapter 10.

3.4 Functional unit

The main study results are presented relative to a functional unit of one Mg (mega-gram or metric tonne) of municipal solid waste at private households, including waste from commercial operations when this is collected together using the same infrastructure as the household waste. Bulky and inert fractions of the household waste, and particularly electrical and electronic equipment, are not included here, as these are assumed to be separately collected and treated, at least in future waste management systems.

The analysed product systems include collection from the household and all subsequent unit processes, but not the upstream processes generating the waste (equivalent to the reasonable assumption that the choice of waste management infrastructure does not affect the composition of the waste itself).

A possible expansion of the modelled system to include the upstream processes generating the waste, would account for credits and burdens from upstream processes inherently associated with wastes, such as from the sequestration of carbon dioxide into crops used for food. However, as these processes are generally unaffected by the waste management scenario, such an inclusion of the upstream burdens and credits of the waste would not affect the relative results when comparing different waste management processes or scenarios. For recycling processes, the modelled systems do include the upstream processes related to the avoided extraction and processing of virgin materials, since these are affected by the choice of waste management scenario, e.g. energy recovery versus materials recycling.

In order to estimate scenario-wide parameters, e.g. costs, a large part of the study describes the activities related to the entire annual municipal solid waste quantities,

[9] SEA Directive 2001/42/EC on the assessment of the effects of certain plans and programmes on the environment.

i.e. 154,000 Mg for Malta[10] and 256,000 Mg for Krakow. Other parts of the results are presented per waste material fraction, in which case 1 Mg of the particular fraction is used as basis for comparing the different treatment options for that fraction. Thus, it is only the main scenario results that are given in relation to the functional unit of one Mg of municipal solid waste.

3.5 Waste definitions and fractions in this study

According to the European Waste Catalogue (Decision 2001/118/EC, 2001), municipal waste is defined as household waste and similar commercial, industrial and institutional wastes including separately collected fractions.

In this study, data for Malta includes household wastes and kitchen wastes from hospitals and restaurants, but not other fractions of commercial/industrial waste, even when classified as municipal solid waste. For Krakow, the analysed fractions include household wastes and similar commercial and industrial wastes and wastes from parks & gardens. Separately collected bulky and inert fractions of the household wastes have not been included. The detailed waste compositions are reported in Chapter 5.1.

In this study, the following waste material fractions are also analysed separately:

- Wet biodegradable wastes,
- Paper and cardboard wastes, subdivided into Cardboard wastes, Newsprint wastes and Other paper wastes,
- Plastics wastes, subdivided into Polyethylene wastes, and Other plastics wastes,
- Glass wastes,
- Iron and steel wastes,
- Aluminium wastes,
- Other wastes (see specification in chapter 5.1.2).

3.6 Waste collection technologies

The study includes kerb-side systems and bring systems. Except for the current waste management scenario (scenario A), 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 to be seen as an alternative to kerb-side collection, but as

[10] Not including commercial waste, except for kitchen waste from hospitals and restaurants.

a complementary element of a multi-faceted collection system that optimises recycling through offering the households different suitable ways to dispose of their wastes.

Therefore a fixed design is used for bring collection for all future scenarios, 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 may typically be done on the way to other errands, and capacity utilization in waste collection is equal for kerb-side and bring systems.

3.7 Waste treatment technologies

The studied waste treatment technologies are:

- uncontrolled landfill
- directive compliant landfill,
- directive compliant incineration with energy recovery,
- home incineration,
- central composting with energy recovery
- central composting without energy recovery,
- home composting,
- material recycling.

4 Life cycle inventory analysis methodology and relation to ISO 14040

A life cycle inventory provides estimates of the emissions and the consumption of resources attributable to a product's life cycle, from 'cradle to grave' – including raw material extractions, energy acquisition, materials production, manufacturing, use, recycling, ultimate disposal, etc. The inventory models the mass and energy balances for the different options, accounting for e.g. the emissions associated with virgin materials that are displaced by recycling.

Two methodology variants are applied:

1) A "conventional" ISO-based[11] life cycle inventory method, which implies a system modelling based on the linking of individual unit processes, using mainstream, experience based, cut-off rules. Depending on which specific database (approaches and quality of data) is used, this can imply that many inputs of capital goods, services and minor inputs are only roughly modelled or completely excluded from the analysed systems.

2) A hybrid life cycle assessment method, which implies a system model completing the "bottom-up" processes of method 1) with the background processes from input-output matrices, based on national accounting statistics combined with national emission statistics (known as NAMEA matrices). This implies that all inputs of capital goods, services and minor inputs are included in the analysed systems.

Due to limitations in the available data (see Chapter 5.3.5), the hybrid method (2) could not be applied consistently throughout the analysed systems. The results are therefore limited to demonstrate the method for some selected parts of the systems (see Chapter 9.10). Other limitations may come from the unclear completeness of the elementary flows covered and other omissions, as well as methodological issues in relation to attributing environmental impacts relative to economic flows between sectors and allocating these also among products of the same sector.

For both methodology variants, ISO 14040 rules are applied, as well as – when relevant – the supplementary assumptions and procedures outlined in the Danish LCA inventory guidelines (Weidema 2003), with the following exceptions:

[11] ISO 14040:2006 "Environmental Management – Life Cycle Assessment – Principles and framework" and ISO 14044:2006 "Environmental management – Life Cycle Assessment – Requirements and guidelines".

- When co-products occur in the studied systems, such as the generation of electricity from waste incineration, priority is given to avoid allocation through the procedure of system expansion. A standard LCA database that consistently applies system expansion for all situations of co-production does not exist. Therefore applied is system expansion to all waste treatment processes, but not necessarily to all upstream processes, when these are taken from available LCA databases such as Ecoinvent that generally apply allocation by economic allocation keys. We have substituted the allocations in the Ecoinvent data with system expansion when the allocations were believed to be of importance for the overall result. All such substitutions are reported in Chapter 5 for each individual process.

- The experience based cut-offs implied in the “conventional” process-based life cycle assessment method suggest the possibility of data gaps in the Ecoinvent database (see Chapter 9.10), which made it essentially impossible within the available resources in this study to fulfil the ISO requirement that when “the study is intended to support a comparative assertion made to the public, the final sensitivity analysis of the inputs and outputs data include the mass, energy and environmental relevance criteria so that all inputs that cumulatively contribute more than a defined percentage to the total are included in the study”, unless this “defined percentage” is put at an unreasonably high level. Equally, methodological limitations or other errors in the input-output data may account for the differences.

- While intended to use data for the processes actually affected, as specified by ISO 14049, especially for all waste treatment processes, this has not been possible for all upstream processes, when these are taken from available LCA databases that generally present data as industry averages.

5 Life cycle inventories

This study does not include the collection of new primary inventory data, besides what was provided by the National environmental authorities of Malta and Krakow in the European Commission's pilot studies (JRC 2005c).

The inventory analysis is performed here by linking the collected data in a matrix and the inventory and impact assessment results are calculated by matrix inversion, as described by Heijungs & Suh (2002), with the aid of the software SimaPro.

5.1 Waste amounts and composition

5.1.1 Baseline waste composition and amounts in Malta and Krakow

Specific waste compositions for Krakow and Malta, respectively, are used as basis for the assessments, see Table 1 and Table 2.

Although the waste compositions are likely to change over time, the same basic waste compositions are assumed for all analysed scenarios, which facilitates comparisons. However, it should be noted that the results should not be used as predictions of future emissions or emissions savings, as these will be influenced by possible changes in consumption and resultant waste composition. For example, the phasing out of the existing re-use system for soft drink bottles on Malta is likely to lead to an increase in the fraction of plastics wastes.

Table 1 The baseline distribution of waste material fractions for Malta (in 2004).

	Household wastes [1]		**Kitchen Wastes from hospitals and restaurants [2]**		**Total baseline**	
	Mg/year	%	Mg/year	%	Mg/year	%
Wet biodegradable wastes	77236	58.0	20166	100	97402	63.4
Paper and cardboard wastes	19785	14.8			19785	12.9
Plastic wastes	13181	9.9			13181	8.6
Glass wastes	5179	3.9			5179	3.3
Iron and steel wastes	4580	3.4			4580	3.0
Aluminium wastes	346	0.3			346	0.2
Other wastes [3]	13168	9.9			13168	8.6
Total	**133475**	**100.0**	**20166**		**153641**	**100.0**

[1] Based on original data from Table 3. Percentage adjusted to 100%.
[2] Other fractions of commercial/industrial waste are not included
[3] See specification of “Other wastes” in Chapter 5.1.2.

Table 2 The baseline distribution of waste material fractions for Krakow (in 2003).

Waste fraction	Household wastes [2]		Commercial MSW [3]		Industrial MSW [3]		Street cleaning wastes [4]		Parks & gardens wastes		Total baseline per year	
	Mg/ year	%	Mg/ year	%	Mg/ year	%	Mg/ year	%	Mg/ year	%	Mg/ year	%
Wet biodegradable	66000	33.0	8640	30.0	1200	30.0	1140	10.0	12000	100.0	88980	34.7
Paper and cardboard	40200	20.1	12960	45.0	1800	45.0	5281	46.3			60241	23.5
Plastic	27000	13.5	3456	12.0	480	12.0	1207	10.6			32143	12.5
Glass	22200	11.1	1440	5.0	200	5.0	377	3.3			24217	9.5
Iron and steel	4400	2.2	720	2.5	100	2.5	151	1.3			5371	2.1
Aluminium	2400	1.2	461	1.6	64	1.6	151	1.3			3076	1.2
Other wastes [1]	37800	18.9	1123	3.9	156	3.9	3093	27.1			42172	16.5
Total amounts	**200000**		**28800**		**4000**		**11400**		**12000**		**256200**	**100.0**

[1] See specification of “Other wastes” in Chapter 5.1.2.
[2] Based on the total amounts and percentages provided by JRC (2005c).
[3] Based on the total amounts and percentages for commercial municipal solid waste (MSW) provided by JRC (2005c). The same percentages are applied here for both commercial and industrial MSW.
[4] In the original data (JRC 2005c) 10% of the street-cleaning waste is identified as biodegradable, the rest of unknown composition. This 90% non-biodegradable street-cleaning waste is assumed to have a composition as “Manually collected road wastes” from Fehringer (2004) (the AWAST-project Annex 5, page 56). However, in Fehringer (2004), biodegradable wastes have a share of 32%. In this study, the share for biodegradable wastes is kept at the original 10%, and hence, the other shares are increased by a factor (100-10)/(100-32) in order to obtain a total of 100%.

The composition of household wastes for Malta is based on a household waste composition survey by The Maltese National Statistics Office, carried out in the year 2002. During one week in every quarter, and every day except Sunday, the domestic wastes of 400 selected households were collected and analyzed. Table 3 presents the results of this survey, provided for each of the four three-month periods of the year and as an average.

Table 3 Results of a household waste composition survey in 2002 for Malta.

Waste material fraction	I 2002 [%]	II 2002 [%]	III 2002 [%]	IV 2002 [%]	Average 2002 [%]
Paper and paperboard	7.7	8.7	9.2	11.0	9.1
Board, cartons	6.6	5.0	5.3	5.9	5.7
Textiles	4.6	2.0	2.5	4.0	3.3
Plastic films	5.3	5.3	4.8	4.4	4.9
Plastic	5.1	5.4	4.8	4.5	5.0
Glass bottles	3.7	4.4	4.3	3.2	3.9
Ferrous materials	3.9	3.2	3.6	3.0	3.4
Aluminium cans	0.2	0.4	0.2	0.2	0.3
Food wastes (and green wastes)	59.7	58.9	55.7	57.8	58.0
Hazardous wastes	0.1	1.9	3.3	3.2	2.1
Other	3.3	4.9	6.3	3.6	4.5
Total [1]	**100.0**	**100.1**	**100.0**	**100.8**	**100.2**

[1] Data are percentages by weight. As can be seen, the totals are not always 100%. In order to obtain the resulting data in Table 1, minor adjustments were introduced.

For Krakow, data are also available on the amount of batteries and accumulators as a part of the municipal solid waste; see Table 4. (JRC, 2005c).

Table 4 Batteries and accumulators as a content of the municipal solid waste in Krakow (in 2003).

Type of waste	Reference	Index	Quantity
Batteries	758,500 capita	0.16 kg/capita	121 Mg/year
Accumulators (apart from deposit system)	333,000 vehicles	0.25 kg/vehicle	83 Mg/year
Total			**204 Mg/year**

In general, it is assumed that the amounts of the recyclable fractions (paper and cardboard wastes, plastic wastes, glass wastes, iron and steel wastes, aluminium wastes), as provided in Table 1 and Table 2, are for the clean fractions after impurities are subtracted. Soiled paper and plastics are assumed to be included in the fraction “Other”. In practice, however, wastes for recycling will always contain some impurities of the other fractions. In general, it is assumed that these impurities are later separated from the recyclable material and placed in either landfill or incineration, depending on the scenario. In practice, some impurities will need to be separated by washing, which will lead to some of the impurities ending up in waste water rather than as solid waste. These emissions to water are not considered in this study, as the influence on the result was estimated to be negligible. Likewise, the study does not include washing of separate waste components in the households prior to collection.

5.1.2 Composition of the fraction “Other wastes”

In order to estimate the composition of the fraction “Other wastes”, the data from Krakow and Malta were compared to data from literature sources. As can be seen from Table 5, the fraction “Other wastes” covers e.g. textiles, natural products (shoes, furniture), minerals (e.g. cement), laminated materials (plastic coated paper), laminated packaging (e.g. Tetrapak), combined goods (e.g. diapers, hygienic pads), electronic goods, rubber, leather and hazardous wastes. On the basis of these data, the compositions of the fractions “Other wastes” for Malta and Krakow are estimated in Table 6 and Table 7.

The US Environmental Protection Agency (2005a & b) collected data are based on the total generation of municipal solid waste in the US in 2003. Their report states that: “Sources of MSW include both residential and commercial locations. Estimated residential wastes (including wastes from apartment houses) amount to 55-65% of total MSW generation.” The references contain very detailed specifications of the content of each material fraction (e.g. plastic types in the fraction “plastic”). The data quality is assumed to be good, however the geographical areas – and thus the likely consumption patterns causing the waste generation - are very different from this study.

Petersen & Domela (2003) presented the results of an analysis of household wastes carried out for the Danish Environmental Protection Agency in 2001-2002, where household wastes from approx. 2,200 households were sorted by hand into 19 fractions. Data are provided with uncertainty information. The data quality is high, and the data are recent.

Data from Spain and Italy (Fabbricino (2001) and Vidal et al. (2001)) are assumed to be from 1998. The data quality is not known.

Table 5 Municipal Solid Waste – distribution of waste material fractions. All in % by weight.

Waste fraction	Wet bio-degradable wastes	Paper and cardboard wastes	Plastic wastes	Glass wastes	Iron and steel wastes	Aluminium wastes	Other wastes
Krakow, Household waste [1]	33.0	20.1	13.5	11.1	2.2	1.2	18.9 *Unknown 17.7* *Textiles 1.2*
Krakow Commercial Waste [2]	30.0	45.0	12.0	5.0	2.5	1.6	3.9 *Unknown 2.9* *Textiles 1.0*
Malta Household municipal solid waste, 2002 [3]	58.0	14.9 *Paper 9.2* *cartons 5.7*	9.9 *Pl films 4.9* *Plastic 5.0*	3.9	3.4	0.3 *Aluminium cans*	9.9 *Unknown 4.5* *Textiles 3.3* *Hazardous wastes 2.1*
Italy, Regione Campania [4]	33.8 *Food 29.9* *Yard 3.9*	23.2	10.9	5.7	3.3		23.2 *Textiles 4.5* *Misc. 8.7* *Ceramic wastes 1.3* *White goods 0.7* *Napkins & sanitary towels 3.3* *Leather, rubber etc. 1.8* *Wood 1.8* *Others 1.1*
Spain, City of Castellon, forecast 2002 [5]	57.1	15.2	10.1	7.1	2.7	1.1	6.7 *Brick 1.8* *Textile 3.4* *Wood 1.0* *Rubber 0.1* *Soil 0.3* *Batteries 0.1*
US data Average municipal solid waste, 2003 [6]	23.8 *Food 11.7* *Yard 12.1*	35.2	11.3	5.3	5.9	1.4	16.5 *Wood 5.8* *Rubber, Leather 2.9* *Textiles 4.5* *Other non-ferrous metals 0.7* *Other 1.8* *Miscellaneous Inorganic 1.5*

Waste fraction	Wet bio-degradable wastes	Paper and cardboard wastes	Plastic wastes	Glass wastes	Iron and steel wastes	Aluminium wastes	Other wastes
Sweden, 2004 [7]	50.4 *Food 43* *Yard 7.4*	16	11.4	2.3	3.0		13.7 *Textiles 2.3* *Nappies 5.4* *Electric & electronics 0.4* *Wood 0.6* *Hazardous 0.3* *Other combustible 4.4* *Other 0.3*
Denmark, 2003 [8]	45.8 *Food 41.4* *Yard 4.4*	10.6	9.1	2.9	3.3		28.3 *Nappies & sanitary towels 6.5* *Other soiled paper & cardboard 9.1* *Absorbent household paper 3.3* *Other combustible 5.2* *Other non-combustible 3.8* *Hazardous wastes 0.2* *Compounded products 0.2*

[1] Data on the composition of Household wastes in Krakow. JRC (2005c). See Table 2
[2] Data on the composition of Commercial wastes in Krakow. JRC (2005c). See Table 2
[3] Data on the composition of Household wastes in Malta. JRC (2005c). See Table 3
[4] Fabbricino (2001). The year of data collection is assumed to be 1998, however it is not specified. There is no distinction between ferrous metals and non-ferrous metals.
[5] Vidal et al. (2001).
[6] US EPA (2003).
[7] RVF 2005
[8] Petersen & Domela (2003).

Table 6 Assumed composition of “Other wastes” for Malta.

Type of “Other wastes”	%	Mg / year	Comment
Textiles	33.2%	4378	3.28% of the total 133,475 Mg of household waste. *(JRC 2005c)*
Batteries and accumulators	1.0%	128	Batteries and accumulators assumed to be 0.32 kg per citizen per year (as in Krakow), 400,000 citizens
Electronic goods	4.0%	534	Assumption: 0.4% of household waste (as in Sweden, 2004). 0.4/100*133475= 534 Mg

Type of “Other wastes”	%	Mg / year	Comment
Hazardous waste (other than Batteries and electronic goods)	16.5%	2168	Hazardous waste: 2.12% of 133,475 Mg household waste *(JRC 2005c)*. Batteries and electronic goods subtracted.
Paper	15.2%	2002	Paper not suitable for recycling (paper in laminated packaging, nappies, soiled kitchen paper etc.) and also wood. Assumption: 1.5% of household waste.
Plastic	15.2%	2002	Plastic not suitable for recycling (e.g. plastic in laminated packaging and nappies) and also rubber this category. Assumption: 1.5% of household waste.
Inert waste, e.g. gravel	14.9%	1956	The rest. Calculated as glass.
Total	100.0%	13168	

Table 7 Assumed composition of “Other wastes” in Krakow.

Type of “Other wastes”	%	Mg / year	Comment
Textiles	6.5%	2728	1.2% of 200,000 Mg (Household waste) + 1% of 28,800 Mg (Commercial waste) + 1% of 4,000 Mg (Industrial waste) = 2,728 Mg per year (1.06% of total MSW). *JRC 2005c.*
Batteries and accumulators	0.5%	204	Batteries and accumulators as a content of the MSW. 758,500 citizens. 0.32 kg per citizen per year *(JRC 2005c)*
Electronic goods	2.4%	1025	Assumption: 0.4% of municipal solid waste (as in Sweden, 2004) 0.4/100*256,200 Mg = 102 Mg
Hazardous wastes (other than Batteries and electronic goods)	10.0%	4202	Hazardous wastes assumed to be 2.12% of total MSW (as for Malta). Batteries and electronic goods subtracted.
Paper	27.3%	11529	Paper not suitable for recycling (paper in laminated packaging, nappies, soiled kitchen paper etc.) and also wood. Assumption: 4.5% of total MSW.
Plastic	27.3%	11529	Plastic that is not suitable for recycling such as plastic in laminated packaging and nappies. Rubber is included under this category. Assumption: 4.5% of total MSW.
Inert wastes, e.g. gravel	26.0%	10955	The rest. Calculated as glass.
Total	100.0%	42172	

The difference between the compositions of ”Other wastes” for Malta and Krakow is due the differences in how much household waste is classified as “Other wastes” in the two study areas (18.9% in Krakow and only 9.6% for Malta; see Table 5). Here it is assumed that in case of Krakow the reason is that a larger amount of plastics and paper wastes are unsorted and become content of “Other wastes”.

5.1.3 Substance composition of waste fractions

In general, the substance compositions given in the Ecoinvent database are adopted from Doka (2003), as shown in Table 8. These were verified against the compositions given in Fehringer et al. (2004) and supplemented with this data when omissions were found.

Table 8 Data sources for substance composition of waste fractions.

Waste fraction	Name of waste fraction from Doka 2003 (Ecoinvent Tool)	Adjustments relative to Doka 2003
Wet biodegradable wastes	Compostable material	
Cardboard wastes	Cardboard	
Newsprint wastes	Newspaper	
Other paper wastes	Average paper	
PE wastes	PE	
Other plastics wastes	Combination of 50% PET, 25% PP, 12.5% PS, 10% PU and 2.5% PVC	
Glass wastes	Glass	Supplemented with data from Fehringer et al. (2004) for N, P, F, Cd, and Hg. SI adjusted slightly to obtain 100%
Iron and steel wastes	Tin sheet inert	Supplemented with data from Fehringer et al. (2004) for S, N, P, F, Hg, Fe, and Al. Fe corrected slightly to obtain 100%.
Aluminium wastes	Alu in municipal solid waste	Supplemented with data from Fehringer et al. (2004) for S, N, P, F, Hg, Fe, Al. Al corrected slightly to obtain 100%. Heating value changes according to Ecoinvent report
Other wastes (Krakow)	-	See specification in Table 9
Other wastes (Malta)	-	See specification in Table 9

Table 9 Substance mass fractions of the "Other wastes" calculated from the composition in Table 6 and Table 7.

kg/kg of waste fraction (unless otherwise stated)	Other wastes (Malta)	Other wastes (Krakow)
Lower heating value (MJ/kg)	15.05	15.25
Water content	1.71E-01	1.17E-01
Oxygen (without O from H2O)	2.54E-01	2.79E-01
Hydrogen (without H from H2O)	5.26E-02	5.39E-02
Carbon	3.64E-01	3.57E-01
Sulfur	7.02E-03	4.79E-03
Nitrogen	1.31E-02	5.52E-03
Phosphor	4.63E-04	3.83E-04
Boron	3.77E-06	5.39E-06
Chlorine	2.36E-02	1.73E-02
Bromium	9.93E-06	1.78E-05
Fluorine	6.21E-04	3.85E-04
Silver	8.41E-06	5.05E-06
Arsenic	5.81E-07	1.04E-06
Barium	3.43E-04	6.00E-04
Cadmium	1.73E-04	1.02E-04
Cobalt	1.68E-05	1.56E-05
Chromium	1.60E-04	9.46E-05
Copper	5.17E-03	3.16E-03
Mercury	5.67E-06	3.04E-06
Manganese	8.12E-04	4.30E-04
Molybdenum	7.94E-07	1.26E-06
Nickel	6.36E-04	3.78E-04
Lead	1.32E-03	8.89E-04
Antimony	1.42E-05	1.59E-05
Selenium	1.14E-06	2.02E-06
Tin	6.27E-04	3.80E-04
Vanadium	4.09E-05	7.34E-05

kg/kg of waste fraction (unless otherwise stated)	**Other wastes (Malta)**	**Other wastes (Krakow)**
Zinc	2.85E-03	1.67E-03
Beryllium	2.19E-07	3.93E-07
Strontium	2.14E-05	3.85E-05
Titanium	1.68E-04	3.02E-04
Thallium	3.18E-07	5.71E-07
Silicon	6.68E-02	1.02E-01
Iron	4.20E-03	3.36E-03
Calcium	7.09E-03	1.20E-02
Aluminium	3.33E-03	5.63E-03
Potassium	1.89E-03	1.27E-03
Magnesium	1.06E-03	1.88E-03
Sodium	1.72E-02	3.01E-02
Sum wet mass (including water content)	1.00	1.00
% degradability of waste in a municipal landfill within 100 years	8.24%	8.42%

Throughout this report used is the notation Ex for 10^x, i.e. 1.71E-01 means 0.171

5.2 Collection systems

In Malta in 2003, household waste was collected at the kerb-side for all 127,500 households, with recyclable fractions being collected also in 100 sets of 4 containers placed around the islands (4,000 inhabitants per collection point). The number of collection, or bring, sites was planned to be expanded to 400 by year 2006 (1,000 inhabitants per collection point).

In Krakow in 2003, household waste was collected in containers at kerb-side for 80% of the 275,800 households, with recyclable fractions being collected also in 150 sets of 4 containers placed around the city (5,050 inhabitants per collection point).

5.2.1 Modelling of kerb-side collection

Kerb-side collection is modelled with specific data on bags/containers, fuel, labour and vehicle requirements. Table 10 reports which data were used.

Table 10 Data for kerb-side collection.

	Default data
Bag/container	One 100-120 litres HDPE container per household; weight 9-11 kg; lifetime 7-10 years; price 35-46 Euro [1]
Distance collected	30-65 km/vehicle-day [2]
Collection per vehicle per day	14-18 Mg/vehicle-day [2]. For separated fractions 7-15 Mg/vehicle-day [3]
Capacity of vehicle	8.2 Mg; 50% load factor [4]
Fuel requirement	60-72 litres/100 km [5]
Vehicle days	186-268 vehicle-days/year [6]
Vehicle life time	7-13 years [7]
Vehicle maintenance	8,000 – 17,000 Euro/year/vehicle [6]
Collection per employee per day	3.4-5.3 Mg/employee/day [2]. For wet biowaste 1.9-3.2 Mg/employee/day; for other separated fractions 2.9-4.9 Mg/employee/day [3]
Total costs	**62-100Euro/Mg, of which 55-73% is labour costs [2]. For wet biowaste 62-130 Euro/Mg, for other separated fractions 70-130 Euro/Mg [3].**
Transfer station/Long distance transport (relevant for Malta)	10-26 Euro/Mg [8] or 23-35 Euro/Mg for sea transport to the continent [9].

[1] Own assumptions.
[2] Kranert et al. (2004), p. 70-72 (main cause for variation in labour costs is the extent of service: set out set back by household or waste operator)
[3] Kranert et al. (2004), p. 74-78. For wet biodegradable waste, interaction with collection of residual waste is included in the cost estimate; see discussion in the main text.
[4] Ecoinvent data, validated against Kranert et al. (2004)
[5] Kranert et al. (2004), p.65. The resulting value for mixed waste is 0.135 kg diesel/Mgkm (0.161 l diesel/Mgkm = 66 litre diesel/100 km * 0.84 kg diesel/l diesel / 4.1 Mg average load) or 1.9 litre diesel/Mg waste (The Ecoinvent database gives two and a half times as high fuel consumption - 0.336 kg diesel/Mgkm – and consequent emissions, based on a value of 4 litre diesel/Mg waste from studies from the early 1990'ies; Sonesson 2000 (ORWARE) use a much lower fuel consumption of 25 litres per 100 km, apparently relevant for Swedish and Australian conditions).
[6] Kranert et al. (2004), p.65
[7] Own estimate. In the Ecoinvent data, an implicit lifetime of 50 years is assumed, based on 540,000 kilometres driven.
[8] Kranert et al. (2004), p.66
[9] Hogg 2001, Annex 10.4 (Greece) for sea transport.

The data in Table 10 are mainly taken from Kranert et al. (2004). In general, they compare well with the data from the European-wide survey by Hogg (2001), except that it appears from Hogg (2001) that light-weight fractions such as aluminium cans and PET bottles may have higher collection costs (200-300 Euro/Mg, i.e. up to an additional 200 Euro/Mg).

For wet biodegradable wastes, Kranert et al. (2004) report a collection cost of 87-150 Euro/Mg. However, this value probably does not include the savings in costs of collection of residual wastes that may result from the separate collection of the wet biodegradable wastes. As pointed out by Ricci (2003), 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 the net additional costs for separate collection of wet biodegradable wastes are unlikely to exceed the additional cost of separate collection of other materials. Thus, the net additional cost of separate collection of wet biodegradable wastes is considered in a range of 0-30 Euro/Mg.

The process and emission data for HDPE containers, vehicles and vehicle operation are taken from the Ecoinvent database, version 1.2 (released June 2005).

An additional long-distance transport by barge from Malta to the European continent has been added to all materials for recycling from Malta. A distance of 800 km has been assumed, corresponding to the distance Malta-Barcelona. In scenario E, which does not foresee an incineration plant in Malta (see Chapter 8.5), this additional long-distance transport is also applied to the wastes transported to mainland Europe for incineration. The distance may be overestimated (the alternative distance to e.g. Naples is 300 km), but this is counter weighed by the fuel use per km being somewhat underestimated by adopting the Ecoinvent data for an "operation, barge" for inland watercrafts.

Data on injuries are calculated on the basis of 1995-2002 statistics on work related accidents as available on the Eurostat website (http://epp.eurostat.cec.eu.int). These are extrapolated to EU25 with a scaling factor of 120% derived from the relative incidence rates of fatalities in the EU15 and EU25, and then disaggregated to the more detailed industries by applying the same proportions between industries as in the US Bureau of Labor Statistics data (http://www.bls.gov/iif/home.htm), and finally related to the industry turnover as provided in the EIPRO dataset for EU25 (Tukker et al. 2005).

5.2.2 Modelling of bring collection

According to Tucker & Speirs (2002), it is unlikely that directive compliant or economically optimal recycling rates can be achieved by bring collection alone. Thus, bring collection should not be seen as an alternative to kerb-side collection, but as a complementary element of a multi-faceted collection system that optimises recycling by offering different suitable ways to dispose of household wastes.

Bring collection is a term covering very different systems, from neighbourhood collection points to central civil amenity sites, and it is therefore difficult to find data that cover all of these options. However, it is unlikely that the recycling targets can be reached with less than 1 collection point per 1,000 inhabitants.

Therefore a fixed design is used for bring collection for all scenarios, based on 1 collection point per 1,000 inhabitants (except the baseline scenario, where the current number of collection points in Krakow and Malta is used), an estimated weight of each container of 36-44 kg, with an estimated lifetime of 7-10 years, the same fuel consumption per Mg waste as for kerb-side collection, and a cost – without fuel – of 94-740 Euro/year/collection point, based on the range of costs given in Kranert et al. (2004).

The costs and environmental exchanges related to bring collection are divided equally over the total amount of recycled wastes (paper and board, plastics, glass and metals) at the 55% recycling target, which are 24,000 Mg/year in Malta and 69,000 Mg in Krakow (see Table 33 and Table 34, respectively).

With 400,000 inhabitants in Malta and 760,000 in Krakow, the total costs of operating the bring systems will be 37,600 – 296,000 Euro/year in Malta and 71,440 – 562,000 Euro/year in Krakow or 0.86 – 6.9 Euro/Mg recyclable waste in Malta and 0.56 – 4.5 Euro/Mg recyclable waste in Krakow.

5.2.3 Input-output data on waste collection

Most national input-output tables have only a general category for waste collection and treatment. In the EIPRO study data (Tukker et al. 2005), waste collection was included in the general category “Trucking and courier services, except air” while waste treatment was included in “Sanitary services, steam supply, and irrigation systems”. The only country for which separate input-output data on waste collection were found is Denmark, where Weidema et al. (2005) provided data for “Refuse collection and sanitation” separate from data on waste treatment.

To compare the Danish data on “Refuse collection and sanitation” with the process-based data for waste collection from Chapter 5.2.1, first cleaned was the data for a 16.42% transfer payment for the waste treatment, and then scaled the data to 1 Mg waste collected, using the total Danish waste amount in 1999 (9.5 Tg annually according to the Danish Waste Statistics). The resulting value of 83 Euro/Mg corresponds well with the average from Table 10.

The Danish “Refuse collection and sanitation” industry uses 850 TJ fuel annually. This is equivalent to 20 Gg or 24 Tl diesel or 2.5 litre diesel/Mg waste. This is slightly more than the 1.9 litre/Mg waste assumed in Chapter 5.2.1, and may be explained by the larger average distances when both rural and urban areas are covered. Also, the Danish “Refuse collection and sanitation” industry provides mainly paper bags (0.87 Euro/Mg waste), where the assumption in Chapter 5.2.1 is that HDPE containers are used. Thus, the diesel input was scaled, as well as the emissions with a factor 1.9/2.5, and the paper bag input was deleted.

Assuming that a scaling via the diesel consumption is appropriate, with these corrections, the resulting process becomes directly comparable to the process-based data (the process “Transport, municipal waste collection, lorry 21t”, which also does not include the HDPE containers); see Table 11. The sum of all inputs of goods and services amounts to 43.64 Euro/Mg. The difference up to 83 Euro/Mg is the value added (i.e. wages, taxes and profits). The comparison in Table 11 presents a larger detail and completeness of the used input-output-based cost data for waste collection compared to the specific process-based data used.

Table 11 **Inputs accounted for in the process-based and input-output-based data for waste collection**

Inputs (supplies)	Ecoinvent process: “Transport, municipal waste collection, lorry 21t” corrected with data from Kranert et al. (2004) [Euro/Mg waste]	Input-Output-based data: “Refuse collection and sanitation, DK” cleaned for transfer payments and paper bags [Euro / Mg waste]
Lorry, maintenance and diesel	10.59 [1]	6.72
Construction (Buildings and civil engineering)	[2]	4.26
Telecommunications and postal services		3.47
Business activities not elsewhere classified		3.13
Wholesale and retail trade		2.68
Consulting engineers, architects etc.		2.54
Software consultancy and supply		2.02
Advertising		1.96
Detergents & other chemical products		1.30
Public infrastructure and administration		1.24
Accounting, book-keeping, auditing etc.		1.21
Electrical machinery		0.98
Computer activities excl. software		0.92
Construction materials		0.92
Radio and communication equipment		0.89

Inputs (supplies)	Ecoinvent process: "Transport, municipal waste collection, lorry 21t" corrected with data from Kranert et al. (2004) [Euro/Mg waste]	Input-Output-based data: "Refuse collection and sanitation, DK" cleaned for transfer payments and paper bags [Euro / Mg waste]
Industrial cleaning		0.80
Air transport		0.49
Taxi operation and coach services		0.43
Agricultural services and landscape gardeners		0.40
Machinery for industries		0.40
Repair and maintenance of buildings		0.38
Non-life insurance		0.38
Activities of membership organisations		0.35
Restaurants and other catering		0.35
Transport via railways		0.32
Marine engines, compressors		0.32
Office machineries and computers		0.30
Laundries and dry cleaners		0.29
Minor inputs (each less than 0.29)		4.19
Total [Euro / Mg waste]		**43.64**

[1] Prices from Kranert et al. 2004, see Table 10, and fuel price of 0.54 Euro/litre
[2] A small amount of road infrastructure and maintenance (0.0076 year-metres) is included in the Ecoinvent process.

5.3 Waste treatment technologies

For each waste treatment technology, specific transfer coefficients link the substance composition of each waste fraction to emissions in the different output compartments (air, water, soil). For example, the incineration specific transfer coefficients for nickel says how much of the nickel in the wastes can be expected to end up in air, surface-water, groundwater, and soil.

Transfer coefficients and consumption data were identified through a systematic search in the journal "Waste Management and Research" (last 5 years), supplemented by other readily available in-house data (see below). In addition, specific data from two municipal solid waste incinerators Vestforbrænding (www.vestfor.dk) and Amagerforbrænding (www.amfor.dk) were collected from literature and green accounts. Amagerforbrænding are using semi-wet flue gas treatment technology. Questions for the Danish plants were addressed by personal contact to Uffe Juul Andersen (Amagerforbrænding) and Niels Groth Andersen (Vestforbrænding) (both 2005.08.16).

It appears that very few real analyses of transfer for specific wastes and waste fractions exist, and that these are cited and re-cited extensively. This study relies especially on Belevi & Moench (2000), Chandler (1994), Chandler et al. (1997), Christensen (2001), Goux & Douce (1995), and Reimann (1994). A specific detailed check was made for lead, using literature data and data from the specific incinerator Amagerforbrænding.

The values provided by Doka (2003) and in the references cited herein were checked explicitly against the other data collected, and it was concluded that the transfer coefficients from Doka (2003) are correctly cited and within the range of the literature data. The data from Doka (2003) are therefore applied in general, with the modifications described below.

Data on injuries are calculated in the same way as for waste collection, see Chapter 5.2.1.

5.3.1 Modelling of landfilling

Table 12 presents the costs of landfilling, based on data from Bozec (2004) calculated for landfills regarding given specifications. The calculated costs for a medium sized (120 Gg/year) uncontrolled or inert landfill (32 Euro/Mg waste) and a similar sized directive compliant landfill (58 Euro/Mg waste) fit well with the range of gate fees, excluding taxes, collected by Hogg (2001). Based on these ranges, the uncertainty on the cost is estimated at +/- 40%, i.e. 34-82 Euro/Mg waste for the large directive compliant landfill, and 19-45 Euro/Mg waste for the large uncontrolled landfill.

Table 12 Landfill specifications and costs.

Waste capacity	**Mg/year**	**80,000**	**120,000**	**160,000**
Waste density	Mg/m^3	0.9	0.9	0.9
Passive security thickness	m	1.5	1.5	1.5
Cover layer thickness	m	1.2	1.2	1.2
Useful height	m	12.3	12.3	12.3
Operation surface (20 years)	ha	20	30	39
Total surface (20 years)	ha	74	93	112
Volume of removed soil (per year)	m^3	29,000	44,000	58,000
Bottom membrane surface (per year)	m^2	12,000	17,000	22,000

Waste capacity	Mg/year	80,000	120,000	160,000
Cost of site preparation, constructions, installations, roads, equipment, engineering and administration in relation to site opening [1]	Euro/Mg waste	20	14	11
Costs for excavation, passive security layer, drainage, cover layer, labour, environmental monitoring and administration [2]	Euro/Mg waste	26	25	25
Closure and post-closure costs	Euro/Mg waste	16	15	14
Leachate treatment costs	Euro/Mg waste	4	3	3
Biogas treatment costs [3]	Euro/Mg waste	1	1	1
Total costs, directive compliant landfill	Euro/Mg waste	67	58	54
Total costs, uncontrolled or inert landfill [4]	Euro/Mg waste	n.r.	32	27

[1] These costs are noted down as “Capital costs” in Bozec (2004)
[2] These costs are noted down as “Operating costs” in Bozec (2004)
[3] Does not include possible income from sale of biogas or electricity
[4] For this type of landfill, costs for leachate and biogas treatment is omitted, and only 10% of “Operating costs”, cf. note 2, are included.

It should be noted that the applied landfill specifications (see Table 12) are somewhat different from the ones used by Doka (2003), i.e. here we have:

- 3 times the land occupation, 3.5 times the amount of gravel, sand and diesel and 1.85 times the materials used for the bottom membrane, due to more realistic design specifications,
- 2/3 of the excavation volume,
- no use of cement for solidification of the wastes, as this is not a very widely used technique, and is not regarded as BAT due to the significant environmental impacts of cement manufacture.

For the emissions, the transfer coefficients of Doka (2003) were applied for both uncontrolled landfill and directive compliant landfill, noting that for uncontrolled landfill the leachate is assumed to go directly to groundwater, with no surface run-off into streams or rivers.

Electricity production from landfill gas (at the directive compliant landfill) is calculated with the assumption from Doka (2003) that 53% of the landfill gas is collected, and applying the same combustion efficiency as for the composting process (38%; see Chapter 5.3.3). The net avoided electricity production is modelled as produced from oil- and coal-fired power plants for Malta and Krakow, respectively.

For upstream processes, i.e. the production of inputs to the landfill process, such as gravel and plastic pipes, data on emissions are taken from the Ecoinvent database, with a few modifications, see Chapter 5.4.

Emissions from landfill fires have not been included in the study. It should be noted that fires in waste materials do not exclusively occur in landfills, but may as well occur in temporary waste deposits, e.g. during storage before incineration or recycling.

5.3.2 Modelling of incineration

Table 13 presents the costs of incineration, based on data from Bozec (2004) for capacities between 150,000 and 250,000 Mg/year and 20 years lifetime. The technology is a grate incinerator with electrostatic precipitator for fly ash, semi-dry flue gas cleaning, and non-catalytic reduction (SNCR) of NOx. Reported uncertainties relate only to differences in scale and variations within the said technologies. Compared to the ranges collected by Hogg (2001) for incineration plants of the same size, these values lie in the lower end.

Table 13 Costs of incineration. Based on Bozec (2004).

Type of cost	($Euro_{2003}$ per Mg waste)
Capital costs	17-20
Operating and maintenance costs	10-11
Reagent costs	3.5
Landfilling of residuals	9
Net sale of electricity [1]	-56 - -40
Total costs	**-16.5 – 3.5**

[1] 0.08 Euro/kWh * (heating value 8.4-11.3 GJ/Mg * 25% efficiency * 278 kWh/GJ – 80kWh/Mg internally used).

As a sensitivity analysis, two other emission reduction technologies were also modelled, generally believed to be more environmentally benign, namely wet flue-gas cleaning and catalytic reduction of NOx (SCR), using the ranges provided in the BREF-note (JRC 2005a). The environmental advantages of these two technologies are largely offset by their additional consumption of especially electricity, see Table 14. Thus, the above configurations (semi-dry FGC and SNCR) are applied in all 5 scenarios.

For Malta, the large amount of wet biodegradable wastes results in quite low heating values for the incinerated wastes (approximately 8.24 MJ/kg) for scenarios B and D, which therefore require the use of support fuel (gas oil). Interpolating the values from Treder & Salamon (2005) for different heating values, gives a support fuel requirement of 0.85 litres of gas oil (= 29 MJ) per Mg waste. The support fuel contributes to the electricity production from waste incineration, but with a lower efficiency (25%) than in the displaced dedicated electricity plant (29%), and thus adds slightly to the overall environmental impacts from waste incineration for Malta.

Table 14 Life cycle impact of incineration of 1 Mg "Other wastes" in Krakow with different flue gas cleaning (semi-dry and wet) and NOx-reduction (SNCR and SCR) technologies.

Impact category	Baseline: SNCR; Semidry FGC	SCR; Semidry FGC	SNCR; Wet FGC
Climate change	294.00	296.00	303.00
Respiratory inorganics	18.40	15.10	13.10
Human toxicity	2.70	2.70	3.80
Ecotoxicity, terrestrial	1.95	1.94	2.98
Nature occupation	1.02	1.08	1.24
Injuries, road or work	1.00	1.00	1.00
Eutrophication, terrestrial	1.20	0.70	1.00
Photochemical ozone - Vegetation	1.05	0.73	0.96
Acidification	0.74	0.91	0.20
Ecotoxicity, aquatic	0.35	0.35	0.67
Respiratory organics	0.04	0.03	0.04
Sum	**322.25**	**320.54**	**327.99**

All impacts measured in $Euro_{2003}$, using the site-generic method described in Chapter 6.

The inputs to the incineration process (besides the wastes) are modelled as in the Ecoinvent database (Doka 2003), but adding to the inventory data 0.6 g activated carbon and 1.5 g lignite coke per kg waste (based on Bozec 2004), as these inputs are missing in the Ecoinvent processes. The lifetime of the incineration plant is reduced from 40 to 20 years (based on Bozec 2004). General modifications to the Ecoinvent data apply as described in Chapter 5.4. The inputs and emissions are furthermore adjusted to model 100% specific non-catalytic NOx reduction, according to Doka (2003, part II, p. 41-45), and semidry flue gas scrubbing (data from Bozec 2004).

The emissions of dioxin were reduced from 3 ng TEQ / kg waste in Ecoinvent (Doka 2003) to 0,3 ng TEQ / kg waste, corresponding to 50% of the emission limit value of Directive 2000/76/EC. Even lower values should be achievable according to JRC (2005a).

The electricity generation is calculated from the lower heating value of the wastes, using an efficiency of 25% based on Bozec (2004). This is the gross efficiency, i.e. before subtracting the electricity use of the plant itself. The net avoided electricity production is modelled as electricity being produced from oil- and coal-fired power plants for Malta and Krakow, respectively.

Home incineration of paper (in Krakow, baseline scenario A) is assumed to replace home incineration of wood, with the same emissions. Thus, the process includes only the wood displaced by the paper incineration.

5.3.3 Modelling of composting

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. Thus, technology description was based 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 data applied are from a full-scale plant in Denmark, as described in Kjellberg et al. (2005).

The composting results in two products: compost and electricity. Per Mg of wet biodegradable waste, 340 kg of compost is produced, at 51% dry matter. Of the nitrogen in the wastes, 38% is lost in the composting process, according to Kjellberg et al. (2005). Depending on the fate of the compost, 0 - 65% of the nitrogen remaining in the compost is assumed to displace nitrogen in fertilizer. As larger quantities of compost will need to be disposed in the future scenarios, it is unlikely that all of this will be utilized in places where the full nitrogen value can be utilized, so the average utilization was assumed to be 1/3 of the 65%. Included was also the transport of the compost (4 - 25 km, with an average of 10 km) and the spreading on agricultural land with a solid manure spreader. It is unlikely that the compost will be transported further than absolutely necessary, due to its relatively small economic value per kg.

The electricity production from the biogas is between 302 and 427 kWh per Mg wet biodegradable waste (at 40% dry matter), with an average of 395 kWh/Mg. The average is based on an efficiency of 38% in the conversion from biogas to electricity; the low end of the range is assuming lower efficiency (29%), while the high end of the range denotes an increased methane yield compared to the process documented in Kjellberg et al. (2005). Per Mg of wet biodegradable waste, the process requires an input of 7.6 l fuel and 6 kWh electricity. The net avoided electricity production is modelled as being produced from oil- and coal-fired power plants for Malta and Krakow, respectively.

The process requires 256 g of structural material per Mg of kitchen waste, which may be garden wastes. Thus assumed was that the wet biodegradable wastes contain adequate amounts of park & garden wastes, so that there is no need for supply of external structural material.

The composting plant is composed of a closed reactor and a biogas facility, which was modelled as a slurry storage of 1.16 m^3 per Mg wet biodegradable waste, and an open composting plant, which was adopted here from the Ecoinvent database (Nemecek et al. 2004).

Emissions of ammonia, carbon monoxide, dinitrogen monoxide and methane are taken from a recent review by Ødegård et al. (2005). An additional 0.5% loss of methane in the valorisation plant is taken from Gunnarsson et al. (2005) and emissions of NOx and particles are modelled with the same data as for combustion of landfill gas (Doka 2003). Emissions of hydrogen sulphide have been taken from Nemecek et al. (2004) and non-methane volatile organic compound (VOC) emissions from DEFRA (2000). VOC emissions are mainly alkenes and have been specified as such to obtain the appropriate characterisation in impact assessment.

Other air emissions are modelled with the same transfer coefficients as for landfill (see Chapter 5.3.1), assuming an 80% degradation of the wet biodegradable wastes during composting. 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.

There are no water emissions from the compost plant, since the excess water is reintroduced into the reactor chamber. Emissions to groundwater from compost deposited on farm or garden soil are modelled analogously to short-term (<100years) releases from a landfill with the same degree of waste decomposition (80%). The compost is not modelled as an emission to soil, since the substance composition assumed (as derived from the composition of wet biodegradable waste in Chapter 5.1.3) does not differ from the composition of ordinary soil.

It should be noted that it is assumed that either there are no impurities or that any impurities from other wastes are separated out at the composting plant, before the compost is deposited on farm and garden soils. The extent to which this will influence the environmental performance of this scenario requires further investigation and is the subject of on-going studies.

In addition to the above-described composting technology, the baseline scenario A also applies a central composting technology without energy recovery. Here, the data from Nemecek et al. (2004) are 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).

The cost of central composting has been modelled according to the data of Bozec (2004) for windrow composting (Scenario A) and anaerobic digestion (Scenario C), resulting in total costs of 19 Euro/Mg for scenario A and 38-56 Euro/Mg for scenario C. The value of the compost is 0-10 Euro/Mg or 0-4.5 Euro/Mg wet biodegradable waste, and at 0.08 Euro/kWh the electricity provides an income of 24-34 Euro/Mg wet biodegradable waste, with a best estimate of 32 Euro/Mg. The net cost of the BAT composting is therefore in the range of -0.5-32 Euro/Mg, with a best estimate of 13 Euro/Mg wet biodegradable waste.

Home-composting – applied in scenario A for Krakow – is modelled as an intermediate between aerobic and anaerobic digestion, resulting in a methane emission of 48 kg/Mg wet biodegradable waste. There are practically no investigations available on how well home composting performs. A midpoint was chosen between best and worst practice. At best, home composting has the same performance as central composting without energy recovery.

For upstream processes, i.e. inputs to the composting processes, data on emissions are taken from the Ecoinvent database, with a few modifications, see Chapter 5.4.

5.3.4 Modelling of material recycling

Material recycling is modelled with processes from the Ecoinvent database. Table 15 provides an overview of the recycling processes applied and the virgin material production displaced (avoided production). Loss of material during recycling (due to reduced quality of scrap relative to virgin materials) is included in the recycling processes.

Table 15 Processes applied for modelling of recycling.

Material	Recycling process	Avoided production
Cardboard	Corrugated board, recycling fibre, single wall, at plant/RER. Avoided electricity by-product added from the original data source (FEFCO et al. 2003).	Corrugated board, fresh fibre, single wall, at plant/RER. Avoided electricity by-product added from the original data source (FEFCO et al. 2003).
Newsprint	Paper, newsprint, DIP containing, at plant/RER	Paper, newsprint, 0% DIP, at plant/RER
Plastics	Plastics recycling at one specific plant “Replast”. Energy use and waste only. From Frees (2002).	The corresponding plastics granulate, at plant/RER
Glass	Packaging glass, brown, at plant/RER	Glass, virgin/RER
Iron and steel	Steel, electric, un- and low-alloyed, at plant/RER	Steel, converter, low-alloyed, at plant/RER

Material	Recycling process	Avoided production
Aluminium	Aluminium, secondary, from old scrap, at plant/RER. Input of zinc for coating ignored, as this is not included for primary aluminium.	Aluminium, primary, at plant/RER

Ecoinvent terminology; RER is the international three-letter abbreviation for Europe. DIP stands for recycled paper.

5.3.5 Input-output data for waste treatment

The commonly given argument related to using input-output-based data when available for specific materials is the completeness of upstream processes and, hence, related emissions and resource consumption data, while the methodology remains to be critically evaluated; see also Chapter 5.4. For the direct emissions from the waste treatment technologies, the above described process-based data are expected to provide a more complete and accurate model than emission data from input-output tables (described in Chapter 5.4). Thus, input-output data for the direct emissions for the waste treatment processes were not applied.

Material recycling poses a particular problem in input-output-tables, since the processing of primary and a secondary raw materials are taking place in the same aggregated industries, thus blurring the important environmental differences between these processing routes. For example, data for the steel industry include both basic oxygen furnaces using primary steel and secondary electrical arc furnaces using scrap raw materials.

For many applications of input-output-data, e.g. for prioritisation among product groups as in Tukker et al. (2005) and Weidema et al. (2005), where it can be assumed that inputs of recycled materials to the analysed systems are equal to the outputs supplied to recycling, this aggregation level does not give this problem. However, for systems, such as waste treatment, where focus is exactly on the output of recycled material, it is crucial to be able to distinguish the two processing routes.

In an attempt at investigating the potential degree of missing completeness of the process data for the materials production compared to input-output data, the US 1998 input-output data of Suh (2003) was compared to the process-based data for primary and secondary materials from Table 15, aggregated to the same level, using the US 1998 proportions between primary and secondary production. This approach did not demonstrate convincingly that the input-output data are more complete. A possible explanation for this may be that process-based data of the Ecoinvent database for the materials producing industries may be of higher quality (and thus completeness) than process-based data for service industries, such as waste collection, where significant incompleteness could be demonstrated (see Chapter 5.2.3).

Thus the conclusion is that for materials recycling, input-output-based data do not currently provide additional information compared to process-based data, although that might be expected also for the materials producing industries. Input-output data potentially could contribute to the completeness of the corresponding process-based data, if the required more fundamental analysis of the method shows the principle correctness of the inventory results.

5.4 Upstream processes

5.4.1 Inventory database

Upstream processes, i.e. inputs to the waste collection and treatment processes are all taken here from the Ecoinvent database, version 1.2 (released June 2005). No comparison with other databases was performed, as this was not in the scope of this project.

Injury data are added to the Ecoinvent road transport processes, since road transport is the by far largest contributor to overall injuries. For this, values of 2.3 fatal injuries and 33 non-fatal injuries per 1E8 vehicle-km are applied, calculated from National Safety Council (2004) incidence rates.

Due to the importance of the electricity supply for the overall results for some impact categories, all supplies of electricity in the Ecoinvent database were changed to electricity supplied by modern coal fired power plants (the long-term marginal electricity for central Europe, according to Weidema 2003). This was done as the study uses the change-oriented (marginal) modelling approach. An exception is the electricity supply for primary aluminium production, where the aluminium industry has documented that their long-term marginal supply is close to the current average supply (Weidema 1999). As proxy for modern coal fired power, data for German average coal fired technology were used.

While data for the processes actually affected is preferable, as specified by ISO 14049, and especially for all waste treatment processes, this was not possible for all upstream processes when these were taken from available LCA databases such as Ecoinvent that generally present data as industry averages.

5.4.2 Input-output data

As stated in Chapter 5.2.3, most national input-output tables have only a general category for waste collection and treatment. In e.g. the EIPRO study data (Tukker et al. 2005), waste treatment is included in “Sanitary services, steam supply, and irrigation systems”. Separate input-output data on waste treatment is available for Denmark, where Weidema et al. (2005) provided data for “Refuse dumps and refuse disposal plants” separately from data on waste collection. Yet, this still does not allow distinguishing between different waste treatment technologies.

Danish refuse disposal is mainly done via incineration. Table 16 therefore compares the Danish data on inputs to “Refuse dumps and refuse disposal plants” with the process-based data for inputs to incineration from the Ecoinvent database, as described in Chapter 5.3.2. The comparison here is for upstream inputs only, which includes for both the process-based data and the I/O data “the supplier of the supplier of the supplier’s emissions”. There is no comparison of the direct emissions from the incineration, as these are best represented by the process data (Chapter 5.3.5).

The total Danish waste amount for treatment was 4.4 Tg in 1999 (2.9 Tg incinerated and 1.5 Tg landfilled according to the Danish Waste Statistics, DEPA 2002) at a treatment cost of 91 Euro/Mg, which is somewhat higher than the costs calculated in Table 12 and Table 13.

Table 16 **Inputs accounted for in the process-based and input-output-based data for waste treatment (incineration, not including waste collection). Data are given in Euro per Mg waste.**

Inputs (supplies)	Process: “Disposal, to municipal incineration” from Ecoinvent database for the composition of waste to incineration in Krakow Scenario B	Input-Output-based data: “Refuse dumps and refuse disposal plants, DK”
Electronics and electrical machinery		11.25
Construction (Building and civil engineering)		8.97
Wholesale and retail trade		7.50
Construction materials	Gravel, sand, cement, bitumen and steel included	6.06
Consulting engineers, architects etc.		5.88
Machinery		4.22
Telecommunications and postal services		2.00
Advertising		1.94
Public infrastructure and administration		1.69
Renting of machinery and equipment etc.		1.54
Industrial cleaning		1.47
Software consultancy and supply		1.39
Business activities not elsewhere		1.36

Inputs (supplies)	Process: "Disposal, to municipal incineration" from Ecoinvent database for the composition of waste to incineration in Krakow Scenario B	Input-Output-based data: "Refuse dumps and refuse disposal plants, DK"
classified		
Accounting, book-keeping, auditing etc.		1.29
Detergents & other chemical products	Most important chemical inputs are included	1.22
Freight transport by road (other than waste)	0.008 (0.065 tkm)	1.17
Furniture		0.90
Computer activities excl. software		0.85
Repair and maintenance of buildings		0.83
Hand tools etc.		0.63
Minor inputs (each less than 0.63)	Energy inputs are accounted	8.60
Total [Euro / Mg waste]		**70.77**

As for waste collection (Chapter 5.2.3), the input-output-based cost data for inputs to waste incineration are more complete than the one available/derived from in the Ecoinvent database.

6 Environmental impact assessment methods

According to ISO 14040, "Life Cycle Impact Assessment (LCIA) is a phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a given product system throughout its life cycle", i.e. from the time natural resources are extracted from the ground and processed through each subsequent stage of manufacturing, transportation, product use, and ultimately, disposal.

Life Cycle Impact Assessment provides indicators and methods for analysing the potential contributors of the inventory data to different impacts categories, such as climate change, contribution to acidification, land use, etc. and, in some cases, in an aggregated way. After compilation, tabulation, and preliminary analysis of the life cycle inventory, it is necessary to calculate, as well as to interpret, indicators of the pressures or impacts that are associated with emissions to the natural environment and the consumption of resources. Life cycle impact assessment provides indicators for the interpretation of the inventory data in terms of contributions to different impact categories, or environmental burdens. The indicator results facilitate the evaluation of products, and each stage in a life cycle, in terms of climate change, contributions to toxicological pressure, land use, etc.

The scope of the evaluation is, with some exceptions, limited to the consideration of contributions to impacts at the regional and global scales. The overall indicator results reflect cumulative contributions to different impact categories, summed over time and space. These regional and global insights compliment information from e.g. more detailed site-specific assessments.

Two impact assessment methods are applied in this study:

- A mainstream "midpoint category method", where the analysed systems are compared at the level of midpoint impact indicators, i.e. with one indicator for each environmental impact category (acidification, ecotoxicity, etc.) and without further aggregation of the results.
- An "endpoint" or "damage category method", where the midpoint category results are here further modelled in damage categories to give cross-comparable indicators ("Human production and consumption efficiency"). In an additional step in this study, usually optional and not recommended in some applications in e.g. ISO standards, these are then weighted to be expressed in monetary units (Euro).

6.1 Midpoint impact assessment method

6.1.1 Choice of impact categories, category indicators and characterisation models

Recent reviews of the state-of-the-art of life cycle impact assessment can be found in e.g. Udo de Haes et al. (2002) and Pennington et al. (2004). Among the different existing impact assessment methods considered, there is a reasonable similarity in terms of which impact categories are included. The difference between the methods is rather in the models applied to characterise each impact category.

For each impact category, a category indicator is chosen and a characterisation model is applied to convert the relevant inventory results to a common unit, i.e. the unit of the category indicator. A combination of characterisation models was generally selected from two recent impact assessment methods, the IMPACT2002+ v. 2.1 and the EDIP2003 methods (Jolliet et al. 2003, Humbert et al. 2005, Hauschild & Potting 2005, Potting & Hauschild 2005). Both methods are second-generation methods, building partly on previous work (e.g. Ecoindicator1999 and EDIP1997, respectively).

The main criterion for choosing a specific characterisation model was the degree of completeness in coverage, both in terms of how much of the impact chain is covered by the model, and in terms of substances included (especially relevant for toxicity).

While still a topic of scientific debate in relation to significance, another criterion for selecting the IMPACT 2002+ and EDIP 2003 characterisation models was 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 EDIP 2003 method provides site-dependent characterisation factors for the impact categories acidification, eutrophication and photochemical ozone formation, for most European countries, including Poland. Characterisation factors for Malta have been developed specifically for this project; see Annex III.

For the ecotoxicity and human toxicity impact categories, an updated version of IMPACT 2002 (Pennington et al. 2005, 2006) was developed for the present project, with spatial European boxes nested in a multi-continental model; see Annex II. The spatially differentiated model allows to apply specific characterisation factors for Malta and Krakow. The introduction of a multi-continental model, including an Eastern-Europe box, is especially important for these two study areas, as they are both very close to the border of the original European region model.

Due to its overall importance to human health, the impact category “Injuries” (see Chapter 6.1.7) is added to complement the impact categories from IMPACT2002+

and EDIP2003. With this addition, the midpoint impact assessment method is likely to cover most important environmental (biophysical) impact categories related to waste management activities.

An issue of particular relevance to waste management is the treatment of emissions to groundwater, including the issue of emissions from landfills over the very long-term (i.e. > 100 years). In general, the methods from IMPACT2002+ and EDIP2003 do not treat emissions to groundwater separately, i.e. characterisation factors for water emissions are all related here to direct emissions to surface waters (or to soils as an alternative). Although groundwater emissions from landfills may eventually reach surface waters, there can be e.g. a significant binding of contaminants to soil particles. To be able to show the importance of applying different characterisation factors to groundwater emissions, separate impact categories are used for these emissions.

In a first test run of the impact assessment, the assessment results were dominated by ecotoxicity and human toxicity caused by emissions of aluminium from the processes derived from the Ecoinvent database. This was considered a potential artefact caused by the lack of distinction in the Ecoinvent database between aluminium in its metallic form and aluminium in its ionic form. The characterisation factors for aluminium in the IMPACT 2002+ method are potentially only intended for aluminium in its bioavailable form, assumed to be the ionic form, and do not take into account e.g. irreversible binding in soils. However, the basis of such characterisation factors in relation to e.g. the fate and toxicity data adopted is not generally transparent. Furthermore, this is the topic of ongoing studies aiming to improve such factors for LCIA. As a working basis, this potential artefact was addressed here by transferring the IMPACT 2002+ characterisation factors for aluminium to "aluminium ion". Metallic emissions of aluminium were thus given characterisation factors of zero. A similar problem may apply to other emissions of metals such as nickel, zinc, copper and chromium, and persistent organic chemicals.

EDIP 2003 characterisation factors are available for both 1990 and 2010, the latter being based on emission forecasts. In general, the 2010 factors are applied, since the state of the environment in year 2010 is more relevant as a background scenario for the emissions studied in the current project.

6.1.2 Acidification

For acidification, the EDIP2003 characterisation model is applied and has the category indicator in “m^2 unprotected ecosystem”, i.e. the ecosystem area that is brought to exceed the critical load for acidification as a consequence of the emission. Specific characterisation factors for Krakow and Malta apply where appropriate (Annex III.).

Earlier characterisation models for acidification are based on the potential of substances to release hydrogen ions, i.e. the theoretical maximum acidification, and thereby did not take into account differences in emission deposition patterns, background deposition levels, and the sensitivity of the receiving ecosystems. The EDIP2003 characterisation model for acidification uses the RAINS model (Amann et al 1995) to overcome these limitations of the earlier models. The RAINS model has also been applied in other policy support studies in relation to acidification.

6.1.3 Ecotoxicity

Aquatic and terrestrial ecotoxicity (i.e. ecotoxic impacts on aquatic and terrestrial ecosystems) are treated separately, i.e. as two separate impact categories, using the IMPACT 2002 model (Pennington et al. 2005). New characterisation factors are calculated for this project, using spatial European boxes nested in a multi-continental model; see Annex II. These factors account for differences in the fate of and expose to chemicals in the environment. The category indicators are here in “kg-equivalents triethylene glycol into water” and “kg-equivalents triethylene glycol into soil”, respectively.

The IMPACT 2002 model for ecotoxicity includes several improvements in fate and exposure modelling compared to earlier methods (Jolliet et al. 2003, Pennington et al. 2004, 2006), and it is continuously being developed further. Also, it covers more substances than other methods available at the time.

For terrestrial ecotoxicity, only characterisation factors for emissions to air and water are included. It would lead here to double-counting if also the more localised impacts of emissions to soil were included, such as copper and zinc to agricultural soils and heavy metals in mining overburden. As stated in Humbert et al. (2005), the local impacts of such emissions are already covered by the impact category "Nature occupation" in the IMPACT 2002+ method; see Chapter 6.1.10. The reasoning is that the impact during the 500 years relaxation after human use, which is included in the impact category "Nature occupation", also accounts for the long-term impacts from ecotoxic emissions to the soil during the human occupation.

6.1.4 Eutrophication

The EDIP2003 characterisation models treat aquatic and terrestrial eutrophication separately, i.e. as two separate impact categories. The category indicator for aquatic eutrophication is “kg NO_3^--equivalents” and for terrestrial eutrophication it is “m^2 unprotected ecosystem”, i.e. the ecosystem area that is brought to exceed the critical load for terrestrial eutrophication as a consequence of the studied emission. Specific characterisation factors for Krakow and Malta apply where applicable (Annex III. §1.2 and §1.3).

Earlier characterisation models for eutrophication did not distinguish between aquatic systems and terrestrial systems, actually modelling both as if they were impacts on aquatic systems. Also, they did take into account differences in emission deposition and transport patterns, background deposition levels, and the sensitivity of the receiving ecosystems. The EDIP2003 characterisation model for terrestrial eutrophication uses the RAINS model (Amman et al. 1995) to overcome these limitations of the earlier models, while for aquatic eutrophication the CARMEN model (Klepper et al. 1995) is applied to estimate the fraction of nutrient emissions that will actually reach and expose inland waters or marine waters.

For aquatic eutrophication, the EDIP2003 method provides specific factors for wastewater emissions and agricultural emissions. As agricultural emissions plays an insignificant role in a waste management context, only the characterisation factors for wastewater emissions were applied.

6.1.5 Climate change

For climate change applied was the IPCC 2001 characterisation model, with a time horizon of 100 years, as also applied by EDIP2003. The category indicator is "kg CO_2-equivalents".

The choice of 100 years as time horizon is recommended by IPCC. It should be noted that this does not imply a cut-off of impacts after 100 years, i.e. absolute impacts after 100 years are still taken into account. The 100 years is a reference time horizon when characterising the different substance contributions relatively (CO_2, CH_4, etc.).

All carbon emissions from waste treatment are handled in the same way, without regard to their origin (fossil or non-fossil). For non-fossil carbon, i.e. carbon of immediate biological origin, the basic assumption is that the human extraction of biomass reduces the CO_2 in the environment. In a life cycle assessment of a biomass-containing product, this avoided ecosystem CO_2 emission is therefore included as a credit in the extracting process, balancing the CO_2 emission when the biomass eventually is combusted. Hence, in a complete life cycle of a biomass-containing product, there is no net contribution to climate change.

In this assessment of waste management, the upstream processes leading to the wastes are not included in the analysed system, as mentioned in relation to the functional unit. Inclusion would not affect the relative results of the study, as the credit would equally apply to whatever waste management option is adopted. However, for displaced processes due to recycling of biomass (such as paper), the waste management system includes the displaced extraction of virgin biomass and thus the emission credit for avoided ecosystem CO_2 release attached to this process.

6.1.6 Human toxicity

For human toxicity, the IMPACT 2002+ methodology (Jolliet et al. 2003) is applied. However, in contrast to IMPACT 2002+, carcinogens and non-carcinogens are treated as one single impact category, for convenience. This implicitly assumes that cancer and non-cancer effects are of equal severity, similar to the common practice of assuming effects within these sub-groups are also equal (see e.g. Crettaz et al. 2001).

For impacts on human health (human toxicity, respiratory inorganics, etc.), the IMPACT 2002+ methodology implies the use of severity weights, allowing different diseases to be expressed relative to death using the concept of Disability Adjusted Life Years (DALYs), as developed by Murray & Lopez (1996). These severity weights can be based on choice modelling, i.e. by soliciting and aggregating value choices across individuals. Formally, where data are not based on statistical years of life lost, this is a kind of weighting in the sense of ISO 14042 (14044).

New characterisation factors are calculated for this project, using spatial European boxes nested in a multi-continental model; see Annex II. The category indicator is here in “kg-equivalents of chloroethylene emitted into air (carcinogenic effects only)”.

The IMPACT 2002 model for human toxicity includes several improvements in fate and exposure and toxicity modelling compared to earlier methods. Also, it covers more substances than other methods available at the time.

6.1.7 Injuries

The impact category "injuries" includes fatal and non-fatal injuries from road traffic and work (occupational injuries). The category indicator is “fatal-injury-equivalents”.

Hofstetter & Norris (2003) suggest a procedure for including work-related injuries in life cycle assessments. Estimated characterisation factors for both occupational and road traffic injuries from the overall proportion of YLL (Years of life lost) to YLD (Years-of-life-equivalents lost due to disability) for these causes in the Global Burden of Disease study (Mathers et al. 2004, using the values without discounting and age-weighting), compared to and using the proportion of reported cases from Eurostat (http://epp.eurostat.cec.eu.int) and the CARE Road Accident Database (http://europa.eu.int/comm/transport/care). With 43 YLL/injury-related death, 0.323YLD/non-fatal road injury, and 0.0333 YLD/non-fatal work injury, obtained is 43/0.323 = 133 non-fatal road injuries / fatal injury (death), and 43/0.0333 = 1300 non-fatal work injuries / fatal injury.

6.1.8 Ionizing radiation

For ionizing radiation, the IMPACT2002+ characterisation model has the category indicator "Bq-equivalents Carbon-14 into air". The IMPACT2002+ characterisation model is taken directly from Ecoindicator99 (Frischknecht et al. 2000).

6.1.9 Mineral extraction

For mineral extraction, the IMPACT2002+ characterisation model has the category indicator "MJ additional energy", "MJ extra" for short, the difference between the current energy requirement for extraction and an estimated future energy requirement for extraction from lower grade ores. The IMPACT2002+ characterisation model is taken directly from Ecoindicator99 (Goedkoop & Spriensma 2001), which is based on Müller-Wenk (1998).

Besides the model of Müller-Wenk, another similar characterisation model for mineral extraction seeks to reflect the damage from mineral extraction, namely that the current dissipation of mineral resources will force future generations to use ores of lower grades with a potential consequent increase in energy use. This model is the one of Steen (1999), which is based on the cost of extracting the resource from bedrock, i.e. the average concentration in the Earth's crust.

6.1.10 Nature occupation (land use)

The impact category "Nature occupation" covers the displacement of nature due to human land use. The category indicator is "m^2-equivalents arable land", representing the impact from the occupation of one m^2 of arable land during one year.

In the IMPACT2002+ method, a similar impact category exists under the name of "Land occupation", taken directly from Ecoindicator99 (Goedkoop & Spriensma 2001), where the impact is assessed on the basis of the duration of area occupied (m^2*years) multiplied with a severity score, representing the potentially disappeared fraction (PDF) of species on that area during the specified time.

Compared to this method, the following modifications were made:

- Application of an estimated severity of 0.8 PDF*m^2*years for the direct impact of urban and intensive agricultural land use (see Millennium Ecosystem Assessment 2005), which is intended to be representative of all species affected, while Goedkoop & Spriensma (2001) arrive at a larger severity, mainly because their value is representative of the more severe situation when looking at plant species only.

- Definition of PDF in terms of the potentially disappeared fraction (PDF) of endemic species, i.e. not including alien species, while the Ecoindicator99 methodology uses the species-area relationship, and therefore assesses all species occurrences as positive. Here, as a contrary, “Green urban land” is assessed as equal to “Continuous urban land”.

- Only 30% of naturally occurring species in pasture areas (meadow lands) are negatively affected by grazing (Landsberg et al. 1997), whereas the Ecoindicator99 method suggests an impact close to that of other agricultural land uses.

- For occupation of arable land (all land with potential for agriculture), inclusion of an additional severity of 0.88 PDF*m^2*years to represent the secondary impacts from current deforestation, calculated as the nature occupation during the later relaxation from deforestation. Current global deforestation is estimated to 1.5E11 m^2. In the absence of an adequate characterisation model, the relaxation time is assumed to be 500 years and the average severity during relaxation as 0.2. The resulting value is allocated over the current global use of arable land (1.7E13 m^2) to arrive at the additional severity of 0.88 PDF*m^2*years for all current uses of arable land.

Table 17 provides the resulting characterisation values.

Table 17 Characterisation factors for 1 m^2*year land occupation for different intensities of occupation.

Intensity of occupation	Direct impact	Deforestation impact	Sum of direct & deforestation impacts	Midpoint indicator
	PDF*m^2*years	PDF*m^2*years	PDF*m^2*years	m^2-equivalents arable land
Urban and intensive agricultural use of arable land				
Continuous urban land	0.80	0.88	1.68	1.00
Construction and dump sites	0.80	0.88	1.68	1.00
Green urban land	0.80	0.88	1.68	1.00
Conventional agriculture	0.80	0.88	1.68	1.00
Integrated agriculture	0.80	0.88	1.68	1.00
Intensive meadow land	0.80	0.88	1.68	1.00
Less intensive uses of arable land [a]				
Organic agriculture	0.71	0.88	1.59	0.95
Organic meadow land	0.61	0.88	1.49	0.89

Intensity of occupation	Direct impact	Deforestation impact	Sum of direct & deforestation impacts	Midpoint indicator
	PDF*m^2*years	PDF*m^2*years	PDF*m^2*years	m^2-equivalents arable land
Discontinuous urban land	0.52	0.88	1.40	0.83
Industrial area	0.34	0.88	1.23	0.73
Rail or road area	0.34	0.88	1.23	0.73
Use of non-arable land				
Pasture in high productivity areas	0.30	0.00	0.30	0.18
Forest land	0.10	0.00	0.10	0.06

[a] These values are adopted from Ecoindicator99 (Goedkoop & Spriensma 2001) by maintaining the original proportion between midpoint indicator values, relative to the values for urban and intensive land uses.

6.1.11 Non-renewable energy

For non-renewable energy resource dissipation, the IMPACT2002+ characterisation model has the category indicator "MJ total primary non-renewable energy", calculated from the upper heating value of the total primary energy of the extracted resources. This category indicator covers situations where the energy resource is rendered unavailable due to dissipation, both when the dissipation occurs through combustion of the energy carrier and when the energy resource is dissipated without combustion (e.g. when plastics are landfilled)."

6.1.12 Ozone layer depletion

For ozone layer depletion, the IMPACT2002+ characterisation model has the category indicator "kg-equivalents of CFC-11 into air" taken from the US Environmental Protection Agency Ozone Depletion Potential List.

In view of the relatively low importance of the overall damage from this impact category, as a result of reduction targets and bans, alternatives were not considered.

6.1.13 Photochemical ozone impacts on vegetation

For photochemical ozone impacts on vegetation, the EDIP2003 characterisation model has the category indicator of "$m^2 \cdot$ppm$\cdot$hours", i.e. the product of the area of vegetation exposed above the 40 ppb threshold of chronic effects (m^2), the annual duration of the exposure above the threshold (hours), and the accumulated hourly mean ozone concentration over the threshold (ppm) during daylight hours in the vegetation period.

Compared to earlier characterisation models, the EDIP 2003 model has separate characterisation models for exposure of vegetation and exposure of humans (see respiratory organics, Chapter 6.1.15). For both types of exposure, the EDIP2003 models have the following suggested advantages over earlier models:

- The ability to represent spatial variability of the ozone formation.
- A more straightforward interpretation of the characterisation result in terms of environmental damage, since it is modelled further along the impact chain to include exposure of human beings and vegetation instead of just predicting the potential formation of ozone.
- The availability of characterisation factors taking into account the situation in year 2010, which is important because of the dependence of the ozone creation potential on the background emission levels. The factors may therefore vary in time in a statistically significant way. The EDIP2003 characterisation factors for photochemical ozone formation have been developed using the RAINS model, which was also used for development of characterisation factors for acidification and terrestrial eutrophication (see Chapter 6.1.2 and 6.1.4).

The EDIP2003 characterisation factors for photochemical ozone formation were developed using the RAINS model, which was also used for development of characterisation factors for acidification and terrestrial eutrophication (see Chapter 6.1.2 and 6.1.4).

6.1.14 Respiratory inorganics

For respiratory inorganics, the IMPACT2002+ characterisation model has the category indicator “kg-equivalents of $PM_{2.5}$ into air”, i.e. particulate matter < 2.5 µm. The IMPACT2002+ characterisation model is taken directly from Ecoindicator99 (Goedkoop & Spriensma 2001), which is again based on Hofstetter (1998).

To avoid potential double-counting of particulate emissions, the three diameter classes (< 2.5 µm, > 2.5µm and < 10 µm calculated as PM_{10}-$PM_{2.5}$, and > 10 µm calculated as TPM-PM_{10}, which have separate characterisation factors, are kept separate in the inventory.

6.1.15 Respiratory organics (photochemical ozone impacts on human health)

For respiratory organics applied was the EDIP2003 characterisation model for photochemical ozone impacts on humans, which has the category indicator “person·ppm·hours”, i.e. the product of the number of persons exposed above the 60 ppb threshold[12] (persons), the annual duration of the exposure above the threshold (hours), and the accumulated hourly mean ozone concentration over the threshold (ppm).

Compared to earlier characterisation models, the EDIP 2003 model for exposure of humans has the same advantages as listed under exposure of vegetation (see photochemical ozone impacts on vegetation, Chapter 6.1.13).

6.1.16 Normalisation

The aim of the normalisation is to express the indicator results relative to a reference value, which should make the results easier to understand. Normalisation transforms a category indicator result by dividing it by the selected reference value. In some LCA software, the normalisation is done as a multiplication by a normalisation factor, which is then the inverse of the normalisation reference.

The normalisation reference applied here is the estimated potential impact per person in Europe for the year 1995. The normalised results therefore express the impact indicators in person-year-equivalents for each category impact. For example, 2 person-year-equivalents then represent the average potential impact attributable to two persons for one year, resulting from the overall contributions to this impact from the European area in year 1995.

These person-year-equivalents express relative impact potentials of each impact category separately, and should not be confused with the DALY concept mentioned in Chapter 6.1.6 or the QALY concept introduced in Chapter 6.2.1, which both express absolute human impact potentials that take into account relative severity and can be compared across impact categories.

Since the normalised results do not express any statement of importance of each impact category, they should not be aggregated or compared across impact categories. For example, 1 person-year for climate change is not directly comparable to 1 person-year for acidification. Comparison would require the consideration of the relative severity of climate change to acidification, a step considered in later sections on endpoint impacts (see next section).

In addition to the environmental indicators, direct economic costs are given in $Euro_{2003}$ (see Chapter 7) and the normalisation value is a GDP of 23,200 $Euro_{2003}$ per person-year. Table 18 provides a summary of the normalisation values.

[12] No threshold for chronic exposure of humans to ozone has been established. Instead, following the revised Air Quality Guidelines for Europe (WHO 1998). The political threshold of 60 ppb is chosen as the long-term environmental objective for the EU ozone strategy proposed by the World Health Organisation.

Table 18 Normalisation references and factors per person in Europe for year 1995.

Impact category	Unit of characterised values	Normalization factors (Europe 1995)		Source
		Characterised unit / person-year (normalisation references)	Person-year / characterised unit (normalisation factors)	
Acidification	m^2 UES	2200	4.55E-04	[1]
Ecotoxicity, aquatic	kg-eq. TEG water	704000	1.42E-06	[2]
Ecotoxicity, terrestrial	kg-eq. TEG soil	71200	1.40E-05	[2]
Eutrophication, aquatic	kg NO_3-eq.	58	1.72E-02	[1]
Eutrophication, terrestrial	m^2 UES	2100	4.76E-04	[1]
Climate change	kg CO_2-eq.	10620	9.41E-05	[3]
Human toxicity	kg C_2H_3Cl-eq.	335	2.99E-03	[2]
Injuries, road or work	fatal injuries-eq.	0.000142	7.04E+03	[4]
Ionizing radiation	Bq C-14-eq.	533000	1.88E-06	[5]
Mineral extraction	MJ extra	292	3.42E-03	[5]
Nature occupation	m^2 arable land	2915	3.43E-04	[6]
Non-renewable energy	MJ primary	152000	6.58E-06	[5]
Ozone layer depletion	kg CFC-11-eq.	0.204	4.91E+00	[5]
Photochemical ozone – Vegetation	m^2*ppm*hours	140000	7.14E-06	[1]
Respiratory inorganics	kg PM2.5-eq.	8.8	1.14E-01	[5]
Respiratory organics	person*ppm*hours	10	1.00E-01	[1]

[1] See Annex III. §2
[2] See Annex II. §6
[3] Gugele et al. (2005)
[4] Calculated from 39400 fatal and 1390000 nonfatal road injuries (Data from the CARE Road Accident Database for EU15 extrapolated to EU25 using a factor 1.32 from Eurostat road fatality data), and 6460 fatal and 5740000 non-fatal work injuries (Eurostat data for EU15 extrapolated to EU25 using a factor 1.2).
[5] IMPACT2002+ v.2.1 (Annex 3 in Humbert et al. 2005)
[6] Calculated from the normalisation data of Humbert et al. (2005), using the characterisation factors from Table 17.

6.1.17 Weighting and relationship to ISO 14042 / 44

Although normalised results do not express any statement of importance of each impact category, it is difficult to avoid an unconscious 1:1 weighting across the normalised indicator results. Some impact assessment methods seek to avoid this by specifying alternative weighting factors that can be applied to the normalised indicator results.

Since both distance-to-target and panel approaches tend to arrive at weights very close to each other, and any difference is likely to be arbitrary, it is more transparent to simply recommend no weighting for the midpoint assessment, with the explicit warning that any attempt at interpreting this as an implicit 1:1 weighting is unlikely to reflect the true differences in importance between the impact categories.

With the decision to recommend no weighting of the normalised impact category results, the entire midpoint assessment is in accordance with the requirements of the ISO 14044 (former 14042) standard for comparative assertions.

The ISO 14044 (former 14042) standard clearly states that weighting shall not be used for comparative assertion disclosed to the public. The concern is that of the value choices involved. However, value choices are already applied in many characterisation models, for example human toxicity, where different toxic substances have different exposure routes, lead to different diseases, of very different significance to human health. When such different impacts are subsumed under one single impact category, it is necessary to apply severity weights to characterise the relative weight of different health impacts. This can involve a very explicit use of value choices, since the severity weights are derived for some effects, depending on weather morbility is significant relative to mortality, by soliciting and aggregating value choices across individuals.

This example shows that the ISO requirement of avoiding weighting for comparative assertions disclosed to the public can be circumvented by defining the impact categories more broadly, so that the weighting becomes part of the characterisation, and therefore not an aggregation “across impact categories”. In this approach, one single impact category (e.g. “Human well-being”) is defined and all environmental impacts are subsumed under this category, using appropriate characterisation factors, including the necessary value choices in the characterisation models. This is the approach applied for the endpoint impact assessment in Chapter 6.2.

Another way of avoiding formal weighting is to perform the normalisation (see Chapter 6.1.16) relative to a scenario that expresses the desired situation, i.e. a reference scenario based on value choices. The normalised result will then be equal to the result that would be obtained by a weighting, while conforming to the formal ISO requirement. This approach is recommended by Stranddorf et al. (2003), but not adopted here.

6.2 Endpoint impact assessment method

6.2.1 Choice of impact categories, category indicators and characterisation models

Three endpoint impact categories are defined:

- Ecosystem impacts, with the category indicator "PDF*m^2*year", i.e. Potentially Disappeared Fraction (PDF) of endemic species, the area affected (m^2), and the period of the effects (year).
- Human well-being, with the category indicator "QALY", i.e. Quality Adjusted Life Years calculated as the number of human life-years affected multiplied by a severity score (quality adjustment) between 0 and 1, where 0 is equal to death and 1 is equal to perfect well-being. 1 QALY = -1 DALY (Disability Adjusted Life Year, as defined by the Global Burden of Disease Study (Mathers et al 2004)).
- Economic production, with the category indicator "EUR_{2003}", i.e. the currency unit Euro at its average value in year 2003.

The starting points for these impact categories are the category indicator results from the midpoint impact assessment method. In principle, the starting point could also be the inventory result, thus circumventing the midpoint indicators. However, it is seen as an advantage that consistent results can be obtained at both midpoint and endpoint level by combining the two methods in a single framework.

As mentioned in Chapter 6.1.1, the midpoint impact assessment kept separate impact categories for groundwater emissions before and after 100 years.

IMPACT2002+ and EDIP2003 characterisation factors for water emissions are all related to direct emissions to surface waters. As some emissions to groundwater can take place over much longer time periods than e.g. many direct emissions from industrial processes, the concentrations of the emissions are can be lower, which can also imply a lower toxic impact, for example when the concentration of an essential element is reduced below a threshold. Also, some emissions can be irreversibly bound to soil particles. No LCA characterisation model is currently available that takes into account these particular special conditions for e.g. metals or groundwater emissions.

The conducted assessment demonstrates (see Chapter 9), the importance of keeping groundwater emissions separate, using either the same characterisation factors as for surface water emissions (i.e. without reduction factors) or e.g. using specific characterisation factors for groundwater emissions, where the original characterisation factors for surface water are reduced.

The reduction factors are calculated here, as a first approach, to represent the reduced concentrations of a groundwater emission over 100 years and over 60,000 years relative to the concentrations that would result from the same quantity of emission in a freshwater system. Assuming that an emission resides within a freshwater system for an average period of 2 weeks, the reduction factors for groundwater becomes 2 weeks / 100 years = 2/(52*100) = 4E-4 and 2 weeks / 60000 years = 2/(52*60,000) = 6E-7 for the emission before and after 100 years, respectively. It should be noted that this distinction between the time period over which an emission occurs is not consistent with the assumption that all mass emitted contributes to an environmental burden, irrespective of when it is emitted or the duration over which it is emitted.

The three endpoint impact categories are later aggregated, first by expressing ecosystem impacts in terms of human well-being (Chapter 6.2.5), and then expressing human well-being in monetary units, in proportion to the value of potential human productivity (Chapter 6.2.6), thus allowing aggregation of all three endpoint indicators in a single impact category "human production and consumption efficiency" (Chapter 6.2.7), measured in the monetary unit EUR_{2003}. Thus, through this type of approach, the endpoint impact assessment method can also be a way to determine the economic externalities of the waste treatment scenarios.

6.2.2 Impacts on ecosystems

Table 19 lists the factors for converting ecosystem impact indicators from a midpoint to an endpoint indicator. Essentially, these conversion factors can be considered as weighting factors based on available scientific models and quantitative knowledge. Normalisation references are also provided, although these are not applied in this report.

For most midpoint impact categories, the modelling from midpoint indicator results to ecosystem impacts in PDF*m^2*years is documented in the same sources as mentioned for the midpoint impact categories in Chapter 6.1. For the midpoint impact categories derived from EDIP2003, the endpoint characterisation models (damage models) are described in Annex III. §4.

Table 19 Damage (endpoint) characterisation factors and normalisation references for impacts on ecosystems.

	Unit of characterized values	PDF*m2*year/ characterised unit	PDF*m2*year / person-year (normalisation references)	Source
Acidification	M^2 UES	5.47E-02	120	[1]
Ecotoxicity, aquatic	kg-eq. TEG water	5.01E-05	35	[2]
Ecotoxicity, terrestrial	kg-eq. TEG soil	7.40E-04	53	[3]

	Unit of characterized values	PDF*m2*year/ characterised unit	PDF*m2*year / person-year (normalisation references)	Source
Eutrophication, aquatic	kg NO_3-eq.	n.a.	n.a.	[4]
Eutrophication, terrestrial	M^2 UES	8.85E-02	186	[1]
Climate change	kg CO_2-eq.	0.582	6180	[5]
Nature occupation	m^2 arable land	1.68	4900	[6]
Photochemical ozone - Vegetation	m^2*ppm*hours	6.59E-04	93	[1]

[1] See Annex III. §4.
[2] Accompanying spreadsheet to Annex II. The values are not very different from those in Humbert et al. (2005). The difference is due to the improved spatial modelling.
[3] Accompanying spreadsheet to Annex II., but reduced with a factor 10; see the text for justification.
[4] As discussed in Annex III. §4.2, an adequate damage model for aquatic eutrophication is not available. For the current project, this is not important, since the contributing emissions from waste treatment are small compared to the similar emissions from agriculture in Poland, and for the Malta scenario it was assumed that the Mediterranean is an oligotrophic sea, where eutrophication is not immediately an important issue. However, if the method is to be applied for agricultural products, it will be important to supplement it with a damage model for aquatic eutrophication.
[5] See the text for details.
[6] See Chapter 6.1.10.

For ecotoxicity, the damage modelling is documented in Humbert et al. (2005). The slightly different values used here, are primarily due to the improved spatial modelling described in Annex II.

The impacts of climate change are estimated as the consequences of a 2.5 K temperature increase corresponding to a central estimate (IPCC 2001, Watson et al. 2001) for a doubling of the CO_2 concentration in the atmosphere, equal to a global concentration increase of 370 ppm by volume or an emission of 8E14 kg C or 2.93E15 kg CO_2. For mid-range climate scenarios, and assuming perfect dispersal, Thomas et al. (2004b) calculate that 4-13% of all species will lose 100% of their climatically suitable areas by year 2050, and 9-32% will lose over 90% of their climatically suitable areas, with 4 and 14% of all species as the central estimates. A loss of 90% of the climatically suitable area is estimated to give a 44% chance of extinction (Thomas et al. 2004b). As to the indicator is not in terms of species extinction, but rather lost species-area (which may eventually lead to extinction), 4% + 0.9*(14%-4%) = 13% of the global species-area as the central estimate is adopted. With a global terrestrial area of 1.3E14 m^2, this corresponds to a lost area of 1.7E13 m^2. Inclusion of species losing over 50% of their climatically suitable area would correspond to 27.5% of the global species-area or 3.6E13 m^2, based on the 47% of species affected according to Thomas et al. (2004b).

Although relaxation from the climate effect (understood as a return to the previous climate vegetation) is less likely to occur than relaxation from deforestation, applied were the same assumptions on relaxation from climate change as applied for relaxation from deforestation, i.e. 500 years relaxation time, and an average severity during relaxation of 0.2 (see Chapter 6.1.10). This resulted with a characterisation factor of 0.582 PDF*m^2*years/kg CO_2-equivalents (0.2 * 500 years * 1.7E13 m^2 / 2.93E15 kg CO_2-equivalents).

Although it is generally advocated to use best estimates for calculation of characterisation factors (rather than low or high estimates), the estimate made here for climate change is a rather low estimate, since a number of modest assumptions are made (perfect dispersal, only including species losing >90% of their climatically suitable area, full relaxation). However, even with this low estimate, the impacts from climate change will dominate the assessments, see Chapter 9.

There is a reduction factor of 10 on the endpoint characterisation factors for terrestrial ecotoxicity derived here using the IMPACT2002 spatial model. This decision was made after an analysis of the size of the resulting European normalisation reference. While the normalisation reference for terrestrial ecotoxicity in IMPACT2002+ (Jolliet et al. 2003, Humbert et al. 2005) – after removal of emissions to soil cf. Chapter 6.1.4 – is 8E9 PDF*m^2*years, corresponding to 0.2% of the European terrestrial area (4E12 m^2), the same normalisation reference becomes 2.27E11 PDF*m^2*years or 5.7% of the European terrestrial area, when applying the spatial model. This is mainly caused by an increase in the characterisation factors for metal emissions. This may be rather a worst-case estimate than a best estimate that terrestrial ecotoxicity should be responsible for damages equal to more than 5% of the total ecosystem area.

The conversion to absolute impacts in terms of PDF*m^2*years relies on a few, very uncertain assumptions, such as the conversion from PAF*m^3 (Potentially Affected Fraction of species) to PDF*m^2 using the formulae 1 PDF*m^2 = ½·PAF*m^3 / h, where h is the mean depth of root-soil (0.3 m). Since terrestrial ecotoxicity is at the same time one of the impact categories most affected by inventory uncertainty (see Chapter 9.5), the conclusion is that one should be very cautious about basing very far-reaching conclusions on the current impact assessment model for terrestrial ecotoxicity. However, rather than completely leaving out this impact category, its overall contribution to the normalisation reference for ecosystem impacts is interpreted as a worst-case, and instead apply a 10 times lower value as a new best estimate (noting that Jolliet et al. 2003 and Humbert et al. 2005 estimate a two orders of magnitude uncertainty on the models for ecotoxicity). The reduced normalisation reference at the endpoint level thus becomes 0.57% of the European ecosystem area, and thus at the level of the results from the original IMPACT2002 model.

6.2.3 Impacts on human well-being

Table 20 lists the damage characterisation factors for human well-being. Normalisation references are also provided, although these are not applied in this report.

Table 20 Damage (endpoint) characterisation factors and normalisation references for impacts on human well-being.

	Unit of characterized values	QALY / characterised unit	QALY / person-year (normalisation references)	Source
Climate change	kg CO_2-eq.	2.11E-08	2.24E-04	[1]
Human toxicity	kg C_2H_3Cl-eq.	5.35E-06	1.78E-03	[2]
Injuries, road or work	fatal injuries-eq.	43	6.09E-03	[3]
Ionizing radiation	Bq C-14-eq.	2.10E-10	1.12E-04	[4]
Ozone layer depletion	kg CFC-11-eq.	1.05E-03	2.14E-04	[4]
Respiratory inorganics	kg PM2.5-eq.	7.00E-04	6.16E-03	[4]
Respiratory organics	person*ppm*hours	2.64E-06	2.64E-05	[5]

[1] See the text for details.
[2] Accompanying spreadsheet to Annex II. The values are larger than those in Humbert et al. (2005) due to the improved spatial modelling.
[3] Mathers et al. (2004)
[4] Humbert et al. (2005)
[5] See Annex III. §3.

The impacts of climate change are again estimated as the consequences of a 2.5 K temperature increase corresponding to a central estimate for a doubling of the CO_2 concentration in the atmosphere, equal to a global concentration increase of 370 ppm by volume or an emission of 800 Gt C or 2.93E15 kg CO_2. The uncertainty range on the temperature increase at CO_2 doubling (known as the climate sensitivity) is 1.5-4.5 K and the temperature response to increasing CO_2 concentration is logarithmic (IPCC 2001, Watson et al. 2001).

Using the same QALYs/case for the different diseases as in Ecoindicator99 (Goedkoop & Spriensma 2001), the impacts on human well-being are 2.1E-8 QALY/kg CO_2-equivalent, caused by 4.8E5 additional cases of vector-borne diseases at 50 QALY/case, 8.8E6 QALYs as a net change in heat and cold related diseases, 4.8E6 relocations due to sea-level rise at 1 QALY per case, and 2.4E7 QALYs as the impact from additional diarrhoea. These incidence values are rough estimates based on interpretation of Tol (2002). This interpretation has not yet been verified by Tol, so caution should be taken in using these values in other contexts.

A comparison shows that value resulting here is an order of magnitude lower to that of Ecoindicator99 (2.1E-7 DALY / kg CO_2). Regarding interpretation, the difference is likely to be caused by the number of cases of malaria, since this dominates the Ecoindicator99 value, which in addition does not include negative damage (i.e. benefit), except when it compensates positive damage within the same region (Goedkoop & Spriensma 2001).

Although the difference between the estimates may seem large, it should be noted that the importance of health impacts is only a small part of the total impact from climate change (0.8% with the estimate here and 7% with that of Ecoindicator99), since the overall impact is dominated by the impact on nature; see Chapter 6.2.5. This means that the even if the health impacts may be underestimated with this interpretation, this would only have a small influence on the overall assessment of the importance of climate change.

For all other midpoint impact categories, the modelling from midpoint indicator results to human well-being impacts in QALY is documented in the same sources as mentioned for the midpoint impact categories in Chapter 6.1, noting that 1 QALY = -1 DALY. For the midpoint impact categories derived from IMPACT2002+, the endpoint characterisation models (damage models) are described in Humbert et al. (2005). The improved spatial modelling described in Annex II. results in higher characterisation factors for human toxicity than in Humbert et al. (2005). For respiratory organics, the damage modelling is described in Annex III. §3.

6.2.4 Impacts on economic production

All the damage characterisation factors for impacts on economic production are provided (see Table 21). Normalisation references are also provided, although these are not applied in this project.

Life Cycle Assessment has traditionally ignored impacts on economic production, with the exception of impacts of resource dissipation. In contrast, impacts on economic production have for many years been in focus of cost-benefit analyses. However, analysing the different estimates provided in the RED database (www.red-externalities.net), it is only human health impacts and the impacts on agricultural production from climate change and photochemical ozone that are of a size that may influence the assessment here. This study is therefore limited to providing characterisation factors for these impacts and – following the tradition in Life Cycle Impact Assessment – the impact of current resource dissipation.

In addition to the direct impact on human well-being recorded in Table 20 (Chapter 6.2.3), the direct health impacts listed there also impact indirectly on economic production in terms of lost labour and/or treatment costs. For each of the midpoint impact categories it would be possible to model this impact on economic production specifically (see e.g. Miller et al. 1998), taking into account the severity and treatment costs for the involved disabilities and taking into account only life-years lost in the productive age. Since such detailed modelling is beyond the scope of this study, this report adopts the general observation that there is a fairly good correlation between QALY values and economic production losses in percentage of GDP per capita when applying the same discounting rates for both (Miller et al. 2000). As a general proxy, the loss of economic production from a health impact of 1 QALY in Europe is estimated here to 23,000 EUR_{2003}, which is the 2003 GDP per capita for EU25.

Climate change arguably has both positive as well as negative influences on agricultural yields. Tol (2002) summarized the available global studies for impacts until year 2200 and for a central 2.5 degrees temperature increase, which are interpretable as a net impact of approx 2.5E12 EUR_{2003} or 8.5E-4 EUR_{2003}/kg CO_2.

The midpoint indicator “Mineral extraction” (Chapter 6.1.9) measures the difference between the current energy requirement for extraction and an estimated future energy requirement for extraction from lower grade ores. As alternative energy sources to fossil fuels are currently becoming competitive, there is no reason to assume that long-term energy prices will exceed the current energy prices for fossil fuels. Hence, a damage (endpoint) characterisation factor of 0.004 EUR_{2003} / MJ extra is adopted, based on current energy prices, without discounting of future costs. The total impact is 1.2 EUR_{2003} / person-year, using the normalisation reference from Table 18.

Assuming that the future energy system will be based on renewable energy sources, the current dissipation of non-renewable energy carriers, rendering them unavailable for other purposes, will not have any influence on the future energy requirement for the provision of energy. Thus, the damage (endpoint) characterisation factor for the midpoint category “Non-renewable energy” (see Chapter 6.1.11) is 0 EUR_{2003} / MJ primary, i.e. zero impact on economic production.

For impacts from photochemical ozone on agricultural crop production assumed was the rough estimate from Annex III. §4.3 of a 10% reduction in crop yields caused by the current emission levels in Europe, which was then applied to the annual crop production value of 1.7E11 EUR_{2003}.

Table 21 Damage (endpoint) characterisation factors and normalisation references for impacts on economic production.

	Unit of characterized values	EUR_{2003} / characterised unit	EUR_{2003} / person-year (normalisation references)	Source
Climate change	kg CO_2-eq.	-3.65E-04	-3.9	[1]
Human toxicity	kg C_2H_3Cl-eq.	1.22E-01	41.0	[2]
Injuries, road or work	fatal injuries-eq.	9.89E+05	140.0	[2]
Ionizing radiation	Bq C-14-eq.	4.83E-06	2.6	[2]
Mineral extraction	MJ extra	4.00E-03	1.2	[3]
Non-renewable energy	MJ primary	0	0.0	[3]
Ozone layer depletion	kg CFC-11-eq.	24	4.9	[2]
Photochemical ozone - Vegetation	m^2*ppm*hours	2.80E-04	39.0	[4]
Respiratory inorganics	kg PM2.5-eq.	16.1	142.0	[2]
Respiratory organics	person*ppm*hours	6.07E-02	0.6	[2]

[1] The negative damage (i.e. benefit) to economic production is the net effect of the health impact on economic production calculated as in note [2] and a net increase in agricultural production of 8.5E-4 EUR_{2003} / kg CO_2.
[2] The QALY values recorded in Table 20 multiplied by 23000 EUR_{2003}.
[3] See the text for explanation.
[4] By applying the rough estimate of a 10% reduction in crop yields to the annual European crop production value of 1.7E11 EUR_{2003} (Annex III.), obtained is a total impact on crop production of 1.7E10 EUR_{2003} / year or 39EUR / person-year. With the normalisation values (Table 18), this gives a damage characterisation factor of 2.8E-4 EUR_{2003} / m^2*ppm*hour.

6.2.5 Expressing ecosystem impacts in terms of human well-being

When it is acceptable to apply choice modelling to derive midpoint characterisation factors (see Chapter 6.1.6), it should also be acceptable to apply choice modelling to express the severity of ecosystem impacts in terms of QALYs. For example, it could be investigated what sacrifice in terms of disabilities or lost life years would be acceptable to protect a certain ecosystem area, or put in other terms: what reduction in life quality is regarded as equivalent to the loss of a certain ecosystem area.

Although choice modelling studies have been performed for specific ecosystems and geographically limited ecosystem services (see the survey of Hanley et al. 2001 for examples), no studies have yet been made at the level of abstraction that allows to relate PDF*m^2*years of ecosystem to impacts on human well-being in QALYs.

Since it is beyond the scope of this project to perform the necessary choice modelling experiments, a proxy value is applied from the protection target expressed in the Convention on Biological Diversity. This convention calls for a protection of 10% of the global ecosystems, which can be based on the global terrestrial[13] species-area of 1.31E14 m^2*years. Comparing this protection target relative to a protection target of 100% for human well-being for the global population (6.225E9 people at perfect well-being = 6.225E9 QALY), gives 0.1 * 6.225E9 / 1.31E14 = 4.75E-6 QALY/m^2*years.

The inverse of 4.75E-6 QALY/m^2*years is 2.1E5 m^2*years or 21 ha*years/QALY. This is supposed to mean that the full protection of an ecosystem of 21 ha (210,000 m^2) for one year has the same value as an extra life-year for one person.

It may be argued that 10% is a modest protection target for ecosystems, but even with this low value, ecosystem impacts will tend to dominate the overall results of the environmental assessments (see Chapter 9). The reason for this is that human impacts currently engage approximately 50% of natural ecosystems, which translates into a loss of 5% of all potential QALYs (0.05 QALY/person-year), while all the impacts on human well-being included in this study only sum to 0.012 QALY/person-year or a loss of 1.2% of all potential QALYs (4^{th} column in Table 20).

Interpreting the 5% loss in QALYs in terms of lost income (see Chapter 6.2.6), it is also unlikely that the current willingness to pay for protection of natural ecosystems exceeds this value. Comparing to current environmental protection expenditures in developed countries (1-2% of GDP) confirms that this proxy value is in a reasonable range.

6.2.6 The monetary value of a QALY

The monetary value of a QALY has an upper limit defined by the budget constraint. Since a QALY by definition is a life-year lived at full well-being, the budget constraint can be determined as the potential annual economic production per capita.

The potential average annual economic production per capita is calculated by Weidema (2005) at a value equivalent to 74,000 EUR_{2003}. This is calculated by taking the current Gross Economic Product (GEP)[14] of USA (39,500 EUR_{2003}) as a starting point – noting that USA has the highest GEP in the World, when ignoring a few untypical economies based heavily on oil or banking – and multiplying it by the factor 1.87 derived in Table 22. Besides the difference in employment, health, trade barriers and education, the current difference between the USA and the global average is

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[13] The endpoint characterisation factor for ecosystems therefore has a bias towards terrestrial impacts, since the current midpoint impact assessment methods (EDIP2003 and IMPACT2002+) do not provide adequate characterisation models for impacts on marine ecosystems. Contributing to this unfortunate "blind spot" is the lack of a damage model for aquatic eutrophication, the focus on freshwater species in the aquatic ecotoxicity category, and the lack of characterisation factors for impacts on sea bottom in the category nature occupation.

[14] GEP is defined by Ironmonger (1994) as the sum of the Gross Domestic Product (GDP) and the Gross Household Production (GHP). The current GHP can be estimated at about 0.5 of the current GDP.

assumed to be due to lacking physical and social infrastructure. There are no other apparent reasons for the GEP of countries to differ.

The resulting value of 74,000 EUR_{2003} / QALY derived from the budget constraint can be compared to the values derived by other monetarisation methods. Hirth et al. (2000) determined the value of a QALY as implied from 42 estimates in the value-of-life literature and found the results to be strongly dependent on the method for soliciting the values. They found median values of 25,000 US $dollars_{1997}$ (approx. 23,000 EUR_{2003}) per QALY for studies using the human capital approach, and 160,000 US $dollars_{1997}$ (approx. 150,000 EUR_{2003}) per QALY for contingent valuation studies, when using a 3% discounting rate (corresponding to 90,000 EUR_{2003} / QALY without discounting). The median values were 93,000 US $dollars_{1997}$ for studies using revealed preferences for non-occupational safety equipment and 428,000 for job-risk studies, both calculated for a 3% discount rate.

Table 22 **Ideal economic production relative to the current economic production of the USA.**

Economic category	Ideal relation	Estimated range	Basis of calculation
Unemployment and underemployment	1.02	1.01 – 1.03	[1]
Health and other work-disabling impacts	1.19	1.16 – 1.22	[2]
Effect of trade barriers	1.05	1.01 – 1.08	[3]
Education	1.46	1.33 – 1.56	[4]
Product of all the above	1.87	*1.57 – 2.12*	

[1] The ideal workforce of 0.485 per capita (97% of a labour force participation of 0.5 at 3% unavoidable frictional and structural unemployment) expressed relative to the current workforce of 0.46 per capita (94.2% of a labour force participation of 0.488 at 5.8% unemployment). Only 30% of the difference between the ideal and the current situation is included, due to the offsetting impact on household production.
[2] A situation of full health expressed relative to the current health gap of approx. 16% (Mathers et al. 2004).
[3] Ideal without trade barriers expressed relative to the current situation, which involves a loss of 5 times the 1% of developed world GDP lost due to trade barriers on goods according to Newfarmer (2001).
[4] Ideal average 18 years of schooling, involving a 6.8% increase in GDP per year of additional schooling between 12 years and 18 years, relative to the current US adults' average 12.2 years (Barro & Lee 2000), i.e. 1.068E(18-12.2).

The human capital approach values only the earning ability, i.e. comparable to the lost economic production impacts assessed in Chapter 6.2.4. It is therefore expected that these values are lower than the values derived from the potential economic production, which takes into account the full earning ability when current barriers for full economic production are removed. The higher values of the contingent valuation studies can be explained by the difficulties to adequately account for the budget constraint in this type of studies. Also, studies based on contingent valuation and revealed preferences most often assess voluntary risk or risk aversion behaviour, and the derived values can best be interpreted as the individuals' evaluation of impacts that occur to themselves, rather than a value that is applicable for general policy purposes, see also the discussion in Markandya et al. (2004).

The global nature of the QALY concept, i.e. that a QALY has the same value for all individuals, supports that the value of a QALY should be derived from the global average budget constraint, rather than the budget constraints and valuations of specific individuals. It is interesting to note that the recent willingness-to-pay studies performed as part of the recent update of the ExternE methodology (Markandya et al. 2004) result in a recommended undiscounted value of a life year of 74,627 Euro, i.e. practically the same adopted here. While this is purely a coincidence, it confirms that the value here is in a reasonable range. The ExternE update is characterised by specifically seeking to address small risk increases from involuntary exposure and is therefore regarded as more relevant for policy analysis of pollution impacts than previous studies.

6.2.7 Aggregating all impacts into one single damage category

A final damage (endpoint) impact category, termed here the "Human production and consumption efficiency", is given here in $EUR_{2003.}$ This impact category combines the impacts on human production efficiency losses (i.e. the impacts on humans, ecosystems and resources that reduce the production output) with the impacts on the so-called human consumption efficiency (i.e. the impacts on human well-being and ecosystems that reduce human ability to fully enjoy a given production output). Table 23 presents aggregation of the impacts on ecosystems, human well-being, and economic production, using the conversion factors of 4.8E-6 QALY / PDF*m^2*years and 74,000 EUR_{2003} / QALY developed in Chapters 6.2.5 and 6.2.6, respectively.

Table 23 Summary of the damage (endpoint) characterisation factors developed in Chapters 6.2.1 to 6.2.6, and aggregation of all impacts into the single-score indicator "Human production and consumption efficiency" measured in EUR_{2003}.

Impact category	Unit of characterised values at midpoint	Impact on ecosystems		Impacts on human well-being		Impacts on production	All impacts aggregated
		PDF*m^2*year /characterised unit at midpoint [1]	EUR2003 / characterised unit at midpoint [2]	QALY / characterised unit at midpoint [3]	EUR2003 / characterised unit at midpoint [4]	EUR2003 / characterised unit at midpoint [5]	EUR2003 / characterised unit at midpoint [6]
Acidification	m^2 UES	5.47E-02	1.92E-02				1.92E-02
Ecotoxicity, aquatic	kg-eq. TEG wat.	5.01E-05	1.76E-05				1.76E-05
Ecotoxicity, terrest.	kg-eq. TEG soil	7.40E-04	2.60E-04				2.60E-04
Eutrophication, aq.	kg NO_3-eq.	n.a.	n.a.				n.a.
Eutrophication, terr	m^2 UES	8.85E-02	3.11E-02				3.11E-02
Climate change	kg CO_2-eq.	0.582	2.05E-01	2.11E-08	1.56E-03	-3.65E-04	2.06E-01
Human toxicity	kg C_2H_3Cl-eq.			5.35E-06	3.96E-01	1.22E-01	5.18E-01
Injuries, road/work	fatal injuries-eq.			43	3182000	9.89E+05	4.17E+06
Ionizing radiation	Bq C-14-eq.			2.10E-10	1.55E-05	4.83E-06	2.04E-05
Mineral extraction	MJ extra					4.00E-03	4.00E-03
Nature occupation	m^2 arable land	1.68	5.91E-01				5.91E-01

Impact category	Unit of characterised values at midpoint	Impact on ecosystems		Impacts on human well-being		Impacts on production	All impacts aggregated
		PDF*m²*year /characterised unit at midpoint [1]	EUR2003 / characterised unit at midpoint [2]	QALY / characterised unit at midpoint [3]	EUR2003 / characterised unit at midpoint [4]	EUR2003 / characterised unit at midpoint [5]	EUR2003 / characterised unit at midpoint [6]
Non-renew. energy	MJ primary						0
Ozone layer deplet.	kg CFC-11-eq.			1.05E-03	7.77E+01	24	1.02E+02
Ph.chem. ozone – veg	m^2*ppm*hours	6.59E-04	2.32E-04			2.77E-04	5.08E-04
Respirat. inorganics	kg PM2.5-eq.			7.00E-04	5.18E+01	16.1	6.79E+01
Respiratory organics	pers*ppm*hours			2.64E-06	1.95E-01	6.07E-02	2.56E-01

[1] From Table 19
[2] Column [1] multiplied by 4.75E-6 QALY / PDF*m^2*years (Chapter 6.2.5), and 74000 EUR_{2003} / QALY (Chapter 6.2.6).
[3] From Table 20
[4] Values from column [3] multiplied by 74000 EUR_{2003} / QALY from Chapter 6.2.6.
[5] From Table 21
[6] Sum of values from column [2], [4] and [5].

As mentioned in Chapter 6.2.6, the relationship between QALYs and monetary units is an equivalence, which means it would also be possible to express all impacts in terms of human well-being measured in QALYs (by multiplying the last column by 1/74,000 = 1.35E-5 QALY/EUR_{2003}). This alternative form of presenting the single-score result has not been applied. Since physical single-score and monetarisation methods were applied to aggregate the results in one impact category ("Human production and consumption efficiency"), neither normalisation nor weighting (in the strict sense of comparing across impact categories) are relevant for this endpoint impact assessment, as explained in Chapter 6.1.17. This assessment method is therefore compliant with ISO 14044 for use in relative comparisons and public disclosures.

6.2.8 Comparison to other monetarisation methods

Earlier monetarisation studies have primarily obtained their values from stated preferences (via contingent valuation or choice modelling) or from revealed preferences. The method applied in this study (i.e. obtaining the monetary values directly via the overall budget constraint in terms of the potential human economic production), requires that all impacts are first expressed in the same physical unit (here QALYs), which has only recently become possible, e.g. as a result of the work in developing the Ecoindicator99 method (Goodkoep & Spriensma 2001).

In general, previous studies combine a number of different methods for monetarisation and solicit separate values for specific pollutants, disabilities and environmental compartments. For example, the ExternE study (Bickel & Friedrich 2005) applies damage values for impacts on health, agriculture and buildings, but resort to preferences revealed in political negotiations for impacts on ecosystems, and a mixed approach for climate change impacts. Furthermore, morbidity and mortality is valued separately, combining different monetarisation studies for different diseases and health endpoints. The more separate studies are combined, the larger the risk of inconsistencies.

An overview of monetisation studies with relevance for waste management has recently been provided by Turner et al. (2004). Table 24 shows the values of this study compared to the values in the summary table of Turner et al. (2004) translated to Euro, using the exchange rate of 1.45 Euro/GBP.

Important impacts are left un-monetarised in previous studies. For example, most studies do not provide consistent damage values for ecosystem impacts. This is especially problematic for climate change, where the ecosystem impact is dominating the potential impacts, but also the important impact from land use is left un-quantified in most studies. The ExternE study does not apply damage values for impacts on ecosystems, but – as also done in this study – resorted to what they call a "second-best" method of revealed preferences from political negotiations. For acidification and eutrophication they derive a range of 63-350 Euro per ha of ecosystem protected. This may be compared to the 3,500 Euro / ha estimate derived here from the 21 ha / QALY relationship in Chapter 6.2.5.

An important cause of the uncertainty found in willingness-to-pay studies, reflected in the wide ranges shown in Table 24, is that the results vary with the geographical location, population and context. While this may indeed provide relevant values for a specific context, it is less useful for deriving values for an abstract concept like QALYs, which is intended to be globally applicable for aggregation of impacts in many different contexts. Using QALYs as a physical single-score has the advantage that it allows to apply a strict formulation of the overall budget constraint, reducing the uncertainty on the monetary value of a QALY (range 62,000-84,000 EUR_{2003} per QALY versus the 27,000-225,000 Euro of the ExternE project (Markandya et al. 2004); see also Chapter 6.3.

Table 24 Comparison of the values of this study to the summary values in Turner et al. (2004). All values in $Euro_{2003}$ per Mg emission.

Substance	Previous studies as reviewed by Turner et al.	This study	Comment
CO_2	1 – 55	206	[1]
CO	2	724	[2]
NOx	2,200 – 42,000	10,700	[3]
PM2.5	2,900 – 435,000	68,200	[4]
PM10	2,600 – 330,000	36,500	[4]
SO_2	2,500 – 23,000	5,680	[5]
VOC	725 – 2,200	330	[6]

[1] 99% of the value is ecosystem impact, while the previous studies have generally not quantified the ecosystem impact. Thus, the value of previous studies mainly captures health and economic production impacts.
[2] The value of 724 Euro is composed of health impacts (70 Euro), agricultural impact (169 Euro), ecosystem impact (141 Euro), climate change impact (324 Euro), and human economic production impacts (20 Euro). The 2 Euro value of previous studies is probably due to insufficient physical modelling rather than differences in monetarisation.
[3] The value of 10700 Euro is composed of health impacts (6600 Euro), human economic production impacts (2100 Euro), ecosystem impacts (1520 Euro), and agricultural impact via photochemical ozone (443 Euro). The values of previous studies are dominated by the health impact, but also include small contributions from fertilization effect (a benefit of 200 Euro) and effects on buildings (300 Euro), both of which were ignored in this study, due to their relatively low importance.
[4] The PM values are for health impacts, except for a small contribution of 200 Euro / Mg PM10 for impacts on buildings, which we have ignored in this study, due to the low importance
[5] The value of 5680 Euro is composed of health impacts (4060 Euro), human economic production impacts (1260 Euro), and ecosystem impact (360 Euro). The values from previous studies are also dominated by the health impact, with 370-962 Euro impacts on buildings, 14 Euro impact on agriculture, and 8 Euro impact on ecosystems.

[6] The value of 330 Euro is composed of health impacts (20 Euro) incl. human economic production impacts, agricultural impact (169 Euro) and ecosystem impacts (140 Euro), while the previous studies have generally not quantified the ecosystem impact. Turner et al. (2004) also give recommended values for the UK based on a study by Watkiss et al. (2004), where the values for health impacts are 4-600 Euro and the value for agricultural impact is 380 Euro. These more recent values are thus closer to estimates of this project.

6.3 Uncertainty in the impact assessment methods

Some information on the uncertainties of the characterisation factors is generally available in the methods supplying the characterisation factors, i.e. EDIP2003 (Hauschild & Potting 2005, Potting & Hauschild 2005) for acidification, eutrophication and photochemical ozone formation (uncertainties also provided in Annex III.), and IMPACT2002+ for human toxicity and ecotoxicity (where e.g. Jolliet et al. 2003 and Humbert et al. 2005 suggested a rough value of a factor 100). For the remaining impact categories taken from IMPACT2002+ (ionising radiation, mineral extraction, non-renewable energy, ozone layer depletion and respiratory inorganics) as well as for the endpoint characterisation factors for the EDIP2003 impact categories, the characterisation models are taken directly from Ecoindicator99 for which the uncertainties are estimated in Goedkoop & Spriensma (2001). It should be noted that the documentation and quality related to uncertainty information varies depending on the impact category and the extent of modelling.

For the midpoint climate change characterisation factors (kg CO_2-equivalents / kg substance), the IPCC suggests an uncertainty of 30% for substances other than CO_2. For the endpoint characterisation factors for climate change, the uncertainties are large, as indicated in Tol (2002) and Thomas et al. (2004b). The uncertainty on the temperature effect of CO_2-doubling is 1.5-4.5 K around the central estimate of 2.5 K and the temperature response to increasing CO_2 concentration is logarithmic (IPCC 2001, Watson et al. 2001). As mentioned in Chapter 6.2.2, the rather low estimate for the dominating ecosystem effects here implies that the effects corresponding to 1.5 K should be seen as a lower bound, while the upper bound will be well beyond the effects corresponding to a “linear” interpretation of the 4.5 K estimate.

For injuries, the uncertainty on the characterisation factors is low, as long as they are applied to the same data sources from which they are derived, i.e. the Eurostat data on work related accidents and the CARE Road Accident Database, and at the same level of aggregation (i.e. the level of industries). When the inventory data are from other sources with different injury definitions, it may be necessary to develop specific characterisation factors suitable for these sources. When applied for specific processes or injuries, the deviation from an average “non-fatal injury at work” or average “non-fatal road injury” may be large, and has to be determined in each individual situation.

For nature occupation, the uncertainty for the impact category “Land use” of Goedkoop & Spriensma (2001) can be applied as a basic uncertainty. For occupation of arable land (all land with potential for agriculture), an additional severity of 0.88 was included to represent the secondary impacts from current deforestation. For this additional severity, the most critical assumption is the relaxation time. According to Dobben et al. (1998), the relaxation time to reach full potential biomass production varies from 50 to 220 years depending on latitude and altitude. Weidema & Lindijer (2001) suggested that the relaxation times for biodiversity are a factor 6 higher, i.e. 300 to 1300 years. This may be taken as a rough estimate of the uncertainty around the applied central estimate here of 500 years relaxation time.

As the geographical uncertainty can be an important factor in the uncertainty of the characterisation for site-generic emissions, the uncertainty of the site-dependent characterisation factors are assumed to be lower than for the corresponding site-generic characterisation factors. The uncertainty is especially reduced for emissions that do not disperse very far and come from very specific locations. However, this potential for reduction in uncertainty for site-specific emissions has not been quantified.

The factor for converting ecosystem impacts in PDF*m^2*years to human well-being impacts in QALYs is based on a very rough estimate using a political target (that the protection target of the Convention on Biological Diversity can be taken as a proxy for the preferences expressed in a properly conducted choice modelling experiment). It was estimated that the applied conversion factor (4.75E-6 QALY / PDF*m^2*years) may vary between 2.4E-6 and 1.2E-5 QALY / PDF*m^2*years (84,000-420,000 PDF*m^2*years /QALY), corresponding to protection targets of 4 to 20% of all ecosystems, or to valuing the total current impact on ecosystems at 2-10% of all potential QALYs (0.02 – 0.1 QALY / person-year).

Uncertainty on the factor for converting QALYs to EUR_{2003} is 62000 – 84000 EUR_{2003} per QALY around the central estimate of 74000 EUR_{2003}, determined by using low and high estimates for each of the component factors in Table 6.2.4. The corresponding willingness-to-pay estimate of the ExternE project of 74,627 Euro is provided with an uncertainty estimate of 27,000-225,000 Euro (Markandya et al. 2004).

7 Life cycle costing

Life cycle costing is the assessment of all costs associated with the life cycle of a product that are directly covered by any one or more of the actors in the product life cycle, i.e. not including externalities, except when anticipated to be internalised in the decision-relevant future.

If the costs of each process or material input in the analysed system are aggregated, all intermediate costs are eliminated (since an intermediate cost to one actor is an income to another actor), and the remaining aggregated cost of the system, i.e. the life cycle cost, is therefore equal to the total net value added over the life cycle, i.e. the wages, taxes and net operating surplus (profits and rent) for all processes or material inputs in the analysed system.

In standard LCA databases, value added is not recorded for each process, so most often the prices of materials excluding indirect taxes were applied. For material and energy inputs in general, world prices for 2003 are used. For the specific waste collection and treatment processes, cost data are taken from the AWAST study (Bozec 2004). These cost data are reported in Chapter 5 for each specific technology.

The value of recycled materials is highly volatile due to the fluctuations that are inherent to markets with constrained supplies. It is therefore difficult to obtain reliable, comparable market data for recycled materials. For example, Hogg report (2001) values between 25 and 440 Euro per Mg for PE, and between 216 and 1115 for aluminium. Both the low and high values are likely to stem from temporary market constraints, and do not reflect the more stable situation of a developed market with a higher degree of recycling than today. Therefore one source of data – that provide comparable prices for all the relevant waste fractions over more than a decade – was used (www.letsrecycle.com). This source, which is for the UK, where the recycling markets are relatively well developed, was verified against the data of Hogg (2001) for the UK, and data from USA (Plunkert 2004, Fenton 2004, SRM 2005), which provide values in the same range.

Table 25 Economic value of recycled material.

Material	Economic value of recycled material [$Euro_{2003}$/Mg]	
	Range	Average
Aluminium	580-970	780
PE	180-220	200
PET	110-230	170
Cardboard	45-100	73

Material	Economic value of recycled material [$Euro_{2003}$/Mg]	
	Range	Average
Newsprint	50-73	62
Iron and steel	36-55	45
Mixed paper	30-50	40
Glass	8-25	17

Based on www.letsrecycle.com, using a conversion rate of 1.45 Euro/GBP.

Obviously, there is a limit to how much material can be captured in separate fractions. Not all households will participate in source separation, and some materials will be too soiled to separate or used for wrapping other wastes. Tucker & Speirs (2002) calculate maximum recovery rates between 80% and 85%, the higher applicable to cardboard, newspaper and glass, the lower applicable to other fractions for which a separate collection is encouraged. Such high recycling rates demand high levels of promotion. The costs of this promotion should therefore be included when calculating the overall costs of recycling. Hogg (2001) quotes on-going annual promotion costs of up to 7 Euro per household, approximately equal to 7 Euro per Mg waste, while normal promotional costs are at 2 Euro per Mg waste.

For waste collection and landfilling, a large part of the costs is related to the amount of waste, i.e. with a constant cost per Mg waste, irrespective of the type of waste; see Table 26. However, for recycling, and to some extent also for incineration, there is a large difference in the value of the recycled materials and energy recovered, respectively, which means that this becomes the decisive parameter for which of the waste fractions that are most economical to recycle, see Table 27.

Table 26 Total costs of waste collection and treatment, divided on costs that are fixed and variable.

	Fixed costs [Euro/Mg waste]	Additional cost that depend on waste type [Euro/Mg waste]	Comments
Collection	62 – 100	0 – 200	[1]
Uncontrolled landfilling	19 – 45	0	
Directive compliant landfilling	38 – 82	-2 – 1	[2]
Incineration	27 – 34	-46 – 28	
Material recycling	0	-670 – 8	

[1] Highest costs may be applicable to lightweight fractions such as PET bottles and aluminium cans. However, these fractions also have the highest value in material recycling. The resulting net costs for recycling these fractions are negative, i.e. recycling is favourable from a purely economical viewpoint; see Table 27.

[2] Gas treatment depends on biodegradable fractions; range depends on whether the gas is used for electricity or not.

Table 27 Calculation of the net costs of incineration and recycling.

Waste fraction	Incineration			Recycling		Net costs without fixed costs of collection	
	Fixed cost of incineration [Euro/Mg]	Assumed lower heating value [GJ/Mg]	Economic value of electricity sold [Euro/Mg] [1]	Additional cost of collection of separate fractions [Euro/Mg] [2]	Economic value of recycled material [Euro/Mg]	Net costs of incineration [Euro/Mg]	Net costs of recycling [Euro/Mg]
Aluminium	40 – 45	4.62	19	23 – 35	580 – 970	21 – 26	-947 – -545
PE	40 – 45	42.47	230	23 – 35	180 – 220	-190 – -185	-197 – -145
PET	40 – 45	22.95	121	23 – 35	110 – 230	-81 – -76	-207 – -75
Cardboard	40 – 45	15.92	82	23 – 35	45 – 100	-42 – -37	-50 – -15
Newsprint	40 – 45	14.20	72	23 – 35	50 – 73	-32 – -27	-47 – -29
Iron & steel	40 – 45	2.00	4 – 5	23 – 35	36 – 55	35 – 41	-32 – -1
Mixed paper	40 – 45	14.12	72	23 – 35	30 – 50	-32 – -27	-27 – 5
Glass	40 – 45	-0.18	-8	23 – 35	8 – 25	48 – 53	-2 – 27
Wet biowastes	40 – 45	4.00	16	5 – 35	-32 – 1 [3]	24 – 29	4 – 67

[1] Only electricity; 0.08Euro/kWh * (heating value * 25% efficiency * 278kWh/GJ - 80-85kWh/Mg internally used).
[2] These costs are the general costs from Kranert et al. (2004) (see Chapter 5.3). Hogg (2001) suggests that lightweight materials (aluminium and plastics) may have additional costs up to 200 Euro, which are not included here, as this is ascribed to un-optimised situations with inadequate compression before transport. An additional 5 Euro is added for promotion for maximum recycling percentages. In the scenarios for Malta, additional sea transport of 23-35 Euro/Mg must be added (Hogg 2001, Annex 10.4).
[3] Net value of sale of electricity and compost after deduction of additional treatment costs, cf. Chapter 5.3.3.

For plastics, cardboard, newsprint, and wet biodegradable wastes, there is an overlap in the ranges of net costs of incineration and recycling in Table 27.

When developing scenarios in Chapter 8, the midpoint of the ranges are applied, which suggest that material recycling is very often preferable, due to the revenue from sale of materials. The exceptions to this general rule appears to be:

- Materials with a high heating value, such as polyethylene (PE), where incineration may be the least cost option, see also Table 28. However, it may be argued that separate incineration of a PE fraction would require separate collection (or later separation), which would partly offset this advantage over recycling. However, it is assumed here that PE follows the residual waste fraction, and is not burdened with separate collection costs, which means that incineration of the PE fraction is still the most economical option, and this is applied in the incineration scenarios. Here, PE represents also other plastics, which may have the same characteristics, such as PS, while the fraction "other plastics" is assumed to be composed of the plastics for which recycling is the most economical option. Also for the fraction "other paper", the relatively high heating value appears to make incineration the most competitive option, when comparing to the average net cost of recycling, see Table 28.
- Situations where specific geographical conditions, such as the geographic isolation of Malta, require that recycled materials be transported longer distances. This affects cardboard and newsprint, where the economic advantage of recycling over incineration is low, but still positive in Krakow, while an assumed additional 23 Euro/Mg for transporting the cardboard and newsprint to the continent from Malta is enough to make incineration economically preferable, see Table 28. However, this change in economic preferences has no consequence for the scenarios, since the recycling targets provide a limit to the amount of paper wastes that can be incinerated.

For wet biodegradable wastes, incineration appears to be the least cost option, see Table 28, due to the relatively high costs of composting. This is in spite of the potentially higher yield of energy from composting than when the biodegradable wastes are incinerated.

Table 28 **Marginal costs (in Euro per Mg waste) of recycling compared to incineration, sorted by lowest marginal cost.**

Waste fraction	Marginal cost of recycling compared to incineration [1]		Marginal cost of recycling compared to incineration, when adding 23-35 Euro additional transport costs for recyclable materials from Malta to the continent	
	Range	Average	Range	Average
Aluminium	-973 – -566	-770	-950 – -530	-740
PET	-131 – 6	-63	-108 – 41	-34
Iron & steel	-73 – -36	-55	-50 – 0	-25
Glass	-55 – -21	-38	-32 – 14	-9
Cardboard	-40 – 32	-4	-17 – 67	25
Newsprint	-23 – 17	-3	0 – 52	26

Waste fraction	Marginal cost of recycling compared to incineration [1]		Marginal cost of recycling compared to incineration, when adding 23-35 Euro additional transport costs for recyclable materials from Malta to the continent	
	Range	Average	Range	Average
Wet biowastes	-25 – 43	9	n.r.	n.r.
PE	-12 – 45	17	11 – 80	46
Mixed paper	0 – 37	19	23 – 72	48

[1] Net costs of recycling minus net costs of incineration from Table 27.

Table 29 Marginal costs (in Euro per Mg waste) of recycling compared to landfilling, sorted by lowest marginal cost.

Waste fraction	Marginal cost of recycling compared to landfilling [1]		Marginal cost of recycling compared to landfilling, when adding 23-35 Euro additional transport costs for recyclable materials from Malta to the continent	
	Range	Average	Range	Average
Aluminium	-1014 – -599	-804	-991 – -564	-775
PE	-264 – -199	-229	-241 – -164	-200
PET	-274 – -129	-199	-251 – -94	-170
Cardboard	-144 – -64	-102	-121 – -29	-73
Newsprint	-117 – -69	-91	-94 – -34	-62
Iron & steel	-99 – -55	-75	-76 – -20	-46
Mixed paper	-94 – -49	-69	-71 – -14	-40
Glass	-69 – -27	-46	-46 – 8	-17
Wet biowastes	-63 – 13	-23	n.r.	n.r.

[1] Net costs of recycling from Table 27, minus net costs of landfilling (54-67 Euro; see Chapter 5.3.1).

8 Waste management infrastructure scenarios and flow diagrams

The technologies described in detail in Chapter 5 are combined in 5 scenarios, which are described in the following sections:

A) Baseline (2003) waste management infrastructure

B) Incineration scenario with increased recycling

C) Composting scenario with increased recycling

D) Economic optimum scenario

E) Societal optimum scenario.

8.1 Baseline (2003) waste management infrastructure (A)

In the baseline scenario for Malta, all municipal solid waste is going to uncontrolled landfill, except for 24% of the wet biodegradable wastes, which go to central composting without energy recovery.

Table 30 Technology combinations in the baseline scenario (A) for Malta.

	Unit	Uncontrolled landfill	Central composting without energy recovery	Sums
Wet biodegradable wastes	Mg	(76%) 73992	(24%) 23410	97402
Paper and cardboard wastes	Mg	19785		19785
Plastic wastes	Mg	13181		13181
Glass wastes	Mg	5179		5179
Iron and steel wastes	Mg	4580		4580
Aluminium wastes	Mg	346		346
Other wastes	Mg	13168		13168

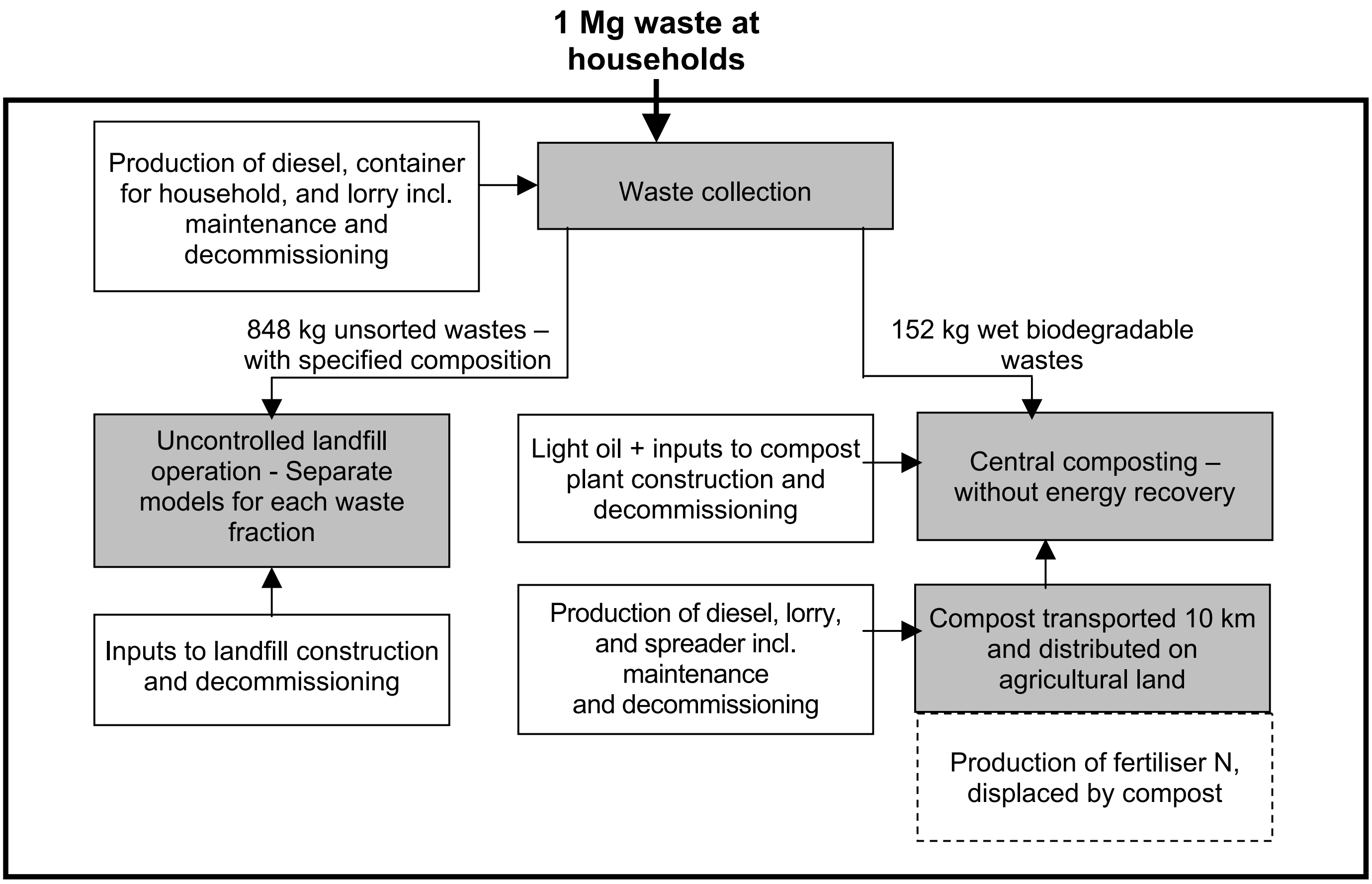

Figure 1 **Flow diagram for the baseline scenario (A) for Malta. Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.**

In the baseline scenario for Krakow, 80% of households are connected to the regular waste collection, and their household waste goes to a directive compliant landfill (JRC 2005c). In addition to the directive compliant landfill, there is a central composting plant that receives 6,000 Mg of park & garden wastes (JRC 2005c).

For the 20% (40,000 Mg) of the Krakow household waste that is generated by the households not connected to the regular waste collection, it is assumed that 8,000 Mg waste paper (20.1% of the 40,000 Mg) is combusted in the households, 13,200 Mg wet biodegradable wastes (33% of the 40,000 Mg) is composted in the households, while the remaining 18,800 Mg are assumed to be landfilled under uncontrolled circumstances with 9,200 Mg being reclaimed and placed in directive compliant landfill. This is only relevant for scenario A. For the other scenarios, all households are expected to be connected to the regular waste collection.

Table 31 presents the amounts of wastes collected for recycling.

Table 31 Recycling in Krakow (2003). Amounts in Mg. Mainly based on assumptions.

Waste material fractions collected for recycling (Mg/year)	Paper and cardboard wastes	Plastic wastes	Glass wastes	Iron and steel wastes 1)	Aluminium wastes 1)	Total collected for recycling
Household waste collected for recycling in "banks" – known composition 2)	235	46	860	82	41	**1,264**
Household waste collected for recycling – assumed composition 2)	881	172	3,222	307	154	**4,736**
Commercial waste collected for recycling – known composition 3)	528	132				**660**
Commercial waste collected for recycling – assumed composition 3)	1,708		5,978	569	285	**8,540**
Total collected for recycling	**3,352**	**350**	**10,060**	**959**	**479**	**15,200**
Total municipal solid waste	**60,241**	**32,143**	**24,217**	**5,371**	**3,076**	
Collected for recycling in percentage of total fraction (%)	5.6%	1.1%	41.5%	17.8%	15.6%	

1) In absence of a specification, it is assumed that the recycled metal is 67% ferrous metals and 33% non-ferrous metals, which is approximately the proportion between the total amounts in the collected municipal solid waste.
2) Total amount collected for recycling is 6,000 Mg/year, out of which the composition is known for the 1,264 Mg (JRC 2005c). It is assumed that the remaining amount has the same composition as the amount for which the composition is known.
3) Total amount collected for recycling is 9,000 Mg/year, out of which the composition is known for the 660 Mg (JRC 2005c). It is assumed that the remaining amount consist of 20% paper, 70% glass, 10% Metal (which approximately corresponds to the distribution of these fractions for household waste collected for recycling.

Table 32 Technology combinations in the baseline scenario (A) for Krakow.

	Uncontrolled landfill		Directive compliant landfill		Incineration at home		Material recycling		Central composting		Home composting		Sums
	Mg	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**
Wet biodegradable			69780	78%					6000	7%	13200	15%	**88980**
Paper and cardboard			48889	81%	8000	13%	3352	6%					**60241**
Plastic wastes	2763	9%	29029	90%			350	1%					**32143**
Glass wastes	2272	9%	11885	49%			10060	42%					**24217**
Iron and steel wastes	450	8%	3962	74%			959	18%					**5371**
Aluminium wastes	246	8%	2351	76%			479	16%					**3076**
Other wastes	3869	9%	38304	91%									**42172**

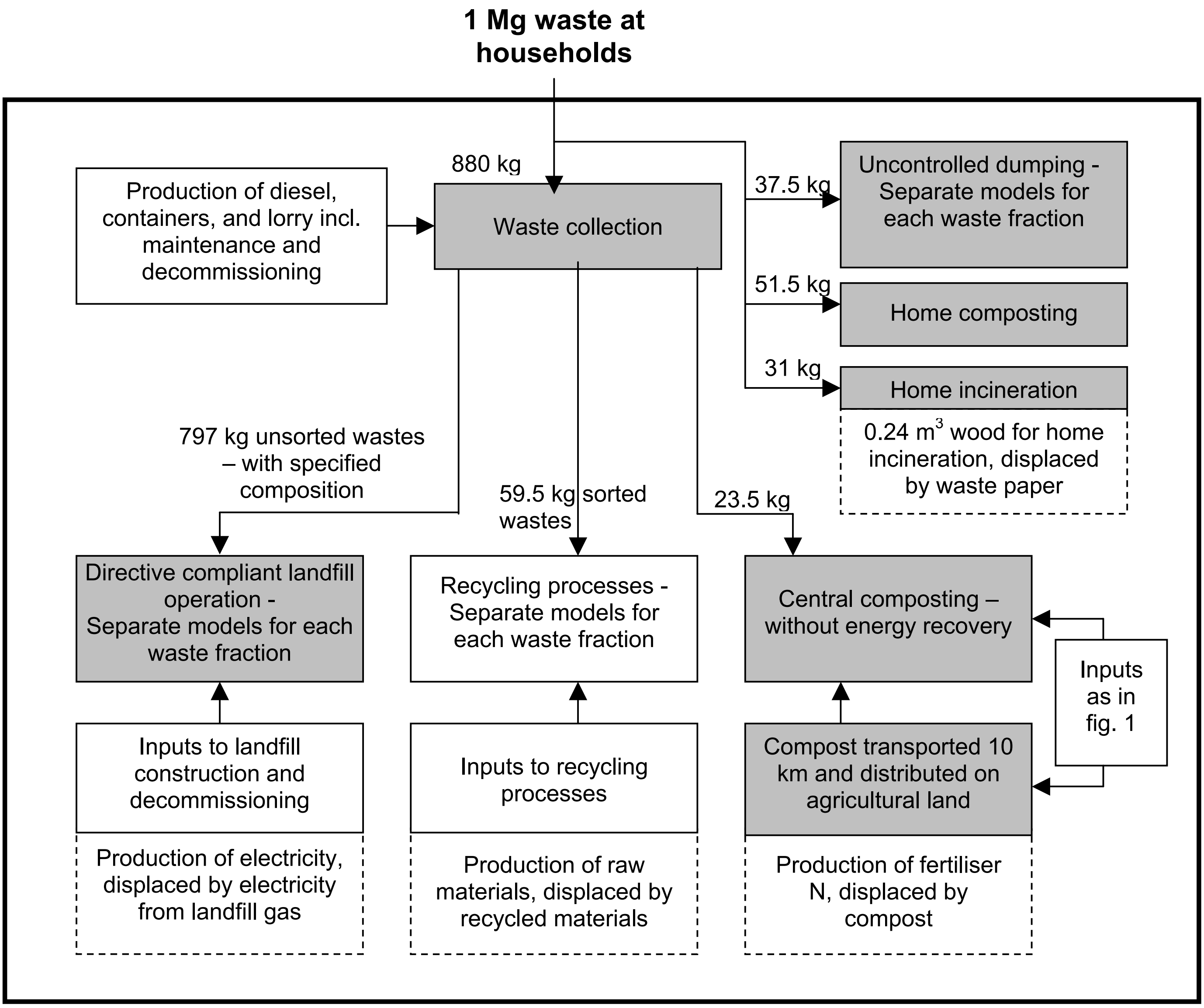

Figure 2 **Flow diagram for the baseline scenario (A) for Krakow. Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.**

It should be noted that electricity production plants in Poland are linked to the European UCTE network, and that the impacts from a change in electricity consumption in Krakow cannot be identified geographically. This is in contrast to the situation for Malta (compare e.g. Figure 1 and Figure 2) where the electricity supply is local.

8.2 Incineration scenario with increased recycling (B)

In the incineration scenarios, all municipal solid waste is assumed to be incinerated, except the amounts that shall be recycled to fulfil the ultimate percentage requirements of the packaging waste directive 94/62/EC, noting again that the ultimate percentage requirements of the packaging waste directive are applied in this study not only to the packaging waste but also to the rest of the waste that belongs to the same waste type.

The ultimate recycling target of 60% from the packaging waste directive is applied here to all glass and paper in the municipal solid waste, the ultimate recycling target of 50% is applied to all metal in the municipal solid waste, and the ultimate recycling target of 22.5% is applied to all plastics in the municipal solid waste. The ultimate overall minimum of 55% of recycling of these fractions is ensured by first increasing the recycling of the fraction that can be recycled with the lowest marginal cost (according to Table 28) up to the maximum attainable recycling percentage for that fraction, then increasing recycling for the fraction with the next-lowest recycling costs, and so on until the overall target of 55% is reached. Alternative recycling options and criteria are considered in Scenarios D and E.

Also the requirement of the landfill directive, that only 35% of the weight of biodegradable waste production in 1995 be landfilled, is met by directing all wet biodegradable wastes to incineration and all paper to either incineration or recycling.

From Table 26 and Table 28, it is the value of the recycled materials that determines which fractions are most economical to recycle. In practice, this implies that aluminium, non-PE plastics, iron and steel recycling is optimised to its maximum 80% (based on the modelling of Tucker & Speirs 2002) in both Malta and Krakow, and recycling of glass is increased to 72% and 64% in Malta and Krakow, respectively, to reach the 55% overall recycling target, while the remaining “Paper and cardboard wastes” fraction is kept at the level of the packaging directive requirements.

Within the fraction “Paper and cardboard wastes”, the packaging directive requirement is reached by increasing to their maximum recycling percentages, those fractions that give the lowest marginal economic costs to recycle, i.e. cardboard recycling is increased to 85% (based on the modelling of Tucker & Speirs 2002), and the rest is taken up by newsprint recycling.

The following tables and figures present the resulting scenarios.

Table 33 Technology combinations in the incineration scenario (B) for Malta.

	Directive compliant incineration		Material recycling		Sums
	Mg	**%**	**Mg**	**%**	**Mg**
Wet biodegradable	97402	100%	0	0%	**97402**
Paper & cardboard wastes	8008	40%	11777	60%	[a] **19785**
- of which: Newsprint	*4079*	*43%*	*5319*	*57%*	***9398***
- of which: Cardboard	*1140*	*15%*	*6458*	*85%*	***7597***

	Directive compliant incineration		Material recycling		Sums
	Mg	%	Mg	%	Mg
- of which: Other paper	*2790*	*100%*	*0*	*0%*	***2790***
Plastic wastes	8963	68%	4218	32%	[a] **13181**
- of which: PE	*7909*	*100%*	*0*	*0%*	***7909***
- of which: Other plastics	*1054*	*20%*	*4218*	*80%*	***5272***
Glass wastes	1445	28%	3734	72%	[a] **5179**
Iron and steel wastes	916	20%	3664	80%	[a] **4580**
Aluminium wastes	69	20%	277	80%	[a] **346**
Other wastes	13168	100%	0	0%	**13168**
Sum	**129971**	**85%**	[a] **23670**	**15%**	**153641**

a) 23670 Mg = 55% of 43071Mg (19785 Mg + 13181 Mg + 5179 Mg + 4580 Mg + 346 Mg)

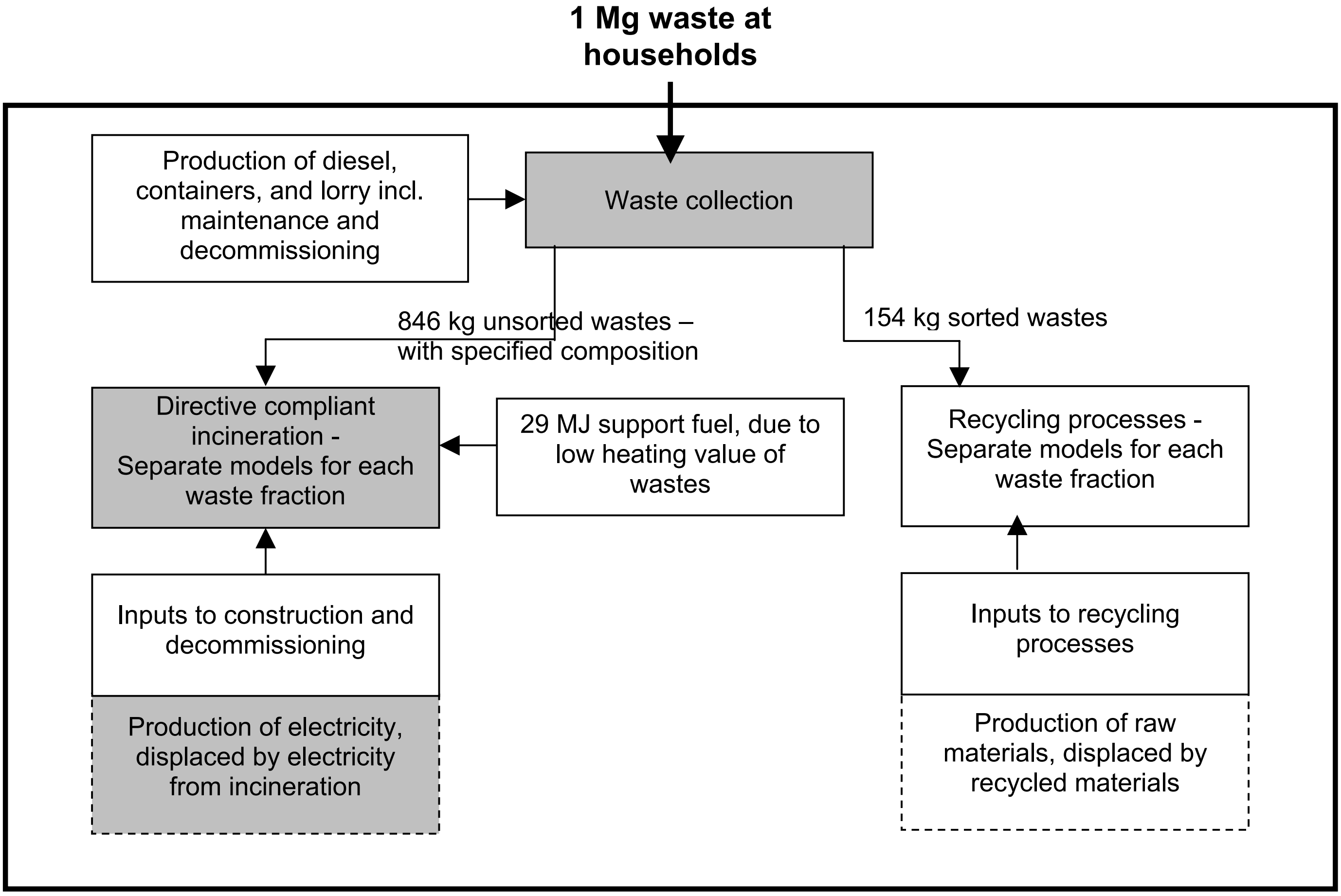

Figure 3 **Flow diagram for the incineration scenario (B) for Malta. Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.**

Table 34 Technology combinations in the incineration scenario (B) for Krakow.

	Directive compliant incineration		Material recycling		Sums
	Mg	**%**	**Mg**	**%**	**Mg**
Wet biodegradable	88980	100%		0%	**88980**
Paper & cardboard wastes	24125	40%	36116	60%	[a] **60241**
- of which: Newsprint	*12161*	*43%*	*16453*	*57%*	***28614***
- of which: Cardboard	*3470*	*15%*	*19663*	*85%*	***23132***
- of which: Other paper	*8494*	*100%*	*0*	*0%*	***8494***
Plastic wastes	21857	68%	10286	32%	[a] **32143**
- of which: PE	*19286*	*100%*	*0*	*0%*	***19286***
- of which: Other plastics	*2571*	*20%*	*10286*	*80%*	***12857***
Glass wastes	8646	36%	15572	64%	[a] **24217**
Iron and steel wastes	1074	20%	4297	80%	[a] **5371**
Aluminium wastes	615	20%	2461	80%	[a] **3076**
Other wastes	42172	100%		0%	**42172**
Sum	**187469**	**73%**	[a] **68731**	**27%**	**256200**

a) 68731 Mg = 55% of 125048 Mg (60241 Mg + 32143 Mg + 24217 Mg + 5371 Mg + 3076 Mg)

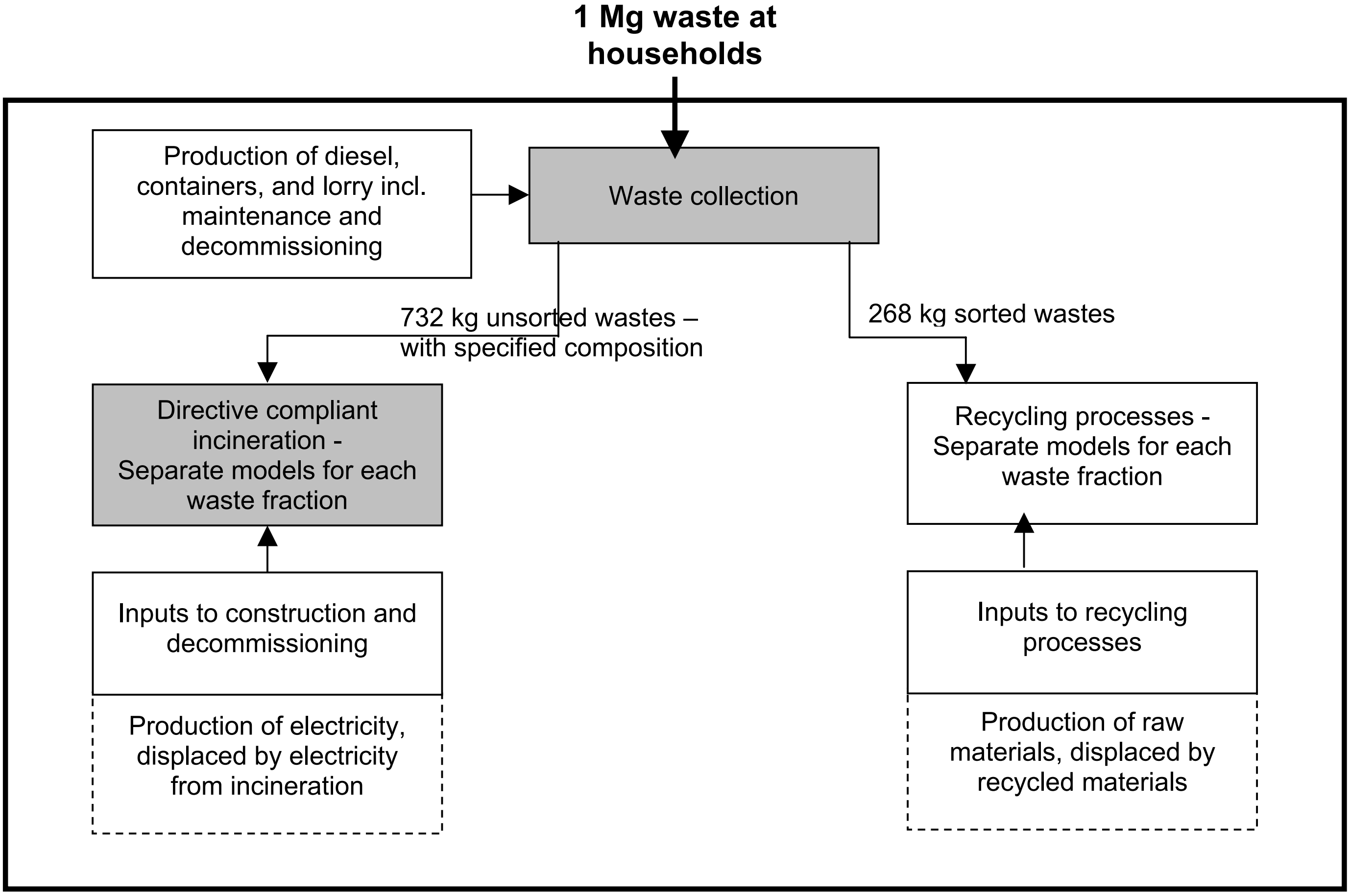

Figure 4 **Flow diagram for the incineration scenario (B) for Krakow. Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.**

8.3 Composting scenario with increased recycling (C)

In the composting scenarios, the requirement of the landfill directive 99/31/EC to divert a specific weight percentage of the biodegradable wastes, including paper and cardboard wastes, away from landfilling, is fulfilled by increasing composting of wet biodegradable wastes and recycling paper.

As in scenario B, recycling is first increased to the ultimate percentage requirements of the packaging waste directive, i.e. 60 % for paper and glass, 50 % for metals, and 22.5 % for plastics, and then increased further to 80% for aluminium and then for plastics to the percentage necessary to fulfil an overall 55% recycling target for these fractions together.

The ultimate landfill directive target is that only 35% of the weight of biodegradable waste production in 1995 be landfilled. This 1995 production is 99,500 Mg for Malta and 112,000 Mg for Krakow (JRC 2005c), which is significantly lower than the 2003 production – 117,187 Mg and 149,221 Mg, respectively. The resulting targets therefore become significantly higher than 65% of the current biodegradable waste levels.

These targets are reached here by increasing composting of wet biodegradable wastes to 72.5% for Malta and 80% for Krakow, the latter being the assumed maximum attainable in practice. Since this is not enough to reach the target in Krakow, recycling of paper is then increased to 64.5%, and the recycling target for plastics at the same time reduced, so that the fulfilment of the overall 55% recycling target is maintained in relation to minimum compliance. The remaining wastes are assumed to be deposited in a directive compliant landfill. The following table and figures present the resulting scenarios.

Table 35 Technology combinations in the composting scenario (C) for Malta.

	Directive compliant landfill		Material recycling		Central composting with energy recovery		Sums
	Mg	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**
Wet biodegradable	[a] 26817	28%	0	0%	70585	72%	**97402**
Paper & cardboard wastes	[a] 8008	40%	11777	60%			**19785**
- of which: Newsprint	*4079*	*43%*	*5319*	*57%*			***9398***
- of which: Cardboard	*1140*	*15%*	*6458*	*85%*			***7597***
- of which: Other paper	*2790*	*100%*	*0*	*0%*			***2790***
Plastic wastes	6962	53%	6219	47%			**13181**
- of which: PE	*5908*	*75%*	*2001*	*25%*			***7909***
- of which: Other plastics	*1054*	*20%*	*4218*	*80%*			***5272***
Glass wastes	2072	40%	3108	60%			**5179**
Iron and steel wastes	2290	20%	2290	80%			**4580**
Aluminium wastes	69	69%	277	80%			**346**
Other wastes	13168	100%	0	0%			**13168**
Sum	**59386**	**39%**	**23670**	**15%**	**70585**	**46%**	**153641**

a) 26817 Mg + 8008 Mg = 34825 Mg = 35% of 99500 Mg (baseline 1995)

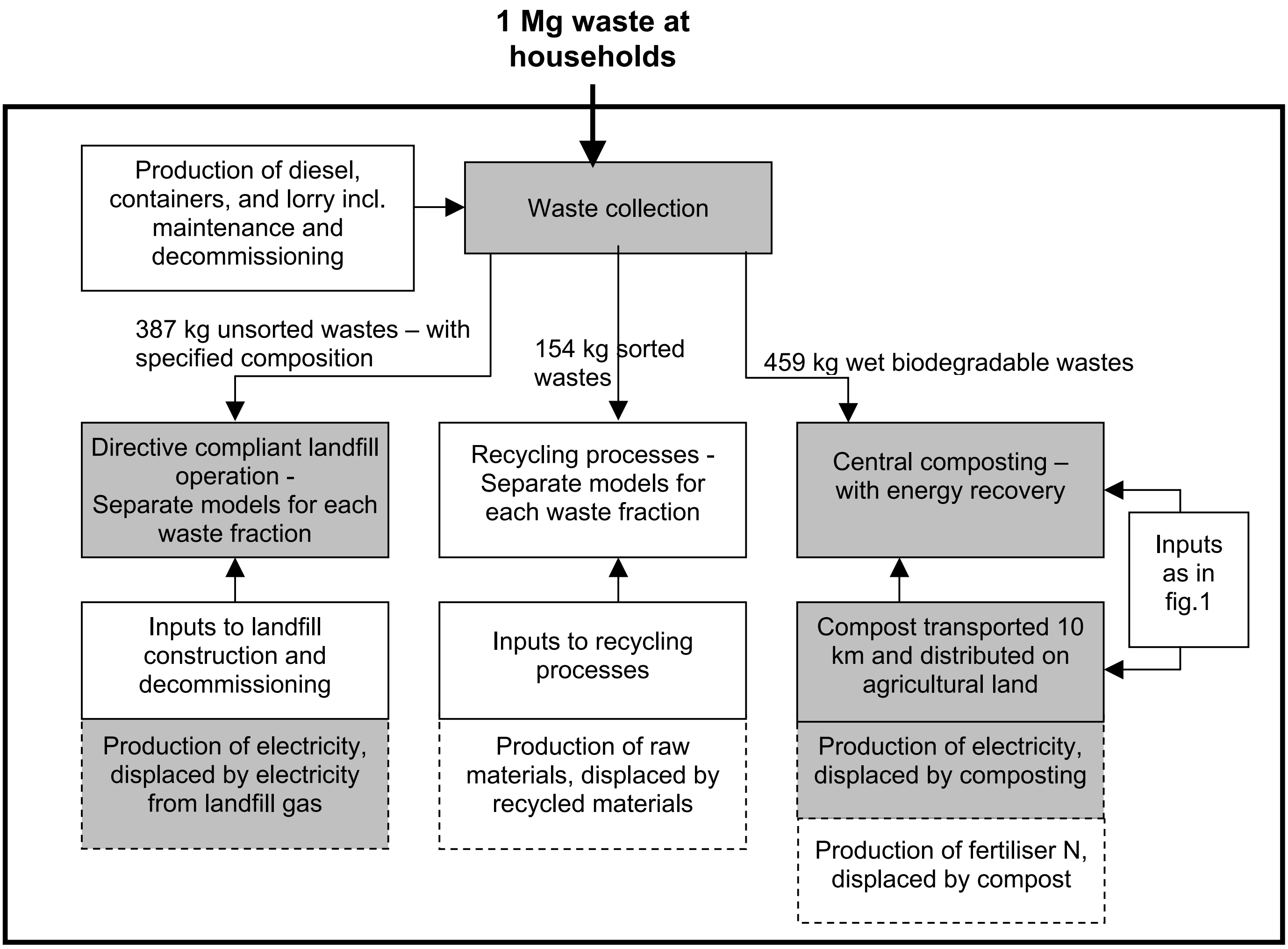

Figure 5 **Flow diagram for the composting scenario (C) for Malta.**
Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.

Table 36 **Technology combinations in the composting scenario (C) for Krakow.**

	Directive compliant landfill		**Material recycling**		**Central composting with energy recovery**		**Sums**
	Mg	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**
Wet biodegradable	[a] 17793	20%	0	0%	71187	80%	**88980**
Paper & cardboard wastes	[a] 21407	36%	38834	64%			**60241**
- of which: Newsprint	*9443*	*33%*	*19172*	*67%*			***28614***
- of which: Cardboard	*3470*	*15%*	*19663*	*85%*			***23132***
- of which: Other paper	*8494*	*100%*	*0*	*0%*			***8494***
Plastic wastes	21856	68%	10286	32%			**32143**

	Directive compliant landfill		Material recycling		Central composting with energy recovery		Sums
	Mg	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**
- of which: PE	*19286*	*100%*	*0*	*0%*			***19286***
- of which: Other plastics	*2571*	*20%*	*10286*	*80%*			***12857***
Glass wastes	9687	40%	14530	60%			**24217**
Iron and steel wastes	2685	50%	2685	50%			**5371**
Aluminium wastes	615	20%	2461	80%			**3076**
Other wastes	42172	100%	0	0%			**42172**
Sum	**116217**	**45%**	**68796**	**27%**	**71187**	**28%**	**256200**

a) 17793 Mg + 21407 Mg = 39200 Mg = 35% of 112000 Mg (baseline 1995)

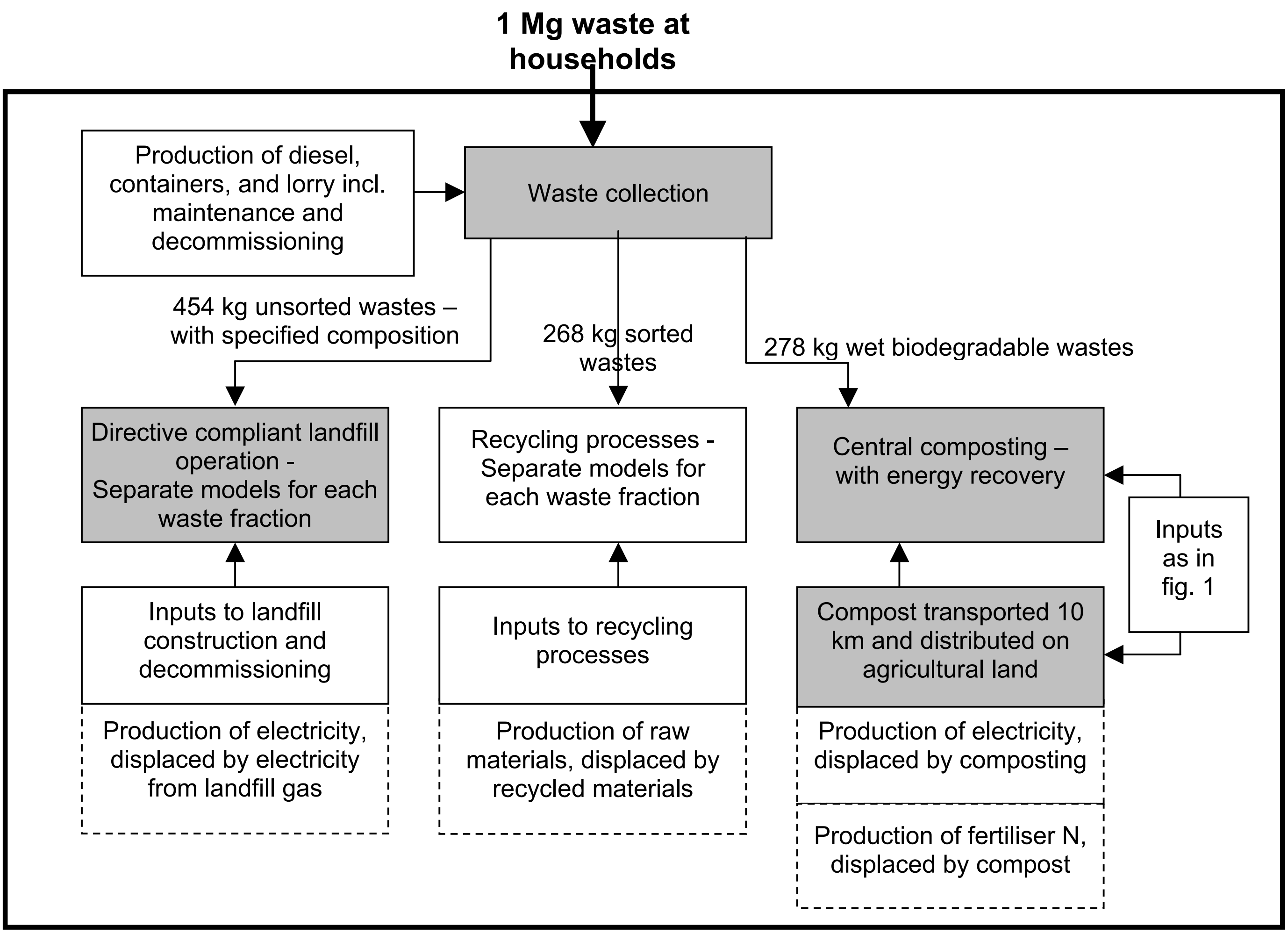

Figure 6 **Flow diagram for the composting scenario (C) for Krakow.**
Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.

8.4 Economic optimum scenario (D)

In this economic optimum scenario, the technologies are combined in such a way that the ultimate directive requirements considered in the previous scenarios are fulfilled, while minimising overall economic costs, according to the cost calculations in Chapter 7. The resultant Scenario is somewhat analogous to Scenario B, with more recycling and incineration.

From Table 27 and Table 28, recycling is most often the least cost option (due to revenues from sale of materials), except for the fractions “Other paper”, PE, and “wet biodegradable”, where incineration is typically the least cost option. For Malta, it is also preferable to incinerate the “Newsprint” fraction, when using average prices. Thus, the economic optimum scenarios use the maximum attainable recycling percentages (85% for glass, paper and cardboard, and 80% for all others) for all fractions, except “Other paper”, PE, and “wet biodegradable”, which are 100% incinerated along with the residual wastes. For Malta, newsprint is only recycled to the extent necessary to reach the overall directive requirement for paper of 60% recycling.

The following tables and figures present the resulting scenarios.

Table 37 Technology combinations in the economic optimum scenario (D) for Malta.

	Directive compliant incineration		Material recycling		Sums
	Mg	**%**	**Mg**	**%**	**Mg**
Wet biodegradable	97402	100%	0	0%	**97402**
Paper and cardboard	8008	40%	11777	60%	**19785**
- of which: Newsprint	*4079*	*43%*	*5319*	*57%*	***9398***
- of which: Cardboard	*1140*	*15%*	*6458*	*85%*	***7597***
- of which: Other paper	*2790*	*100%*	*0*	*0%*	***2790***
Plastic wastes	8963	68%	4218	32%	**13181**
- of which: PE	*7909*	*100%*	*0*	*0%*	***7909***
- of which: Other plastics	*1054*	*20%*	*4218*	*80%*	***5272***
Glass wastes	777	15%	4402	85%	**5179**
Iron and steel wastes	916	20%	3664	80%	**4580**
Aluminium wastes	69	20%	277	80%	**346**
Other wastes	13168	100%	0	0%	**13168**
Sum	**129303**	**84%**	[a] **24338**	**16%**	**153641**

a) 24338 Mg = 56.5% of 43071Mg (the total weight of the material recyclable fractions)

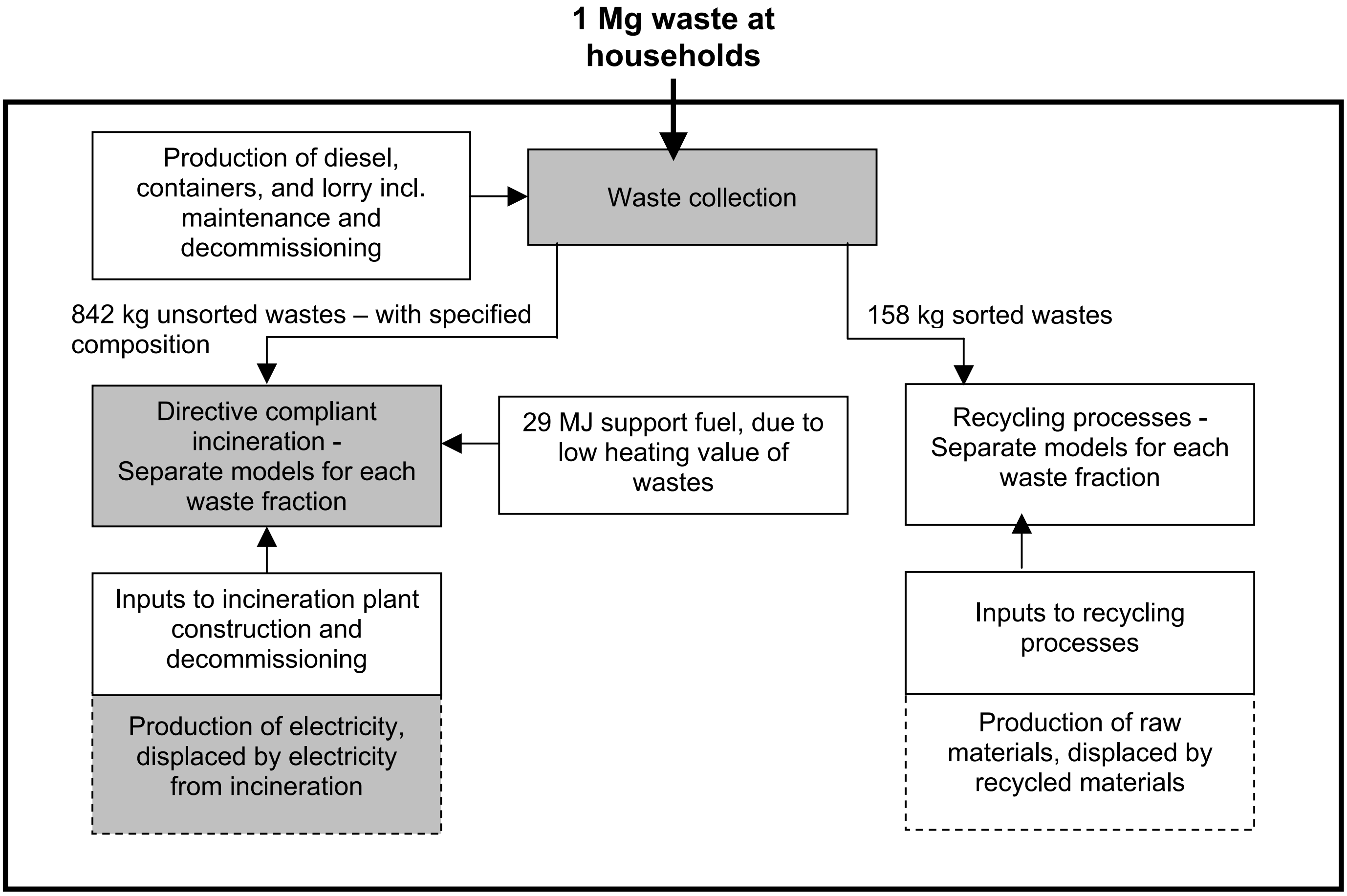

Figure 7 **Flow diagram for the economic optimum scenario (D) for Malta. Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.**

Table 38 **Technology combinations in the economic optimum scenario (D) for Krakow.**

	Directive compliant incineration		Material recycling		Sums
	Mg	**%**	**Mg**	**%**	**Mg**
Wet biodegradable	88980	100%	0	0%	**88980**
Paper & cardboard wastes	16256	27%	43985	73%	**60241**
- of which: Newsprint	*4292*	*15%*	*24322*	*85%*	***28614***
- of which: Cardboard	*3470*	*15%*	*19663*	*85%*	***23132***
- of which: Other paper	*8494*	*100%*	*0*	*0%*	***8494***
Plastic wastes	21857	68%	10286	32%	**32143**
- of which: PE	*19286*	*100%*	*0*	*0%*	***19286***
- of which: Other plastics	*2571*	*20%*	*10286*	*80%*	***12857***
Glass wastes	3633	15%	20585	85%	**24217**

	Directive compliant incineration		Material recycling		Sums
	Mg	**%**	**Mg**	**%**	**Mg**
Iron and steel wastes	1074	20%	4297	80%	**5371**
Aluminium wastes	615	20%	2461	80%	**3076**
Other wastes	42172	100%		0%	**42172**
Sum	**174587**	**68%**	[a] **81613**	**32%**	**256200**

a) 81613 Mg = 65.3 % of 125048 Mg (the total weight of the material recyclable fractions)

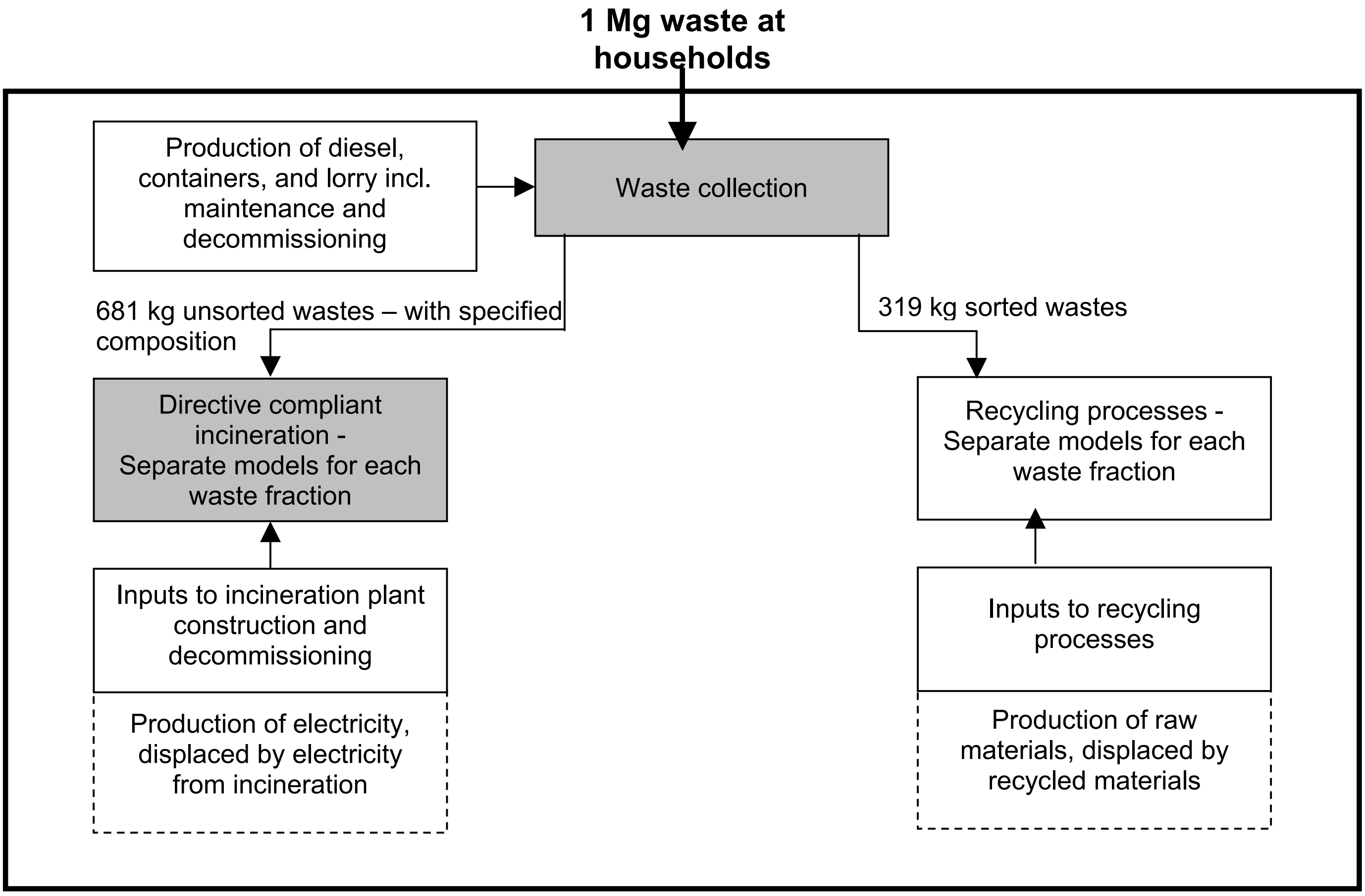

Figure 8 **Flow diagram for the economic optimum scenario (D) for Krakow. Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.**

8.5 Societal optimum scenario (E)

In this so-called “societal” optimum scenario, the technologies are combined in such a way that the ultimate directive requirements considered in the previous scenarios are fulfilled, while minimising overall costs, including externalities. Externalities are calculated as described in Chapter 6.2. The construction of this scenario is based on the results of comparing the different technological options presented in Chapter 9.6. The resultant Scenario is somewhat analogous to Scenario C, but with more recycling, more biocomposting and the addition of some incineration.

As presented in Chapter 9.6, composting with recycling is practically always the least cost option, when externalities are considered. For the remaining wastes ("Other wastes" and the residual that is not separately collected) incineration is the least cost option. Incineration for the fraction "Other paper" was also maintained, since the quality of this fraction for recycling is unknown, while all other separately collected fractions are recycled. Thus, the societal optimum scenarios use the maximum attainable recycling percentages (85% for glass, paper and cardboard, and 80% for all other fractions) for all fractions, including PE and "wet biodegradable" for composting.

Both for Malta and Krakow, the high degree of recycling implies that the amounts of remaining wastes are too small (42,000 Mg and 88,000 Mg, respectively) to justify the building of an incineration plant for the wastes from these two study areas only. Thus, the wastes for incineration will be transported an additional distance. For Malta, transport to the European continent (e.g. Barcelona) has been assumed, as for the scrap, see Chapter 5.2.1. For Krakow, an additional transport of 100 km with large (32t) trucks is assumed. The following tables and figures present the resulting scenarios.

Table 39 Technology combinations in the societal optimum scenario (E) for Malta.

	Directive compliant incineration		Material recycling		Central composting with energy recovery		Sums
	Mg	%	Mg	%	Mg	%	Mg
Wet biodegradable	19480	20%	0	0%	77921	80%	**97402**
Paper and cardboard	5339	27%	14446	73%			**19785**
- of which: Newsprint	1410	15%	7988	85%			**9398**
- of which: Cardboard	1140	15%	6458	85%			**7597**
- of which: Other paper	2790	100%	0	0%			**2790**
Plastic wastes	2636	20%	10545	80%			**13181**
- of which: PE	1582	20%	6327	80%			**7909**
- of which: Other plastics	1054	20%	4218	80%			**5272**
Glass wastes	777	15%	4402	85%			**5179**
Iron and steel wastes	916	20%	3664	80%			**4580**
Aluminium wastes	69	20%	277	80%			**346**

	Directive compliant incineration		Material recycling		Central composting with energy recovery		Sums
	Mg	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**
Other wastes	13168	100%	0	0%			**13168**
Sum	**42385**	**27%**	[a] **33334**	**22%**	**77921**	**51%**	**153641**

a) 33334 Mg = 77.4% of 43071Mg (the total weight of the material recyclable fractions)

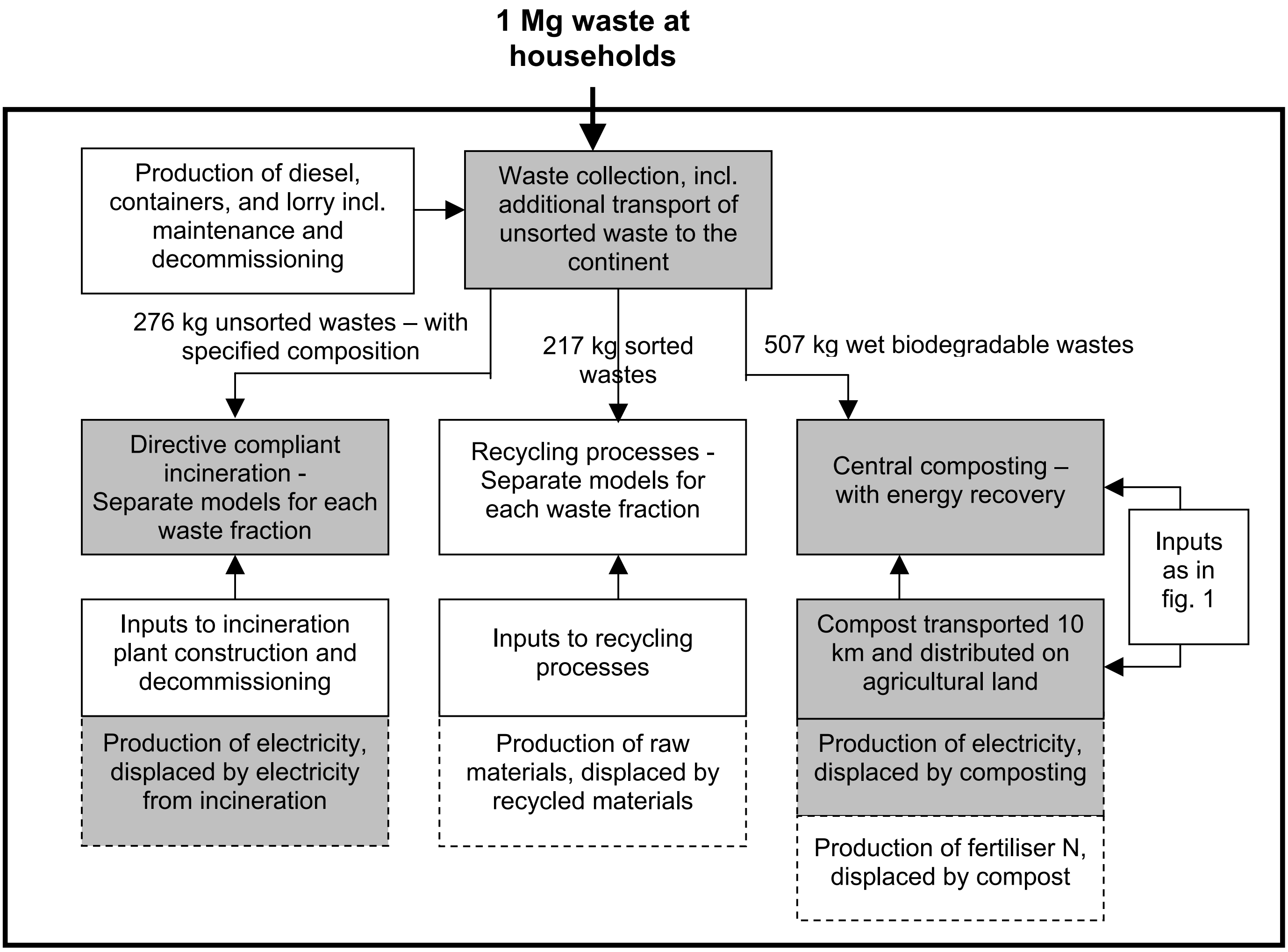

Figure 9 **Flow diagram for the societal optimum scenario (E) for Malta. Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.**

Table 40 Technology combinations in the societal optimum scenario (E) for Krakow.

	Directive compliant incineration		Material recycling		Central composting with energy recovery		Sums
	Mg	**%**	**Mg**	**%**	**Mg**	**%**	**Mg**
Wet biodegradable	17796	20%	0	0%	71184	80%	**88980**
Paper & cardboard wastes	16256	27%	43985	73%			**60241**
- of which: Newsprint	*4292*	*15%*	*24322*	*85%*			***28614***
- of which: Cardboard	*3470*	*15%*	*19663*	*85%*			***23132***
- of which: Other paper	*8494*	*100%*	*0*	*0%*			***8494***
Plastic wastes	6429	20%	25714	80%			**32143**
- of which: PE	*3857*	*20%*	*15429*	*80%*			***19286***
- of which: Other plastics	*2571*	*20%*	*10286*	*80%*			***12857***
Glass wastes	3633	15%	20585	85%			**24217**
Iron and steel wastes	1074	20%	4297	80%			**5371**
Aluminium wastes	615	20%	2461	80%			**3076**
Other wastes	42172	100%	0	0%			**42172**
Sum	**87975**	**34%**	[a] **97041**	**38%**	**71184**	**28%**	**256200**

a) 97041 Mg = 77.6% of 125048 Mg (the total weight of the material recyclable fractions)

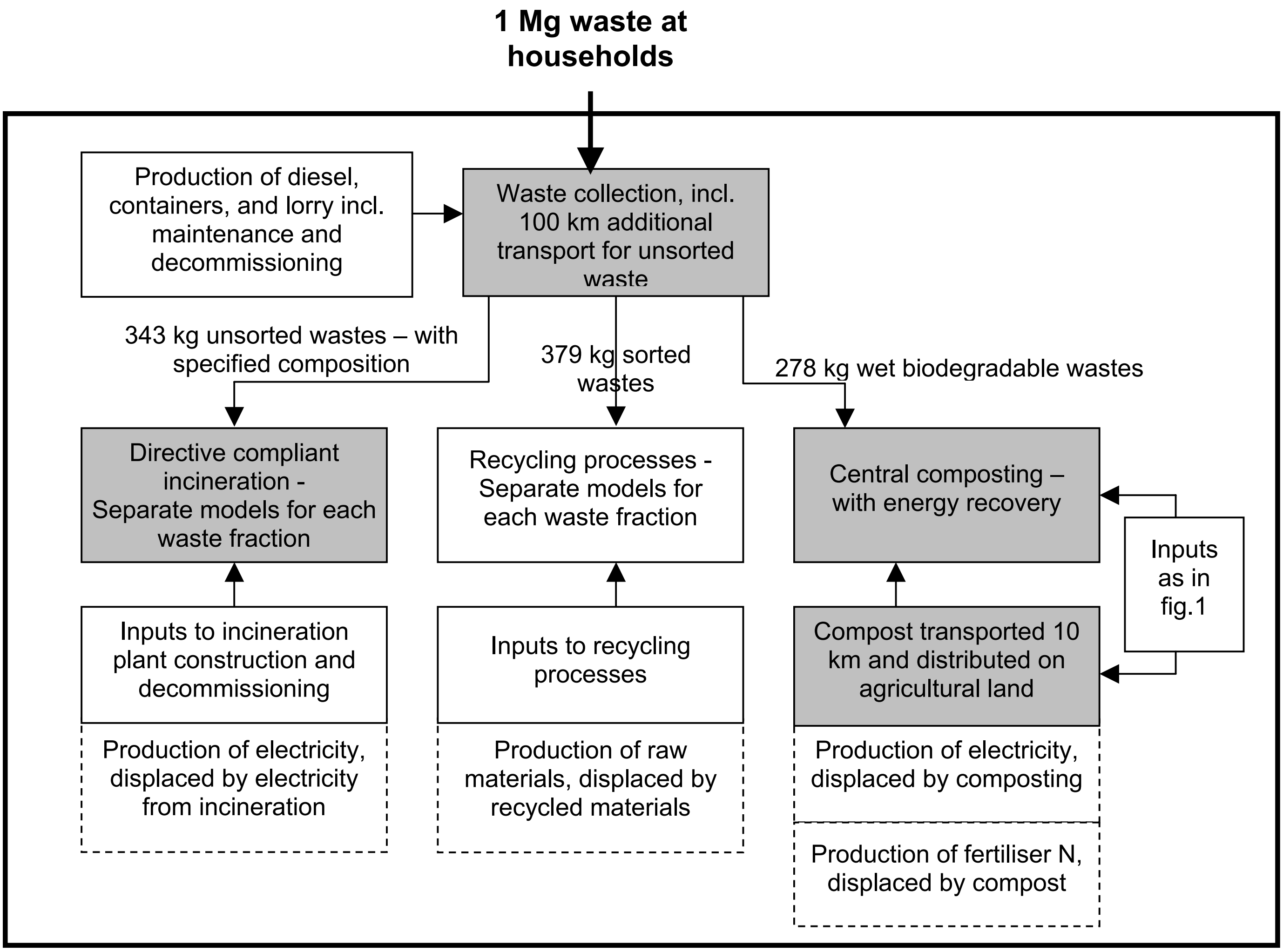

Figure 10 **Flow diagram for the societal optimum scenario (E) for Krakow. Shaded boxes represent processes for which site-dependent emission models are applied. Emissions from boxes with dotted lines (representing displaced processes) are subtracted from the overall emissions of the system.**

9 Life cycle assessment results

When looking at the different tables and figures in this Chapter, it should be remembered that the data shown are net results for each of the included impact categories, i.e. results are often the sum of both positive and negative values from a large number of processes, releases, and impact types. For example, in the below Table 41, the value for acidification for Krakow scenario A of –6.21 m^2 UES (unprotected ecosystem) is in fact the net result of an impact of 8.69 m^2 UES from a total of 255 unit processes and a displaced (avoided) impact of 14.90 m^2 UES from other 241 unit processes. The overall result is therefore a net avoidance of acidification that would occur due to other existing processes.

9.1 Midpoint results for the Krakow scenarios

Table 41 presents the midpoint indicator results per impact category. Table 42 provides the results for emissions to groundwater, without reduction factors. The latter values imply that emissions to groundwater are given the same characterisation factors as emissions to surface waters, i.e. this is a worst case assumption that there is no degradation or loss due to e.g. binding with soils prior to reaching surface waters (see Chapters 6.1.1 and 6.1.2 for further discussion). In Table 43 and Table 44, the same results are presented relative to the annual emissions per person in Europe (see Table 18).

Table 41 Midpoint indicator results for the Krakow scenarios, with reduced factors for emissions to groundwater (see also Chapters 6.1.1 and 6.1.2).

Impact category	Unit	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
		per 1 Mg waste (Krakow)				
Acidification	m^2 UES	*-6*	-43	-29	-46	**-57**
Ecotoxicity, aquatic	Kg-eq. TEG water	*1,790*	-47,200	-29,400	**-48,400**	-45,300
Ecotoxicity, aquatic, groundw.	Kg-eq. TEG water	*-34*	-351	-184	**-363**	-273
Ecotox., aquatic, long-term gr.	Kg TEG water	**1.6E+05**	3.4E+05	4.0E+05	3.9E+05	*4.4E+05*
Ecotoxicity, terrestrial	kg TEG soil	*-8,740*	-39,700	-34,500	-44,500	**-44,900**
Eutrophication, aquatic	kg NO_3-eq.	*3.64*	-0.17	1.47	**-0.18**	-0.15
Eutrophic., aquatic, groundw.	kg NO_3-eq.	0.34	**0.10**	0.92	0.10	*0.95*

Impact category	**Unit**	**Scenario A**	**Scenario B**	**Scenario C**	**Scenario D**	**Scenario E**
		per 1 Mg waste (Krakow)				
Eutrophic., aquatic, long-term	kg NO_3-eq.	*5.18*	0.20	2.91	0.19	**0.04**
Eutrophication, terrestrial	m^2 UES	*-2*	-40	-23	-44	**-48**
Climate change	kg CO_2-eq.	*499*	-30	-47	-74	**-180**
Human toxicity	Kg C_2H_3Cl-eq.	*-0.5*	-7.6	-9.5	-10.8	**-11.1**
Human toxicity, groundwater	Kg C_2H_3Cl-eq.	**0.3**	*6.5*	0.9	6.1	4.1
Human toxicity, long-term gr.	Kg C_2H_3Cl-eq.	*105.0*	22.0	87.3	19.6	**14.8**
Injuries, road or work	fatal injuries-eq.	2.2E-07	2.1E-07	**1.7E-07**	2.0E-07	*2.4E-07*
Ionizing radiation	Bq C-14-eq.	*-589*	-3,920	-3,370	**-3,970**	-3,740
Mineral extraction	MJ extra	*-17*	-87	-73	-87	**-88**
Nature occupation	m^2 arable land	*5*	-19	-13	**-24**	-23
Non-renewable energy	MJ primary	*-885*	-12,000	-7,040	-12,300	**-14,400**
Ozone layer depletion	Kg CFC-11-eq.	*-2.1E-07*	-5.4E-07	-4.5E-07	**-6.3E-07**	-5.7E-07
Photochemical ozone – Veg.	m^2*ppm*hours	*5,260*	-3,710	-800	-3,900	**-4,720**
Respiratory inorganics	Kg PM2.5-eq.	*-0.12*	-0.75	-0.56	-0.80	**-0.93**
Respiratory organics	pers*ppm*hours	*0.63*	-0.42	-0.04	-0.44	**-0.48**

The best results are **bolded,** the worst are italic.

Table 42 **Midpoint indicator results for groundwater emissions in the Krakow scenarios, when applying the same characterisation factors as for emissions to surface water.**

Impact category	Unit	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
		per 1 Mg waste (Krakow)				
Ecotoxicity, aquatic, groundw.	kg-eq. TEG water	*-85,250*	-877,500	-460,000	**-907,500**	-682,500
Ecotox., aquatic, long-term gr.	kg TEG water	**2.7E+11**	5.7E+11	6.7E+11	6.5E+11	*7.3E+11*
Eutrophic., aquatic, groundw.	kg NO_3-eq.	857	**248**	2,307	250	*2,375*
Eutrophic., aquatic, long-term	kg NO_3-eq.	*8.6E+06*	3.4E+05	4.8E+06	3.2E+05	**6.3E+04**
Human toxicity, groundwater	kg C_2H_3Cl-eq.	**823**	*16,225*	2,203	15,350	10,225
Human toxicity, long-term gr.	kg C_2H_3Cl-eq.	*1.7E+08*	3.7E+07	1.5E+08	3.3E+07	**2.5E+07**

The best results are **bolded,** the worst are italic.

Table 43 **Midpoint indicator results for the Krakow scenarios, with reduced factors for emissions to groundwater, relative to the annual emissions per person in Europe.**

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Krakow)				
Acidification	*-2.8E-03*	-2.0E-02	-1.3E-02	-2.1E-02	**-2.6E-02**
Ecotoxicity, aquatic	*2.5E-03*	-6.7E-02	-4.2E-02	**-6.9E-02**	-6.4E-02
Ecotoxicity, aquatic, groundw.	*-1.9E-08*	-2.0E-07	-1.0E-07	**-2.1E-07**	-1.5E-07
Ecotox., aquatic, long-term gr.	**1.4E-07**	2.9E-07	3.4E-07	3.3E-07	*3.7E-07*
Ecotoxicity, terrestrial	*-1.2E-01*	-5.6E-01	-4.8E-01	-6.2E-01	**-6.3E-01**
Eutrophication, aquatic	*6.3E-02*	-2.8E-03	2.5E-02	**-3.0E-03**	-2.7E-03
Eutrophic., aquatic, groundw.	2.4E-06	**6.8E-07**	6.3E-06	6.9E-07	*6.5E-06*
Eutrophic., aquatic, long-term	*5.3E-08*	2.1E-09	3.0E-08	2.0E-09	**3.9E-10**
Eutrophication, terrestrial	*-9.2E-04*	-1.9E-02	-1.1E-02	-2.1E-02	**-2.3E-02**
Climate change	*4.7E-02*	-2.9E-03	-4.4E-03	-7.0E-03	**-1.7E-02**
Human toxicity	*-1.5E-03*	-2.3E-02	-2.8E-02	-3.2E-02	**-3.3E-02**
Human toxicity, groundwater	**3.9E-07**	*7.8E-06*	1.1E-06	7.4E-06	4.9E-06
Human toxicity, long-term gr.	*1.9E-07*	4.0E-08	1.6E-07	3.5E-08	**2.7E-08**
Injuries, road or work	1.6E-03	1.5E-03	**1.2E-03**	1.4E-03	*1.7E-03*

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Krakow)				
Ionizing radiation	*-1.1E-03*	-7.4E-03	-6.3E-03	**-7.5E-03**	-7.0E-03
Mineral extraction	*-5.7E-02*	-3.0E-01	-2.5E-01	-3.0E-01	**-3.0E-01**
Nature occupation	*1.6E-03*	-6.7E-03	-4.5E-03	**-8.3E-03**	-7.9E-03
Non-renewable energy	*-5.8E-03*	-7.9E-02	-4.6E-02	-8.1E-02	**-9.5E-02**
Ozone layer depletion	*-1.0E-06*	-2.6E-06	-2.2E-06	**-3.1E-06**	-2.8E-06
Photochemical ozone – Veg.	*3.7E-02*	-2.6E-02	-5.7E-03	-2.8E-02	**-3.4E-02**
Respiratory inorganics	*-1.4E-02*	-8.6E-02	-6.4E-02	-9.2E-02	**-1.1E-01**
Respiratory organics	*6.3E-02*	-4.2E-02	-4.5E-03	-4.4E-02	**-4.8E-02**

The best results are **bolded,** the worst are italic.

Table 44 Midpoint indicator results for groundwater emissions in the Krakow scenarios, relative to the annual emissions per person in Europe (see Table 18), when applying the same characterisation factors as for emissions to surface water.

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Krakow)				
Ecotoxicity, aquatic, groundw.	*-4.8E-05*	-5.0E-04	-2.6E-04	**-5.1E-04**	-3.9E-04
Ecotox., aquatic, long-term gr.	**2.3E-01**	4.9E-01	5.7E-01	5.6E-01	*6.2E-01*
Eutrophic., aquatic, groundw.	5.9E-03	**1.7E-03**	1.6E-02	1.7E-03	*1.6E-02*
Eutrophic., aquatic, long-term	*8.9E-02*	3.5E-03	5.0E-02	3.3E-03	**6.5E-04**
Human toxicity, groundwater	**9.8E-04**	*1.9E-02*	2.6E-03	1.8E-02	1.2E-02
Human toxicity, long-term gr.	*3.1E-01*	6.6E-02	2.6E-01	5.9E-02	**4.4E-02**

The best results are **bolded,** the worst are italic.

From Table 44, 1 Mg of waste (approximately the annual waste production from 1 household) to landfill (scenario A) gives a human toxicity from long-term emissions to groundwater equal to 31% of all toxic emissions for one person in a year (Table 44, last row, first number).

Long-term emissions from landfills are not included in the normalisation data, which may partly account for this high value.

Although the emissions from the 1 Mg of waste are integrated over the long-term (i.e. including the time when leachate is no longer collected and the bottom membranes of the landfills are penetrated) and thus equals the emission to water of practically all toxic metals in the waste, the large size of the long-term emissions (suggesting that a large share of total toxic emissions are from landfilled waste rather than from the rest of the life cycle) suggests, however, that groundwater emissions may be overestimated when applying the same characterisation factors for groundwater emissions as for emissions to surface waters.

This is even more apparent when looking at the results for long-term ecotoxicity, where the groundwater emissions from landfilling of residuals from incineration and recycling of 1 Mg waste add up to a net 62% of the annual ecotoxic emissions of one person (Table 44, second row, last number).

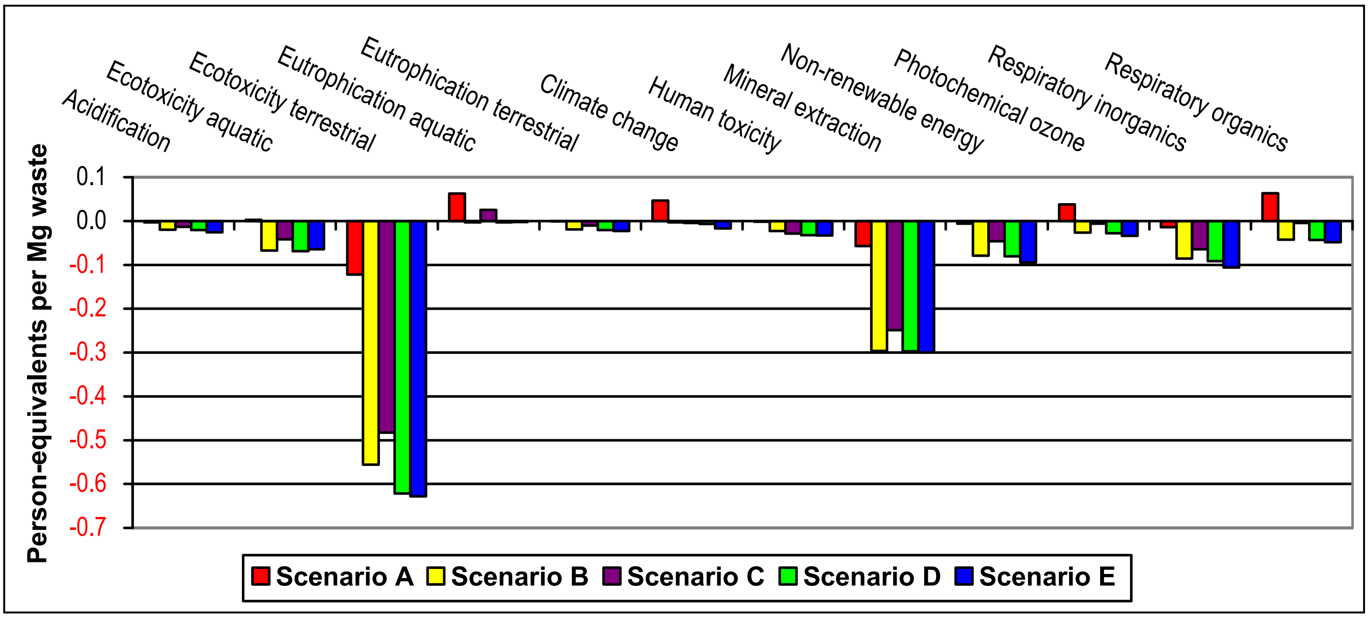

Figure 11 **Midpoint results for the Krakow scenarios, with reduced factors for groundwater emissions, relative to the annual emissions per person in Europe. Negative numbers reflect net displaced (i.e. avoided) impacts; positive numbers reflect net environmental impacts.**

Figure 11 gives a graphical representation of the most important data from Table 43. The normalised results in Table 43 and Figure 11 appear dominated by the avoided ecotoxicity and mineral resources related to materials recycling. The more recycling, the more dominating these avoided impacts are, while displaced energy production also contributes significantly to these results. At the same time, caution is required when comparing across impact categories using these results. Cross-comparison here suggests a 1:1 equivalence in severity of the results from one category to another (see Chapter 6.1.17).

For most impact categories, the results suggest an ordering of the scenarios from best to worst: E > D > B > C > A, determined first of all by the recycling rates, secondly by the degree of energy recovery.

Small deviations from this general pattern can be seen for aquatic ecotoxicity, where the economic optimum scenario D performs better than the societal optimum scenario E, and for climate change and human toxicity, where the composting scenario C performs better than the incineration scenario B. The latter differences are not, however, statistically significant.

The results for aquatic ecotoxicity are dominated by avoided heavy metal emissions from iron and steel production. As scenarios B, D and E all have the same amount of iron recycling, the small difference occurs due to the larger amount of handling of the waste in scenario E (more transport, more plants for composting and recycling).

The reason for the "incineration scenarios" (B and D) generally performing moderately better than the composting scenario C for most impact categories, is the better utilisation of the waste considered here when combining recycling and incineration, while the composting scenario still has a large amount of wastes that go "unutilized" to landfill. However, for wet biodegradable wastes, composting is considered to give a slightly better energy utilisation than incineration (see Chapter 9.6.1), and this is enough to give better average results for the compost scenario C for climate change compared to the incineration scenario B.

For human toxicity, the avoided burdens from recycling and energy recovery in scenario B is here counterweighted by the impact of dioxins from waste incineration (see Chapter 5.3.2), leading to the moderately better average performance of the composting scenario C for human toxicity. In scenario D, the recycling is increased at the expense of incineration, which reduces the net human toxicity burden enough to make scenario D perform typically better than scenario C.

9.2 Midpoint results for the Malta scenarios

Table 45 presents the midpoint indicator results per impact category. Table 46 gives the results for emissions to groundwater without reduction factors. In Table 47 and Table 48, the same results are presented relative to annual emissions per person in Europe (see Table 18).

Table 45 Midpoint indicator results for the Malta scenarios, with reduced factors for emissions to groundwater (see also Chapters 6.1.1 and 6.1.2).

Impact category	Unit	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
		per 1 Mg waste (Malta)				
Acidification	m^2 UES	*1*	-23	-21	-23	**-35**
Ecotoxicity, aquatic	kg-eq. TEG water	*18,100*	-76,900	-43,400	-76,900	**-78,300**
Ecotoxicity, aquatic, groundw.	kg-eq. TEG water	*161*	-29	26	**-29**	42
Ecotox., aquatic, long-term gr.	kg-eq. TEG water	**1.3E+05**	2.1E+05	*2.6E+05*	2.1E+05	2.6E+05
Ecotoxicity, terrestrial	kg TEG soil	*1,960*	-17,900	-15,100	-18,800	**-20,800**

Impact category	Unit	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
		per 1 Mg waste (Malta)				
Eutrophication, aquatic	kg NO_3-eq.	0.19	-0.07	*3.75*	**-0.08**	0.51
Eutrophic., aquatic, groundw.	kg NO_3-eq.	*2.80*	0.10	1.46	**0.10**	1.65
Eutrophic., aquatic, long-term	kg NO_3-eq.	*6.50*	0.44	3.92	0.43	**0.12**
Eutrophication, terrestrial	m^2 UES	*1*	-27	-23	-27	**-38**
Climate change	kg CO_2-eq.	*801*	174	97	169	**3**
Human toxicity	kg C_2H_3Cl-eq.	*0.4*	-5.1	-4.9	-5.1	**-7.6**
Human toxicity, groundwater	kg C_2H_3Cl-eq.	1.7	7.5	**1.4**	7.5	4.0
Human toxicity, long-term gr.	kg C_2H_3Cl-eq.	*40.2*	15.9	26.6	15.5	**7.9**
Injuries, road or work	fatal injuries-eq.	**2.0E-07**	2.5E-07	2.2-07	*2.5E-07*	2.2E-07
Ionizing radiation	Bq C-14-eq.	*71*	-1,190	-892	-1,190	-1,150
Mineral extraction	MJ extra	*1*	-62	-42	-62	**-63**
Nature occupation	m^2 arable land	*8*	-10	-5	*-10*	**-12**
Non-renewable energy	MJ primary	*367*	-9,280	-6,520	-9,290	**-11,300**
Ozone layer depletion	kg CFC-11-eq.	*1.3E-08*	-2.1E-07	-1.5E-07	**-2.3E-07**	-2.2E-07
Photochemical ozone – Veg.	m^2*ppm*hours	*8,900*	-2,500	-71	-2,510	**-3,160**
Respiratory inorganics	kg PM2.5-eq.	*0.04*	-0.43	-0.34	-0.44	**-0.55**
Respiratory organics	pers*ppm*hours	*1.09*	-0.25	0.04	-0.25	**-0.30**

The best results are **bolded,** the worst are italic.

Table 46 Midpoint indicator results for groundwater emissions in the Malta scenarios, when applying the same characterisation factors as for emissions to surface water.

Impact category	Unit	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
		per 1 Mg waste (Malta)				
Ecotoxicity, aquatic, groundw.	kg-eq. TEG water	*4.0E+05*	-7.2E+04	6.4E+04	**-7.3E+04**	1.1E+05
Ecotox., aquatic, long-term gr.	kg TEG water	**2.2E+11**	3.5E+11	*4.4E+11*	3.5E+11	4.3E+11
Eutrophic., aquatic, groundw.	kg NO_3-eq.	*7.0E+03*	2.5E+02	3.6E+03	**2.4E+02**	4.1E+03
Eutrophic., aquatic, long-term	kg NO_3-eq.	*1.1E+07*	7.3E+05	6.5E+06	7.2E+05	**2.1E+05**
Human toxicity, groundwater	kg C_2H_3Cl-eq.	4.2E+03	1.9E+04	**3.5E+03**	1.9E+04	1.0E+04
Human toxicity, long-term gr.	kg C_2H_3Cl-eq.	*6.7E+07*	2.6E+07	4.4E+07	2.6E+07	**1.3E+07**

The best results are **bolded,** the worst are italic.

Table 47 Midpoint indicator results for the Malta scenarios, with reduced factors for groundwater emissions, relative to the annual emissions per person in Europe.

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Malta)				
Acidification	*4.1E-04*	-1.0E-02	-9.5E-03	-1.1E-02	**-1.6E-02**
Ecotoxicity, aquatic	*2.6E-02*	-1.1E-01	-6.2E-02	-1.1E-01	**-1.1E-01**
Ecotoxicity, aquatic, groundw.	*9.1E-08*	-1.6E-08	1.4E-08	**-1.7E-08**	2.4E-08
Ecotox., aquatic, long-term gr.	**1.1E-07**	1.8E-07	*2.2E-07*	1.8E-07	2.2E-07
Ecotoxicity, terrestrial	*2.7E-02*	-2.5E-01	-2.1E-01	-2.6E-01	**-2.9E-01**
Eutrophication, aquatic	3.2E-03	-1.3E-03	*6.4E-02*	**-1.3E-03**	8.8E-03
Eutrophic., aquatic, groundw.	*1.9E-05*	6.8E-07	1.0E-05	**6.7E-07**	1.1E-05
Eutrophic., aquatic, long-term	*6.7E-08*	4.5E-09	4.0E-08	4.5E-09	**1.3E-09**
Eutrophication, terrestrial	*5.5E-04*	-1.3E-02	-1.1E-02	-1.3E-02	**-1.8E-02**
Climate change	*7.5E-02*	1.6E-02	9.1E-03	1.6E-02	**3.2E-04**
Human toxicity	*1.3E-03*	-1.5E-02	-1.5E-02	-1.5E-02	**-2.3E-02**
Human toxicity, groundwater	2.0E-06	9.0E-06	**1.7E-06**	9.0E-06	4.8E-06
Human toxicity, long-term gr.	*7.2E-08*	2.9E-08	4.8E-08	2.8E-08	**1.4E-08**
Injuries, road or work	**1.4E-03**	1.8E-03	1.5E-03	1.8E-03	1.6E-03

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Malta)				
Ionizing radiation	*1.3E-04*	-2.2E-03	-1.7E-03	-2.2E-03	-2.2E-03
Mineral extraction	*4.2E-03*	-2.1E-01	-1.4E-01	-2.1E-01	**-2.1E-01**
Nature occupation	*2.8E-03*	-3.3E-03	-1.6E-03	-3.3E-03	**-4.1E-03**
Non-renewable energy	*2.4E-03*	-6.1E-02	-4.3E-02	-6.1E-02	**-7.5E-02**
Ozone layer depletion	*6.2E-08*	-1.0E-06	-7.2E-07	**-1.1E-06**	-1.1E-06
Photochemical ozone – Veg.	*6.3E-02*	-1.8E-02	-5.1E-04	-1.8E-02	**-2.2E-02**
Respiratory inorganics	*5.0E-03*	-4.9E-02	-3.9E-02	-5.0E-02	**-6.2E-02**
Respiratory organics	*1.1E-01*	-2.5E-02	4.1E-03	-2.5E-02	**-3.0E-02**

The best results are **bolded,** the worst are italic.

Table 48 **Midpoint indicator results for groundwater emissions in the Malta scenarios, relative to the annual emissions per person in Europe (see Table 18), when applying the same characterisation factors as for emissions to surface water.**

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Malta)				
Ecotoxicity, aquatic, groundw.	*2.3E-04*	-4.1E-05	3.6E-05	**-4.1E-05**	6.0E-05
Ecotox., aquatic, long-term gr.	**1.9E-01**	2.9E-01	*3.7E-01*	3.0E-01	3.7E-01
Eutrophic., aquatic, groundw.	*4.8E-02*	1.7E-03	2.5E-02	1.7E-03	2.8E-02
Eutrophic., aquatic, long-term	*1.1E-01*	7.5E-03	6.7E-02	7.4E-03	**2.1E-03**
Human toxicity, groundwater	5.0E-03	2.3E-02	**4.2E-03**	2.3E-02	1.2E-02
Human toxicity, long-term gr.	*1.2E-01*	4.8E-02	8.0E-02	4.6E-02	**2.4E-02**

The best results are **bolded,** the worst are italic.

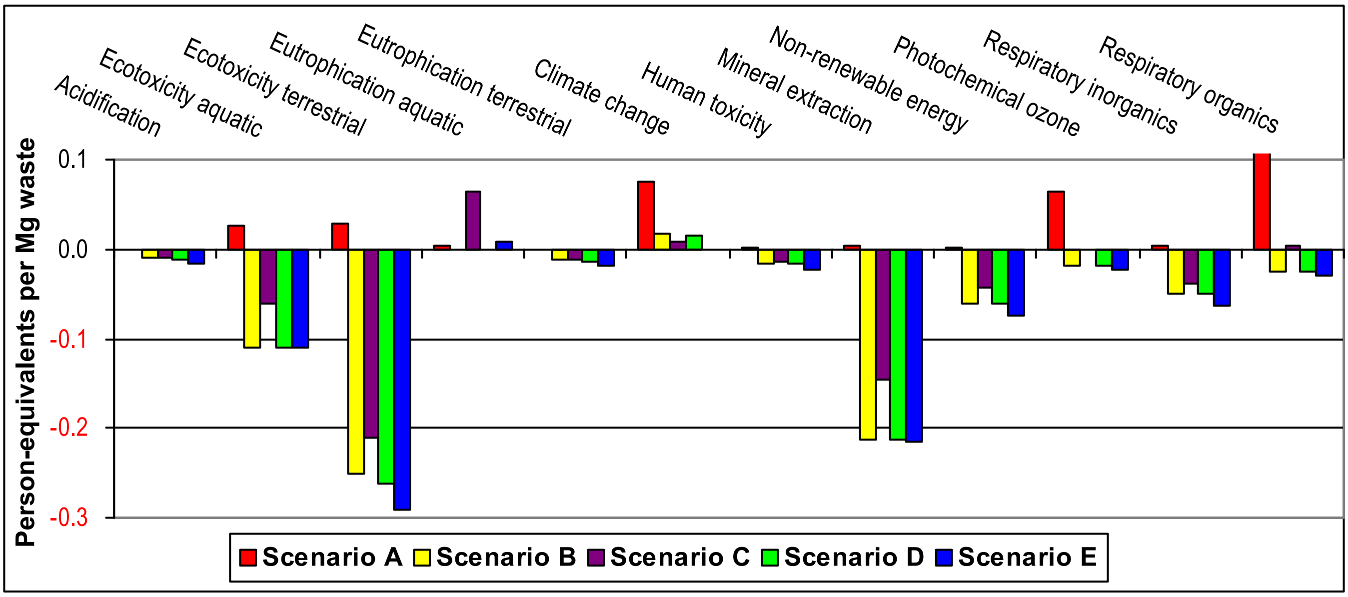

Figure 12 **Midpoint indicator results for the Malta scenarios, with reduced factors for groundwater emissions, relative to the annual emissions per person in Europe. Negative numbers reflect net displaced impacts; positive numbers reflect net environmental impacts.**

The results for Malta are quite similar to the results for Krakow (Chapter 9.1). For most impact categories there is an ordering of the scenarios from best to worst: E > D > B > C > A, although the differences between the primarily incineration/recycling-based scenarios B and D are very small. Nevertheless, the differences - even for scenarios B and D - are significant for most impact categories (Chapter 9.5). The deviations from the general ordering are discussed below.

Similar to the results for Krakow, the composting scenario C performs on average slightly better for climate change than the incineration scenario B. For Malta, scenario C even performs on average better than the economic-optimum scenario D. The reason is the same, namely that composting gives on average in this study a better energy utilisation of wet biodegradable wastes than incineration. With the high percentage of wet biodegradable wastes on Malta (compared to Krakow), this dominates the results for climate change when comparing the “compost scenarios” (C, E) with the “incineration scenarios” (B, D).

For aquatic eutrophication, the composting scenario (C) performs significantly worse than the landfilling scenario (A). This is mainly explained by much of the remaining wastes in the composting scenario still going to landfill, and while the landfill in scenario A is an uncontrolled landfill, where all the leachate goes to groundwater and no leachate is released to surface waters, the landfill in scenario C is a directive compliant landfill, where the leachate is captured and, in spite of waste-water treatment, a significant part of the nutrients are eventually released to surface waters contributing to the impact category “Aquatic eutrophication” (without the reduced characterisation factors for emissions to groundwater, scenario A would perform worse than scenario C for aquatic eutrophication, see Table 48).

Contributing to the higher aquatic eutrophication for scenario C is also the ammonia emission from the composting process, which is another reason for scenario E to perform significantly worse than scenarios B and D. The reason for not seeing a similar effect for the Krakow scenarios is the larger share of biodegradable wastes in Malta. However, as noted in Chapter 6.2.2 (Note [4] to Table 19), nutrient emissions from Malta to the Mediterranean Sea were not considered here likely to lead to eutrophication in practice.

9.3 Endpoint indicator results for the Krakow scenarios.

The results are presented here in terms of the endpoint indicator results converted into Euro and combined.

9.3.1 Environmental assessment for the Krakow scenarios

Table 49 and Figure 13 provide the endpoint indicator results per impact category. Unlike midpoint results, these indicators can be compared across impact categories.

Compared to the midpoint indicator results in Chapter 9.1, three impact categories are left out:

- Aquatic eutrophication, for which no endpoint characterisation factors are available,
- Non-renewable energy, for which the endpoint characterisation factor here is zero,
- Ozone depletion, for which impacts for all scenarios are below 0.005 Euro per Mg waste.

Furthermore, the separate impact categories for emissions to groundwater are not included in Table 49, since these impacts were all below 0.005 Euro when the reduction factors were applied.

Table 50 presents the endpoint results for these impact categories without reduction factors.

Table 49 **Endpoint indicator results for the Krakow scenarios, with reduced factors for groundwater emissions. All values in Euro$_{2003}$.**

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Krakow)				
Acidification	-0.12	-0.83	-0.57	-0.89	-1.09
Ecotoxicity, aquatic	0.03	-0.83	-0.52	-0.85	-0.80
Ecotoxicity, terrestrial	-2.26	-10.30	-8.93	-11.50	-11.60
Eutrophication, terrestrial	-0.06	-1.25	-0.70	-1.36	-1.48
Climate change	103.00	-6.27	-9.72	-15.20	-37.10
Human toxicity	-0.25	-3.95	-4.94	-5.60	-5.76
Injuries, road or work	0.93	0.89	0.71	0.83	0.99
Ionizing radiation	-0.01	-0.08	-0.07	-0.08	-0.08
Mineral extraction	-0.07	-0.35	-0.29	-0.35	-0.35
Nature occupation	2.76	-11.50	-7.80	-14.30	-13.50
Photochemical ozone – Veg.	2.67	-1.89	-0.41	-1.98	-2.40
Respiratory inorganics	-8.28	-51.30	-38.40	-54.70	-63.60
Respiratory organics	0.16	-0.11	-0.01	-0.11	-0.12
Total	**98.50**	**-87.77**	**-71.65**	**-106.09**	**-136.89**

Table 50 **Endpoint indicator results for groundwater emissions in the Krakow scenarios, when applying the same characterisation factors as for emissions to surface water. All values in Euro$_{2003}$.**

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste Krakow				
Ecotoxicity, aquatic, groundw.	0.00	-0.01	0.00	-0.01	0.00
Ecotox., aquatic, long-term gr.	2.87	6.03	7.13	6.92	7.70
Human toxicity, groundwater	0.17	3.38	0.46	3.18	2.12
Human toxicity, long-term gr.	54.33	11.42	45.33	10.15	7.68

The results have an ordering of the scenarios from best to worst: E > D > B > C > A. This was similar to the midpoint indicator results for most, although not all, impact categories. However, the difference between scenarios B and C is not statistically significant, nor is the difference between scenario E and scenarios B and D (see Chapter 9.5). Explanations for the causes of the ordering have already been given in Chapter 6.1, as the results here differ only by the different factors used to convert from midpoint to endpoint indicators to facilitate cross-comparison.

If the results for groundwater emissions without reduction factors (from

Table 50) were added to the results from Table 49, this would only enhance the differences between the scenarios and emphasise the E > D > B > C > A ordering. The same is true if a reduction factor for terrestrial ecotoxicity would not be applied.

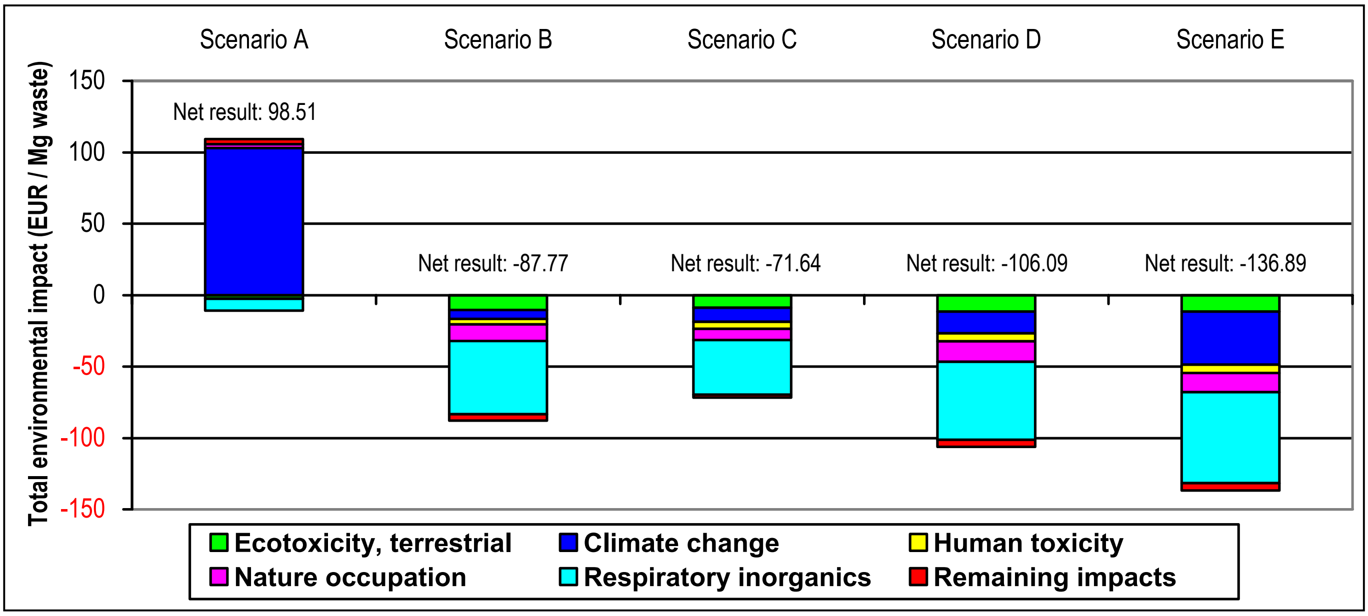

Figure 13 **Endpoint results for the Krakow scenarios, with reduced factors for groundwater emissions.**
To obtain the total value for a column, the negative values should be subtracted from the positive.

9.3.2 Cost-benefit assessment for the Krakow scenarios

Table 51 reports the life cycle costs resulting from the costing assessment outlined in Chapter 7.

Table 51 **Life cycle costs for the Krakow scenarios. All values in Euro$_{2003}$.**

Activity	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Krakow)				
Waste container, household	3.26	5.08	5.08	5.08	5.08
Waste collection, kerb-side	60.80	76.00	76.00	76.00	76.00
Waste container, bring cube	0.01	0.05	0.05	0.06	0.07
Additional effort for recyclates	1.56	6.83	6.80	7.99	9.61

Activity	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Krakow)				
Additional effort for separate biowastes	0.47		5.56		5.56
Waste incineration	0.11	30.80	0.04	28.70	14.50
Landfill	46.30		26.30		
Composting	0.33		11.70		11.70
Additional transport to incineration					4.50
Recycling - Iron and steel	-0.17	-0.78	-0.49	-0.78	-0.78
Recycling - Glass	-1.20	-1.86	-1.74	-2.46	-2.46
Recycling - PE	-0.28	0.00	-0.26	0.00	-12.00
Recycling - Other plastics		-6.83	-6.83	-6.83	-6.83
Recycling - Corrugated board	-0.55	-6.19	-6.19	-6.19	-6.19
Recycling - Newsprint	-0.51	-5.27	-6.14	-7.79	-7.78
Recycling - Aluminium	-1.42	-7.27	-7.27	-7.27	-7.27
Net electricity recovered	-5.63	-44.10	-11.30	-42.00	-32.40
Total costs	**103.08**	**46.46**	**91.31**	**44.51**	**51.31**

It is worth noting that also from a cost perspective, the order of the four first scenarios from best to worst is D > B > C > A, i.e. the same order as resulting from the environmental assessment, with the economic optimum scenario as the most environmentally preferable. The societal optimum scenario E cannot be more favourable than the economic optimum scenario, until adding the environmental externalities (the monetary value of the environmental impacts) from Table 49, as done in Table 52.

Table 52 Cost-benefit results for the Krakow scenarios. All values in Euro$_{2003}$.

Activity	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Krakow)				
Life cycle costs (from Table 9.3.3)	103.1	46.5	91.3	44.5	51.3
Environmental costs (externalities)	98.0	-87.7	-72.0	-106.0	-137.0
Societal costs (sum of the above)	**201.1**	**-41.2**	**19.7**	**-61.5**	**-85.7**

Figure 14 shows the cost-benefit results in graphical format.

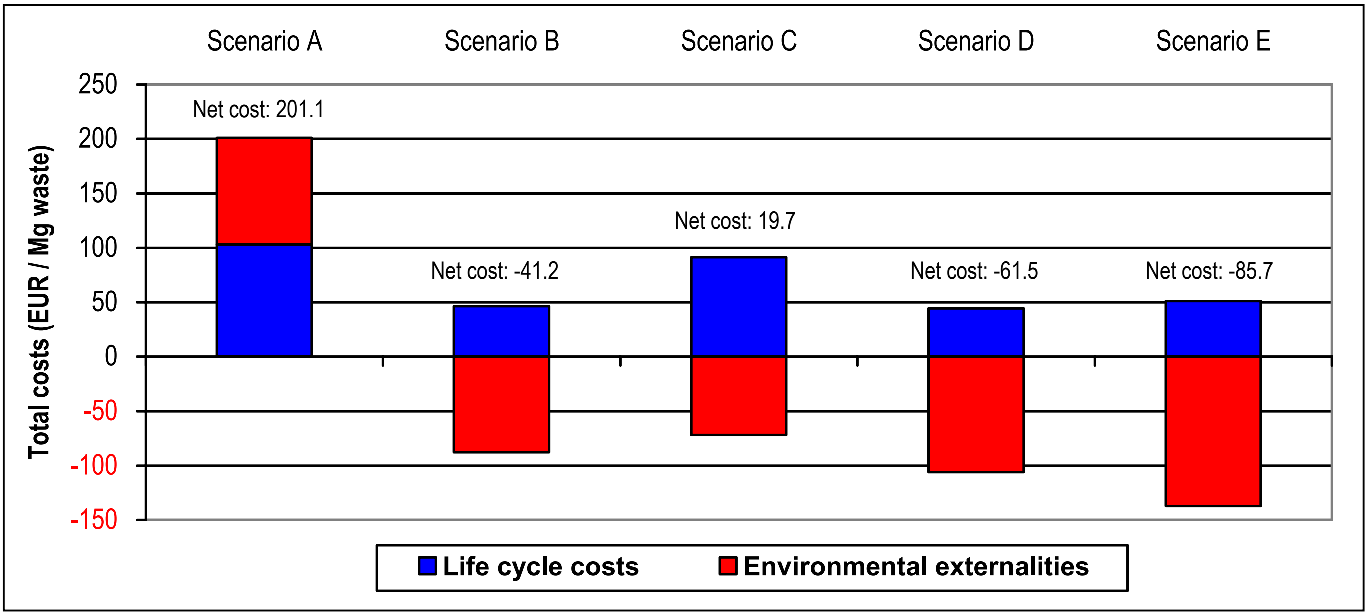

Figure 14 **Cost-benefit results for the Krakow scenarios, in Euro$_{2003}$ per Mg waste. To obtain the total value for a column, the benefits should be subtracted from the costs. A negative cost is a benefit.**

The cost-benefit results show again an ordering of the scenarios from best to worst: E > D > B > C > A. However, the differences between scenario E and scenarios D and B are not significant.

9.4 Endpoint indicator results for the Malta scenarios

The results are presented here in terms of the endpoint indicator results converted into Euro and combined.

9.4.1 Environmental assessment for the Malta scenarios

Table 53 and Figure 15 provide the endpoint indicator results per impact category. Unlike the midpoint results, these can also be combined and compared across impact categories.

The separate impact categories for emissions to groundwater are not included in Table 53, since these impacts were all below 0.005 Euro when the reduction factors were applied.

Table 54 presents the endpoint results for these impact categories without reduction factors.

Table 53 Endpoint indicator results for the Malta scenarios, with reduced factors for groundwater emissions. All values in Euro$_{2003}$.

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Malta)				
Acidification	0.02	-0.44	-0.40	-0.45	-0.67
Ecotoxicity, aquatic	0.32	-1.35	-0.77	-1.35	-1.38
Ecotoxicity, terrestrial	0.51	-4.65	-3.91	-4.86	-5.39
Eutrophication, terrestrial	0.04	-0.83	-0.73	-0.84	-1.20
Climate change	165.00	35.90	20.00	34.90	0.69
Human toxicity	0.23	-2.64	-2.56	-2.64	-3.96
Injuries, road or work	0.84	1.06	0.90	1.06	0.93
Ionizing radiation	0.00	-0.02	-0.02	-0.02	-0.02
Mineral extraction	0.00	-0.25	-0.17	-0.25	-0.25
Nature occupation	4.84	-5.68	-2.73	-5.69	-7.09
Photochemical ozone – Veg.	4.52	-1.27	-0.04	-1.28	-1.60
Respiratory inorganics	2.98	-29.40	-23.40	-29.90	-37.20
Respiratory organics	0.28	-0.07	0.01	-0.07	-0.08
Totals	**179.58**	**-9.64**	**-13.82**	**-11.39**	**-57.22**

Table 54 Endpoint indicator results for groundwater emissions in the Malta scenarios, when applying the same characterisation factors as for emissions to surface water. All values in Euro$_{2003}$.

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Malta)				
Ecotoxicity, aquatic, groundw.	0.00	0.00	0.00	0.00	0.00
Ecotox., aquatic, long-term gr.	2.35	3.67	4.65	3.72	4.57
Human toxicity, groundwater	0.87	3.90	0.72	3.90	2.08
Human toxicity, long-term gr.	20.83	8.27	13.82	8.05	4.10

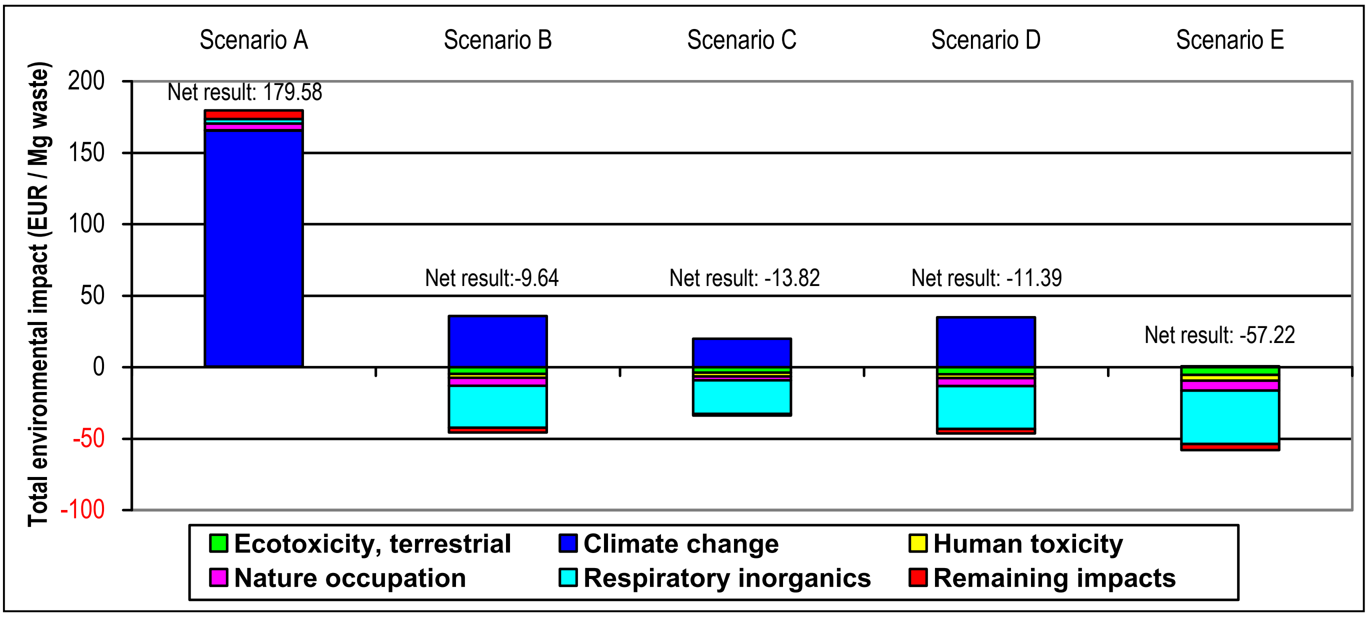

Figure 15 **Endpoint indicator results for the Malta scenarios, with reduced factors for groundwater emissions.**
To obtain the total value for a column, the negative values should be subtracted from the positive.

The results show again that scenario E performs clearly better and scenario A clearly worse than the others. The better energy utilisation of wet biodegradable wastes in this study in the composting scenario C reduces the climate change impact, which outweighs the positive impacts from the larger recycling in scenarios B and D. This implies here that composting-based management would be preferable to incineration in this case with high fractions of biodegradable wastes. However, the differences between scenario C and scenarios B and D are not significant.

If the results for groundwater emissions without reduction factors (from

Table 54) were added to the results from Table 53, this would not change the relative position of the five scenarios. The same is true if a reduction factor for terrestrial ecotoxicity would not be applied.

9.4.2 Cost-benefit assessment for the Malta scenarios

Table 55 presents the life cycle costs resulting from the cost assessment in Chapter 7.

Table 55 Life cycle costs for the Malta scenarios. All values in Euro$_{2003}$.

Activity	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Malta)				
Waste container, household	3.77	3.77	3.77	3.77	3.77
Waste collection, kerb-side	76.00	76.00	76.00	76.00	76.00
Waste container, bring cube		0.05	0.05	0.05	0.06
Additional effort for recyclates		8.31	8.37	8.56	11.70
Additional effort for separate biowastes	3.05		9.19		10.10
Waste incineration		35.60	0.04	35.40	11.60
Landfill	27.20		22.40		
Composting	2.89		19.30		21.30
Additional transport to incineration					8.00
Recycling - Iron and steel		-1.10	-0.69	-1.10	-1.10
Recycling – Glass		-0.74	-0.62	-0.88	-0.88
Recycling – PE		0.00	-2.80	0.00	-8.24
Recycling - Other plastics		-4.67	-4.67	-4.67	-4.67
Recycling - Corrugated board		-3.39	-3.39	-3.39	-3.39
Recycling – Newsprint		-2.84	-2.84	-2.84	-4.26
Recycling – Aluminium		-1.36	-1.36	-1.36	-1.36
Net electricity recovered	0.10	-33.60	-16.60	-33.60	-30.50
Total costs	**113.01**	**76.03**	**106.15**	**75.94**	**88.13**

From a cost perspective, there is no noteworthy difference between the incineration-based scenarios B and D, while the bio-composting scenario C is here distinctly more costly. This is caused by a combination of lower income from electricity generation and increased costs for treatment and separate biowaste collection. Equally, some waste is sent to landfill in scenario C unlike scenarios B and D.

The fifth scenario E cannot be more favourable than the economic optimum scenario, unless adding the environmental externalities from Table 53, as done in Table 56. Figure 16 shows the cost-benefit results in graphical format.

Table 56 Cost-benefit results for the Malta scenarios. All values in Euro$_{2003}$.

Activity	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	per 1 Mg waste (Malta)				
Life cycle costs (from Table 9.4.3)	113	76	106	76	88
Environmental costs (externalities)	180	-10	-14	-11	-57
Societal costs (sum of the above)	**293**	**66**	**92**	**65**	**31**

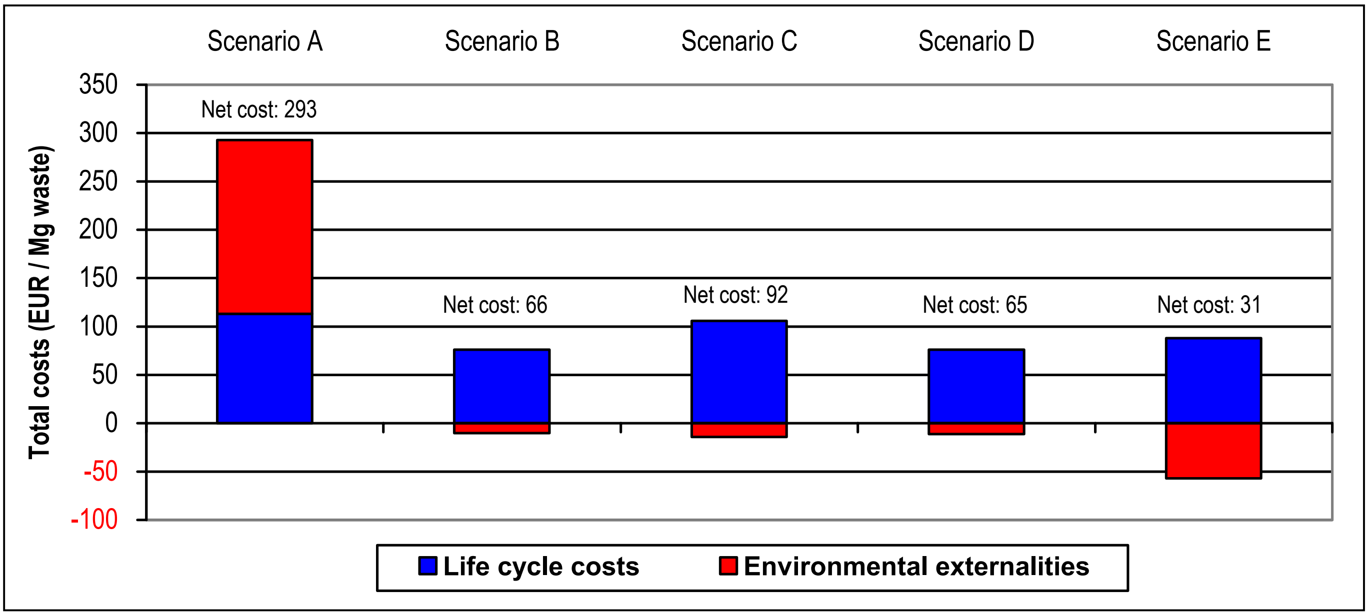

Figure 16 Cost-benefit results for the Malta scenarios; in Euro2003 per Mg waste. To obtain the total value for a column, the benefits should be subtracted from the costs. A negative cost is a benefit.

The cost-benefit results show a better performance of scenario E over the other scenarios, as this is the optimum in this context. The difference to scenarios B and D is noticeable, similarly to the difference from scenarios B and D to scenario C. Scenario A clearly has the worst performance.

9.5 Uncertainty of the results

Unless otherwise stated, statistical significance is here relative to the 95th percentile confidence interval assuming a log-normal distribution.

9.5.1 Inventory uncertainty

For the endpoint comparisons of the five scenarios for Krakow and Malta, respectively, Monte Carlo simulations were conducted with 1000 iterations, taking into account data uncertainty in the inventory. The uncertainty analysis on the impact assessment is separately reported in Chapter 9.5.2.

It should be noted that most uncertainties in the inventory data are not measured, but estimated, and reflect only uncertainties of data for the included processes (i.e. system incompleteness is not reflected). This kind of analysis does not include e.g. the consideration of missing inventory data, but only addresses the certainty of given data.

As the different waste fractions are treated in the same plant and the many processes appear in more than one scenario, there can be important covariations between the processes and scenarios. To avoid this, an approach of coupled sampling was adopted, i.e. the same process is sampled only once in each iteration. When adequate to show significant difference at the 95% confidence level, the coupled sampling was limited to the covariation between scenarios and the most important uncertainties.

Monte Carlo simulations performed without applying the reduction factors for groundwater emissions and terrestrial ecotoxicity, show that these impact categories dominate the overall uncertainty. When the reduction factors for these impact categories are applied, other impact categories obtain more weight in the overall result, and the largest uncertainties are now found for climate change, which typically explains more than 50% of the uncertainty of the overall result. This is mainly due to the uncertainty on the energy efficiency of the incineration and composting plants, i.e. how much CO_2 emission is displaced by these waste treatment options.

The results of the Monte Carlo simulations show that for both Krakow and Malta:

- All scenarios B to E have significantly smaller overall environmental impact and total societal cost than the baseline scenario A at the 97% confidence level.
- The societal optimum scenario E has significantly smaller overall environmental impact and total societal cost than the composting scenario C at 95% confidence level.
- The economic optimum scenario D has significantly smaller overall environmental impact and total societal cost than the incineration scenario B at the 99% confidence level. (These two scenarios are very similar in structure; scenario D has a little more recycling and a little less incineration).
- However, the incineration scenario B does not have significant difference in overall environmental impact or total societal cost from that of the composting scenario C (when significance is understood as a difference that is significant at the 95% confidence level, i.e. that more than 97.5% of the iterations show dominance of one scenario over the other).

The Monte Carlo simulations (performed at the 95% confidence level) furthermore show that:

- For Krakow:
    - The economic optimum scenario D has significantly smaller overall environmental impact and total societal cost than the composting scenario C.
    - The societal optimum scenario E does not have significant difference in overall environmental impact or total societal cost compared to that of the scenarios B and D.
- For Malta:
    - The societal optimum scenario E has significantly smaller overall environmental impact and total societal cost than scenarios B and D.
    - The economic optimum scenario D does not have significant difference in overall environmental impact or total societal cost when compared to the composting scenario C.

It may be questioned whether the 95% confidence level is the relevant confidence level for policy decisions. For example, for total environmental impacts, the difference between Krakow scenarios B and C is significant at the 80% confidence level, which means that in 90% of the iterations, scenario B has lower environmental impact than scenario C, and only in 10% of the iterations, the opposite is the case (the other 10% outside the 80% confidence interval is at the other side of the confidence interval, and thus shows a large difference between the scenarios). It can be argued that for practical decision-making, a 9:1 chance of a positive outcome may be sufficient.

Similarly, the following differences are significant at the 90% confidence level (i.e. at least 95% of the iterations show that one scenario has lower environmental impacts than the other):

- Total societal cost for Krakow scenarios B and C
- Total environmental impact for Krakow scenarios E and B.

The fact that the uncertainty is dominated by the CO_2-emissions holds promise for reducing the uncertainty through use of more specific information, especially on the energy conversion efficiencies of the different waste treatment technologies.

9.5.2 Impact assessment uncertainty

Since the endpoint indicator results are dominated by a few impact categories, notably climate change and respiratory inorganic effects on human health, it is also likely the uncertainty on these impact categories (see Chapter 6.3) that will dominate the overall impact uncertainty in this assessment.

Since it is to a large extent the same substances (CO_2 and particles) that contribute to the overall result for these impact categories in all of the five scenarios, the impact assessment uncertainty does not affect the significance of the relative ranking of the scenarios. The largest additional contribution of uncertainty is from the emission of methane, which is particularly large in the scenarios with landfilling (scenarios A and C). However, since these scenarios are typically the ones with the highest environmental impacts and total costs, the additional uncertainty from the characterisation factor of methane does not affect the conclusions. The overall size of the endpoint results is more likely to be underestimated than overestimated, due to the conservative estimate on the climate change impact (see Chapter 6.2).

9.6 Comparing treatment options

This section compares the overall environmental impact and total societal costs of the relevant treatment options independently for each waste fraction using endpoint indicators. All data presented are calculated with reduced factors for groundwater emissions.

The comparison is made on the Krakow dataset and site-generic impact assessment, since this provides the most relevant results for other situations in Europe. However, since the site-dependent impact assessment has little importance here for the overall results (see Chapter 9.8), and most of the inventory data are generic, and therefore the same for Malta and Krakow, results made with the Malta dataset and site-dependent modelling will not give results significantly different.

The comparisons are made for 1 Mg of each specific waste fraction, e.g. 1 Mg of wet biodegradable waste, and thus *not* the quantity going into any specific waste scenario.

9.6.1 Comparing treatment options for the wet biodegradable fractions

Figure 17 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 in this study, and thus the lower net emission of greenhouse gases. Due to uncertainty on the energy conversion efficiencies, the difference to incineration is significant at the 90% confidence level, i.e. there is a 5% chance that incineration has lower environmental impacts in this study.

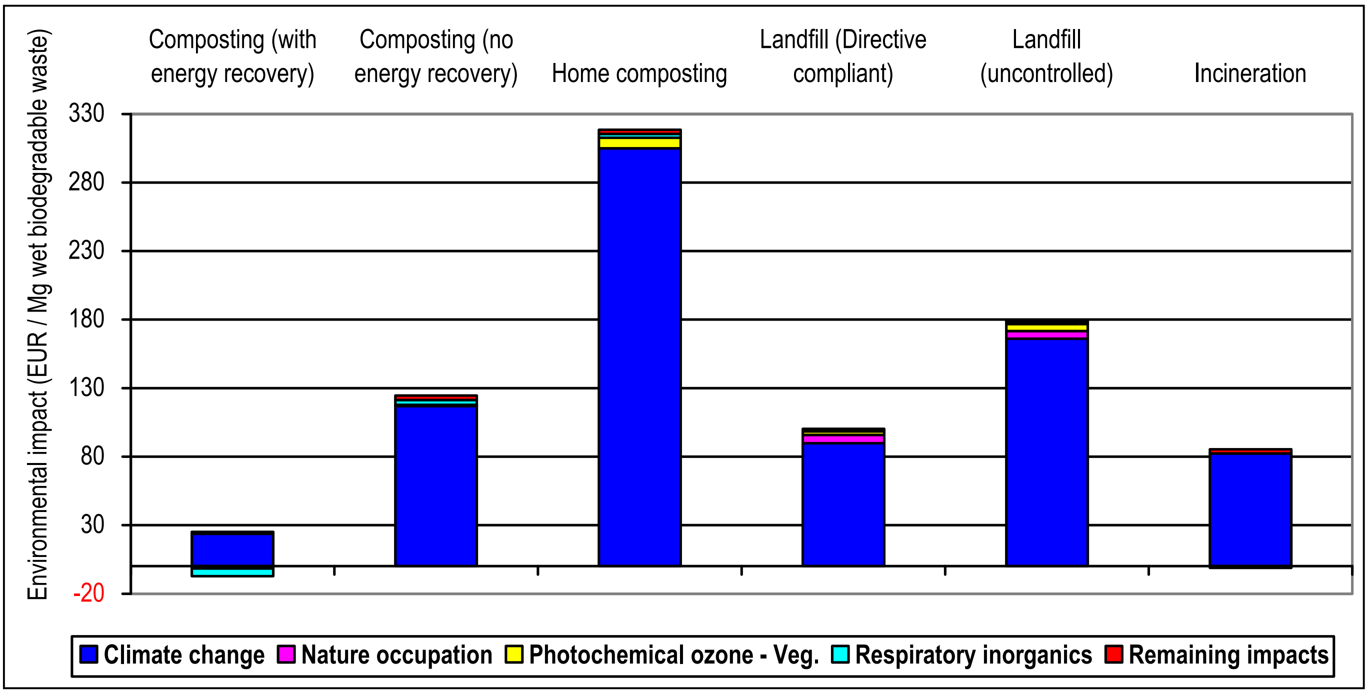

Figure 17 **Environmental impacts for different treatment options for the wet biodegradable fractions of municipal solid waste (waste collection not included). To obtain the total value for a column, the negative values should be subtracted from the positive.**

The home composting option turns out to be problematic, due to the assumption of partly anaerobic digestion, i.e. that some home composting 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% over the first 100 years), the resulting climate change dominates the result. In a directive compliant landfill, a larger amount of the methane is captured and combusted.

The result for incineration is also dominated by the climate change 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.

In the scenario comparisons for Malta and Krakow in Chapter 9.3 and 9.4, the composting scenarios C appeared to perform worse than the incineration scenarios B. This is mainly due to the influence of other fractions than wet biodegradable wastes, which are incinerated in the incineration scenarios, while being landfilled in the composting scenarios. Thus, the worse performance of the composting scenarios is not due to the performance of composting relative to incineration. This can also be seen by comparing the economic optimum scenarios D and the societal optimum scenarios E, for which one of the main differences is exactly that the wet biodegradable waste is composted in the societal optimum scenario, while being incinerated in the economic optimum scenario (see Chapter 8).

When including costs in the calculations, see Figure 18, composting with energy recovery still comes out the best solution, in spite of the higher costs of collection and treatment. However, the difference to incineration is now only significant at the 80% confidence level, i.e. there is a 10% chance that incineration has lower total societal costs.

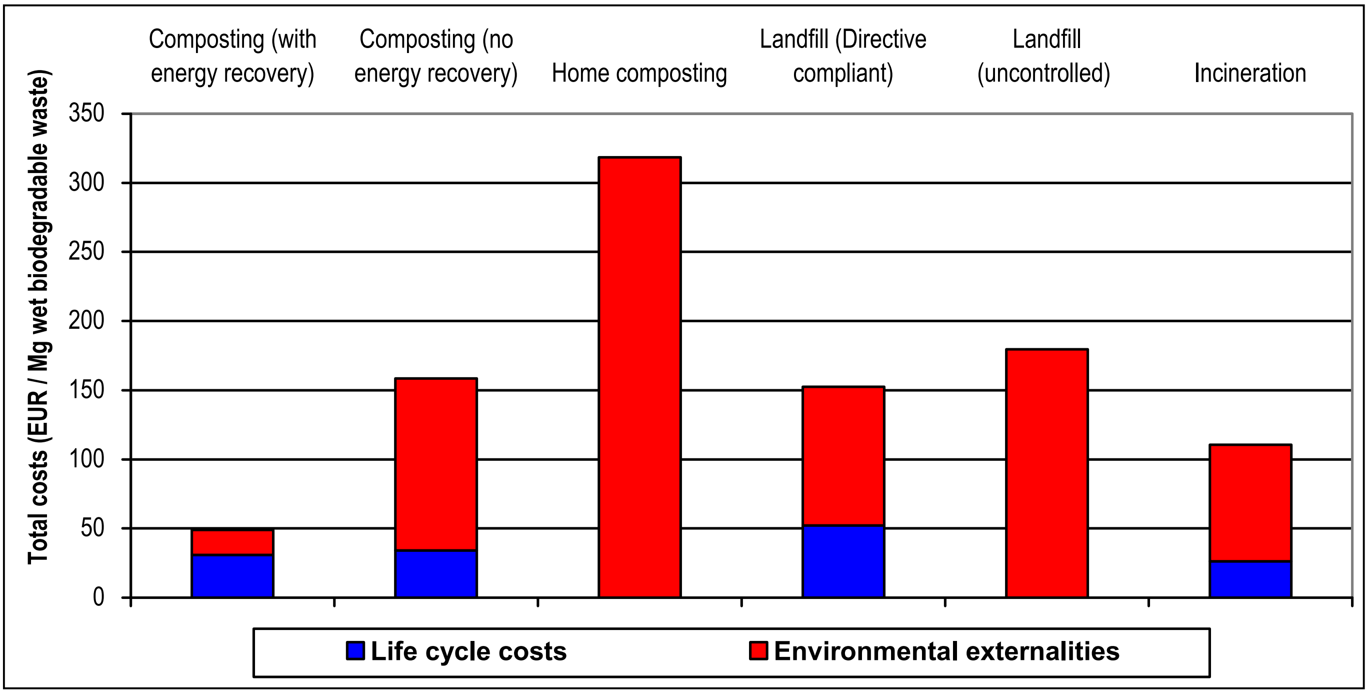

Figure 18 **Societal costs for different treatment options for the wet biodegradable fractions of MSW.**
Waste collection not included, but an additional cost of separate collection of 20 Euro is added to the two composting options.

9.6.2 Comparing treatment options for the paper and paperboard fractions

Figure 19 shows the environmental impacts for different treatment options for the paper and paperboard fractions of municipal solid waste. It shows lower environmental impact for recycling with incineration as the second-best option. Recycling of board gives more environmental benefit than recycling of newsprint, but this difference is not statistically significant.

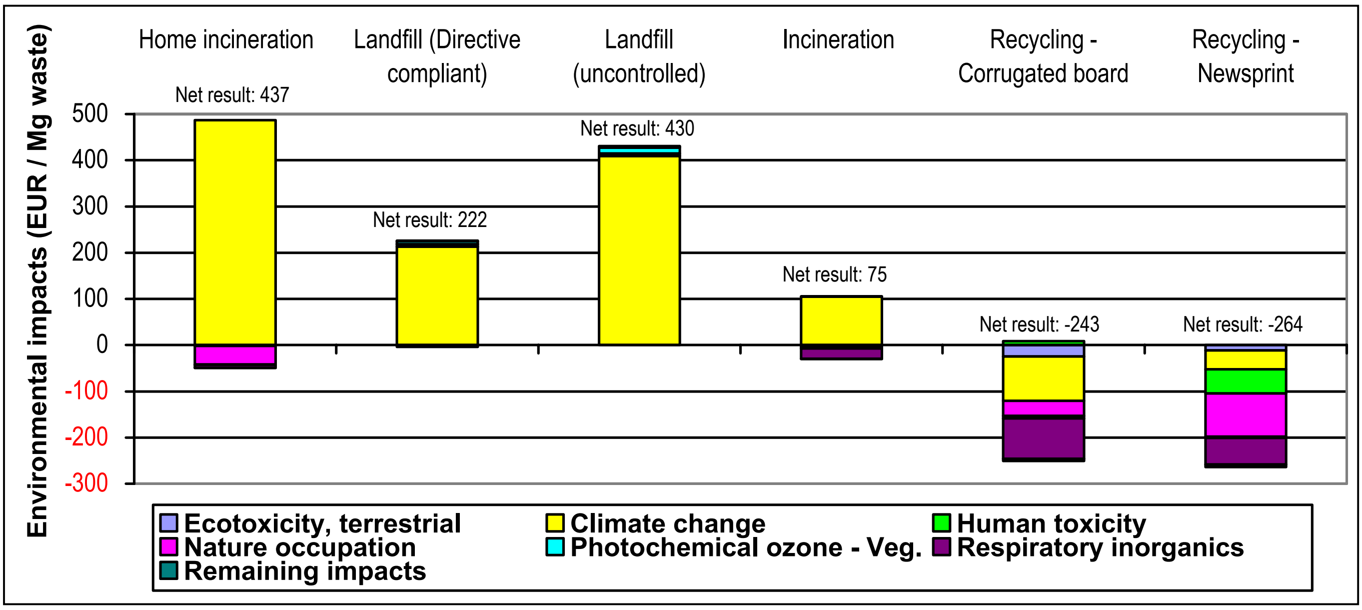

Figure 19 **Environmental impacts for different treatment options for the paper and paperboard fractions of municipal solid waste (waste collection not included). To obtain the total value for a column, the negative values should be subtracted from the positive.**

For both home-incineration and uncontrolled landfilling, the climate change impact dominates the results. No significant difference can be found between these two options. It should be noted that the home-incineration here is assumed to displace home incineration of wood with the same emissions. Home incineration is often associated with significant emissions of particulates and dioxins, due to low combustion temperatures. If the home incineration of paper and paperboard was assumed to take place under uncontrolled conditions, such as in a backyard barrel, then this would perform significantly worse than uncontrolled landfilling. Directive compliant landfilling has less emission of greenhouse gases, due to the methane capture.

The displaced ecotoxicity impact, which is visible in the columns for the recycling and municipal solid waste incineration options in Figure 19, is due to avoided emissions of heavy metals from the displaced paper production (in the recycling options) and from the displaced coal combustion for electricity (in the incineration option). When including costs in the calculations, see Figure 20, recycling still comes out the best solution, in spite of the higher costs of separate collection.

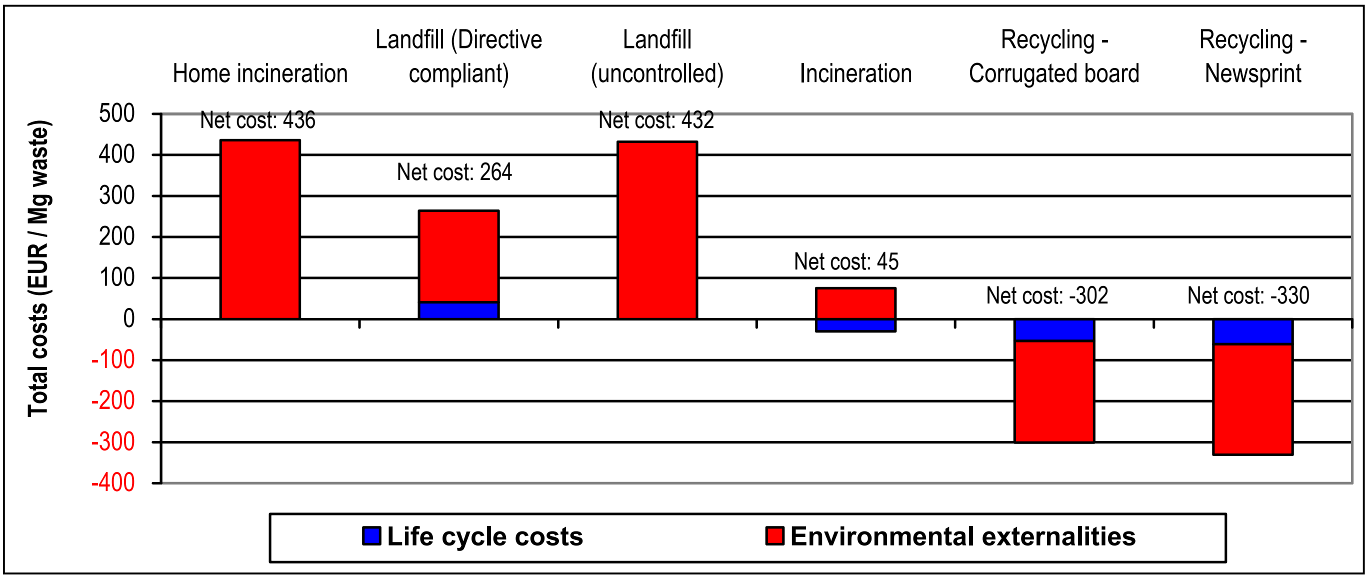

Figure 20 **Societal costs for different treatment options for the paper and paperboard fractions of MSW.**
Waste collection not included, but an additional cost of separate collection of 29 Euro is added to the two recycling options. A negative cost is a benefit. To obtain the total value for a column, the benefits should be subtracted from the costs.

9.6.3 Comparing treatment options for the plastics fractions

Figure 21 presents the environmental impacts for different treatment options for the PE (polyethylene) fractions of municipal solid waste. There are lower environmental impacts for recycling with incineration as the second-best option.

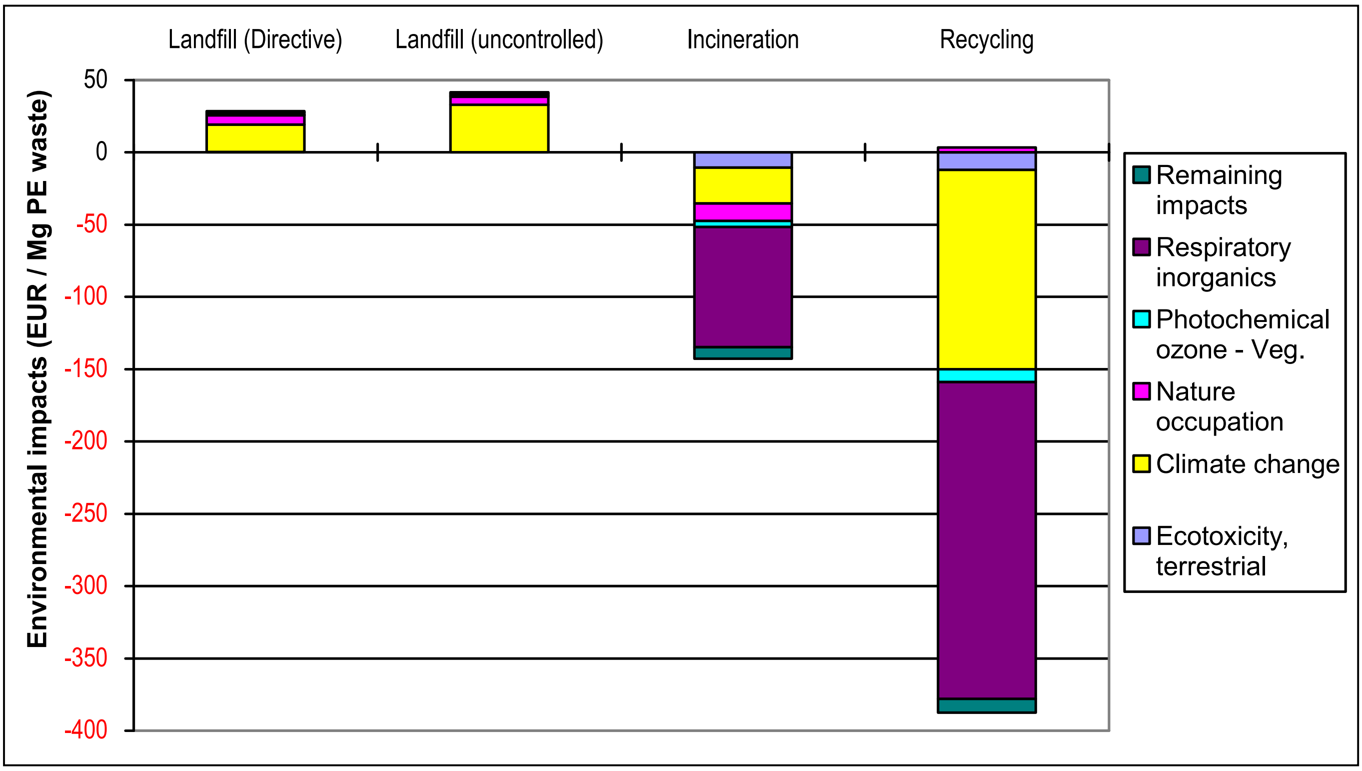

Figure 21 **Environmental impacts for different treatment options for the PE fractions of municipal solid waste (waste collection not included).**
To obtain the total value for a column, the negative values should be subtracted from the positive.

When including costs in the calculations, see Figure 22, recycling still comes out the best solution, in spite of the lower net economic gain from selling PE as waste materials compared to selling the energy from the material.

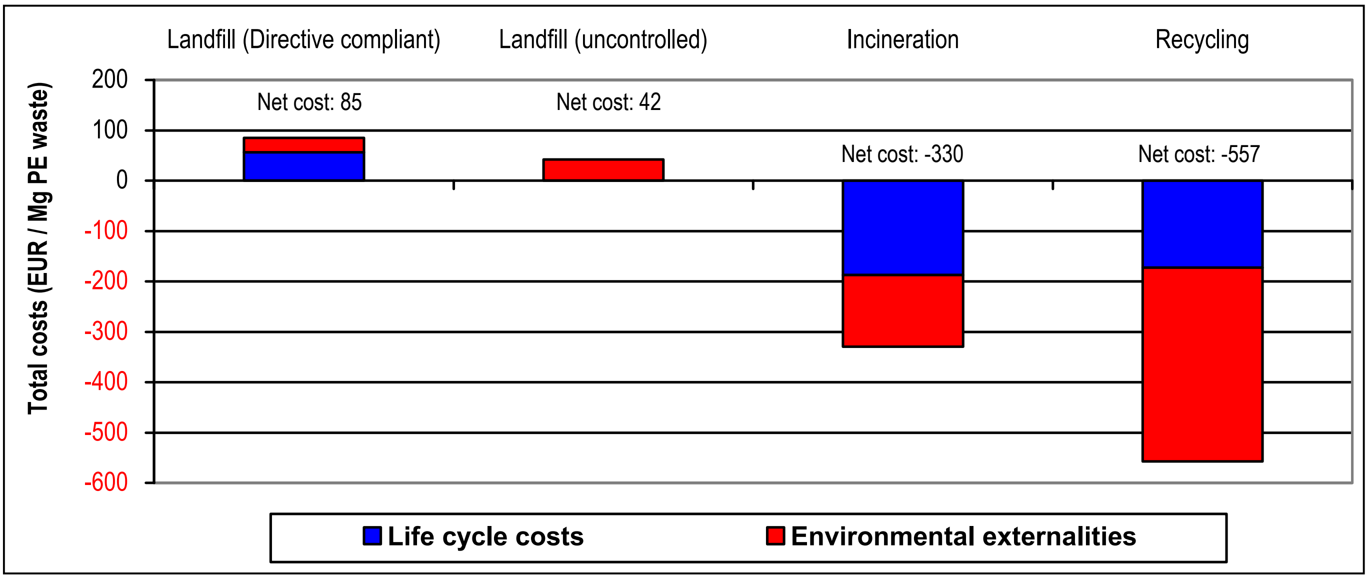

Figure 22 **Societal costs for different treatment options for the PE fractions of municipal solid waste.**
A negative cost is a benefit.
Waste collection not included, but an additional cost of separate collection of 29 Euro is added to the recycling option.

Since the landfilling option is clearly not interesting, the comparison for the other plastic types to a comparison between incineration and recycling is limited, see Figure 23 and Figure 24. These results support the lower environmental impact for the recycling options.

The differences between the plastic types in the results for recycling should be regarded with some caution, since the data for plastics production and recycling of plastics are considered to be of low quality.

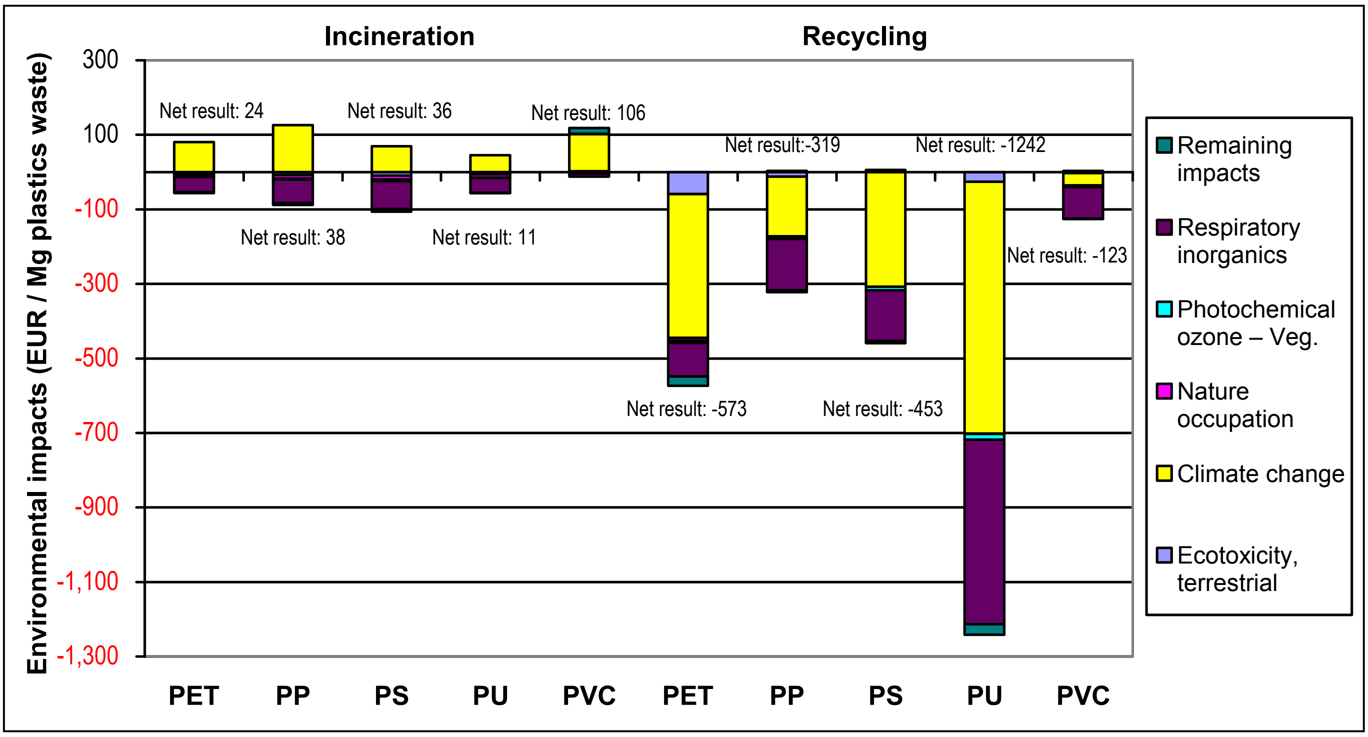

Figure 23 **Environmental impacts of incineration and recycling of different plastics fractions of MSW.**
PET = Polyethyleneterephthalate; PP = Polypropylene; PS = Polystyrene; PU = Polyurethane; PVC = Polyvinylchloride.
Waste collection not included.
To obtain the total value for a column, the negative values should be subtracted from the positive.

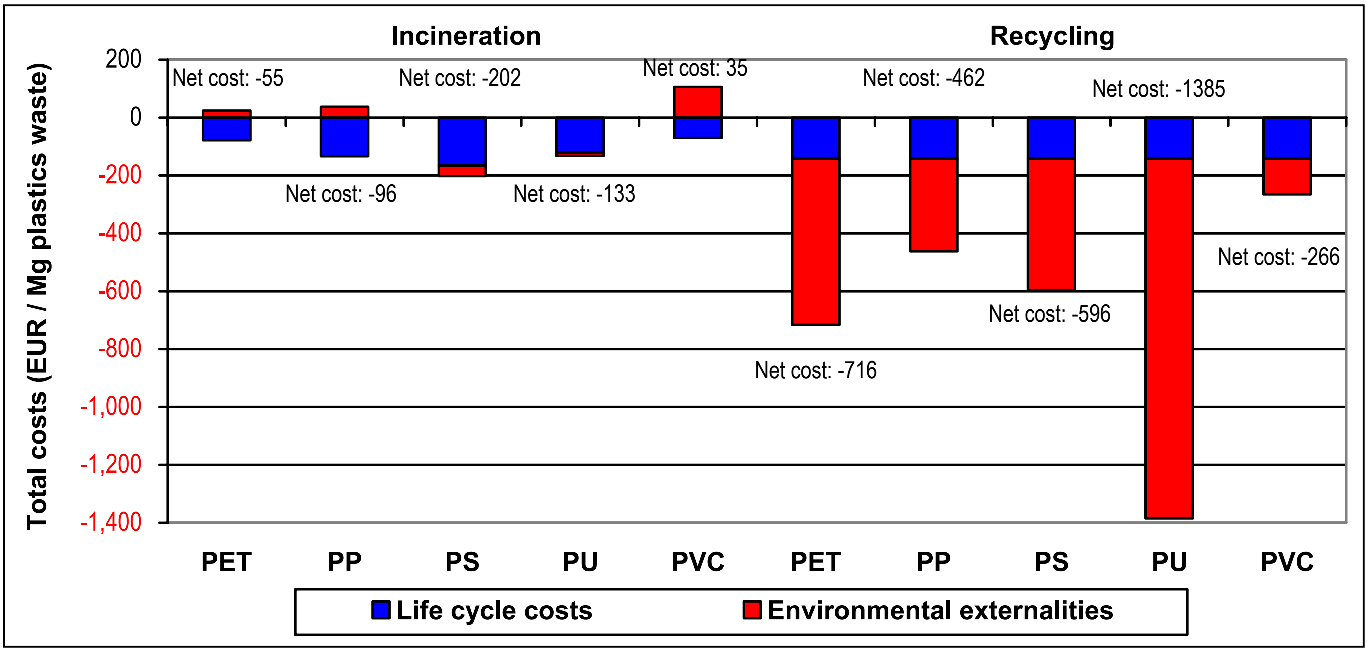

Figure 24 **Societal costs for incineration and recycling of the different plastics fractions of MSW.**
PET = Polyethyleneterephthalate; PP = Polypropylene; PS = Polystyrene; PU = Polyurethane; PVC = Polyvinylchloride.

Waste collection not included, but an additional cost of separate collection of 29 Euro is added to the recycling options.
The value of recycled PET (see Chapter 7) is used for all the recycled plastics in this figure.
A negative cost is a benefit.
To obtain the total value for a column, the benefits should be subtracted from the costs.

9.6.4 Comparing treatment options for the glass fractions

Figure 25 presents the environmental impacts for different treatment options for the glass fractions of municipal solid waste. Recycling is the best option, with landfilling as the second-best. Incineration of glass is obviously not a good idea, since the inert glass takes energy to heat, but produces no output of value.

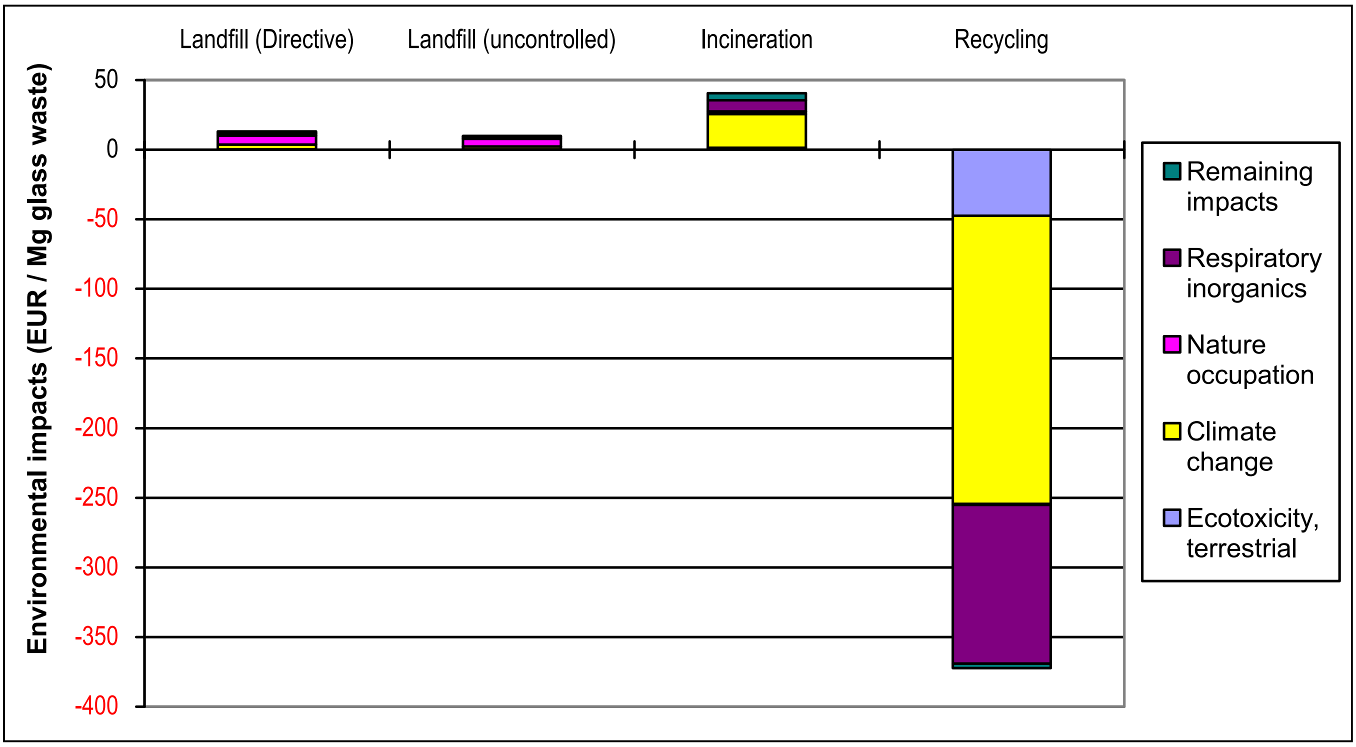

Figure 25 **Environmental impacts of different treatment options for the glass fractions of municipal solid waste (waste collection not included).**

Including costs in the calculations, see Figure 26, does not affect the results much, since recycling is close to cost-neutral. The landfill option has the highest life cycle costs but not enough to make incineration preferable.

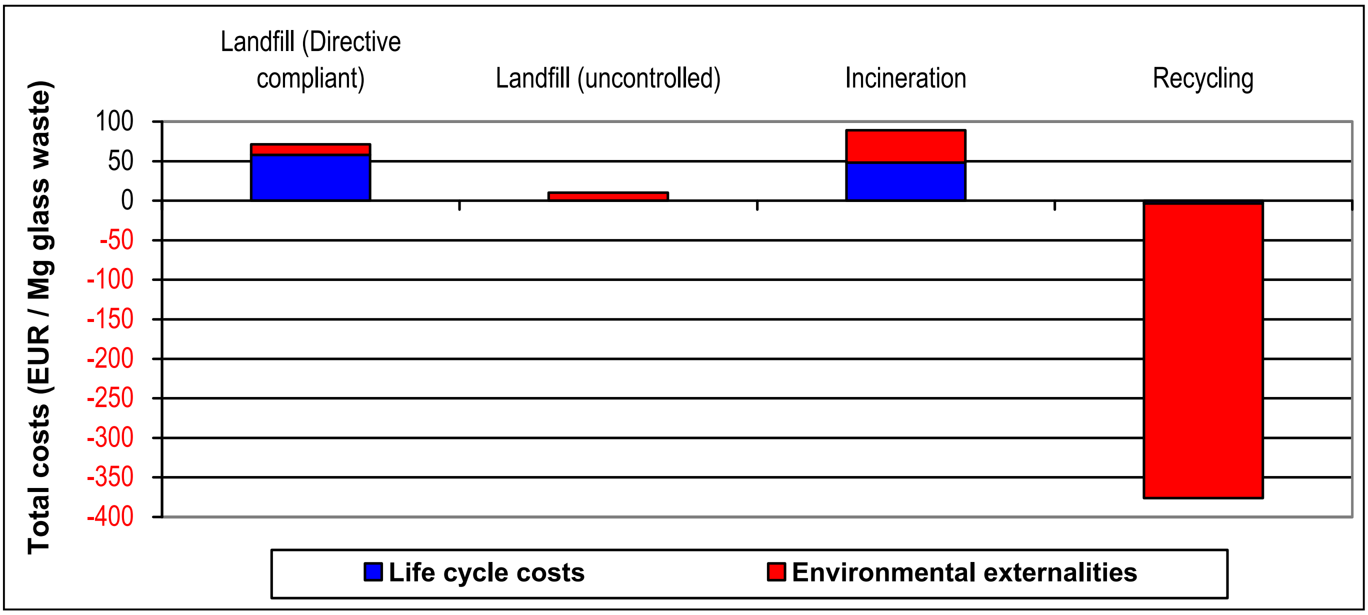

Figure 26 **Societal costs for different treatment options for the glass fractions of MSW. A negative cost is a benefit. Waste collection not included, but an additional cost of separate collection of 29 Euro is added to the recycling option.**

9.6.5 Comparing treatment options for the iron and steel fractions

Figure 27 shows the environmental impacts for different treatment options for the iron and steel fractions of municipal solid waste. It shows that recycling is the option with lowest net environmental impact, although the recycling process is assigned more greenhouse gas and ecotoxic emissions and more injuries than the displaced virgin steel production. (The virgin steel production generally has higher emissions, especially of particulates, while the recycling process is more dependent on electricity, resulting in higher emissions of greenhouse gases. Contaminants in the recycled steel and a larger road transport for recycled steel also explain why recycled steel is not having the lowest impact for all categories.)

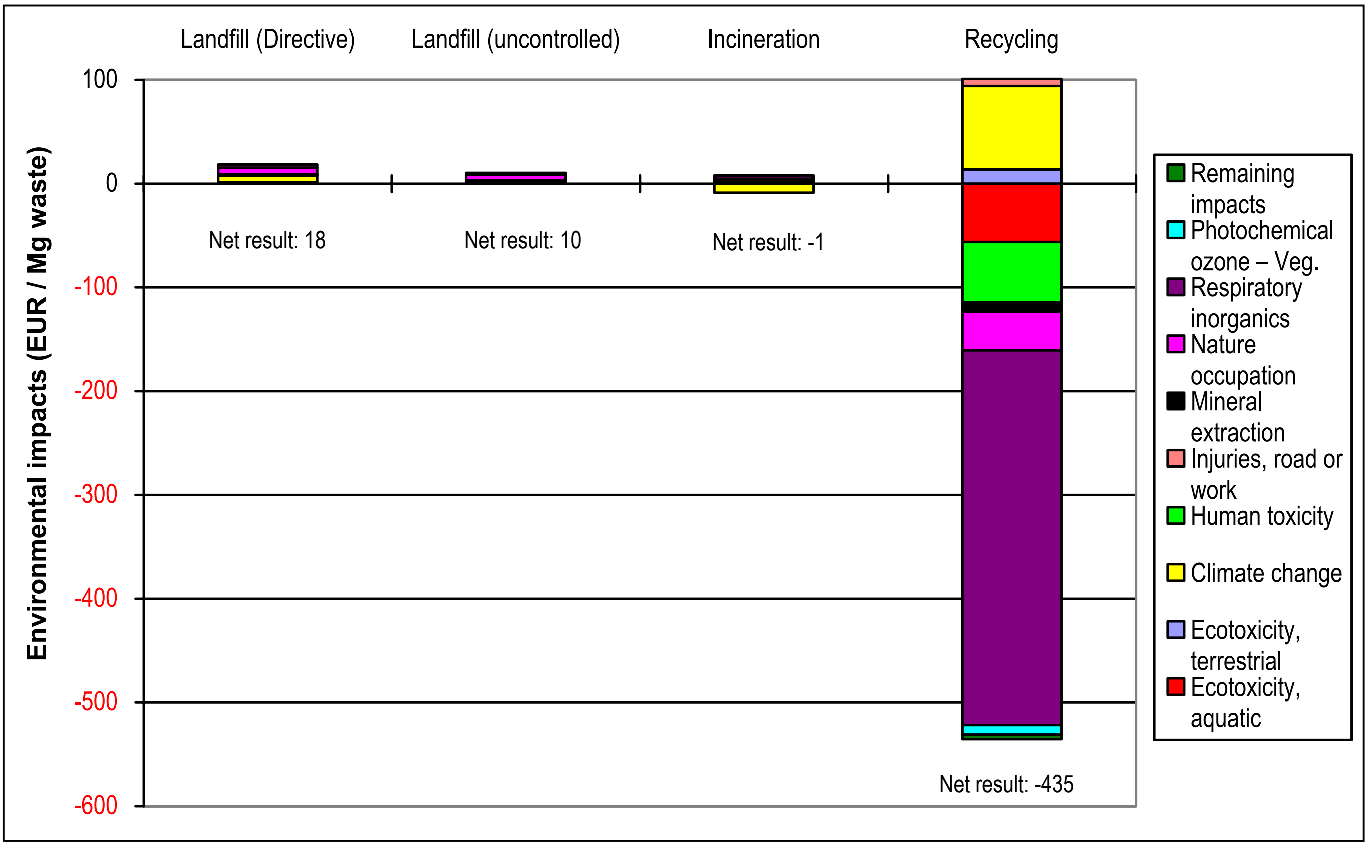

Figure 27 **Environmental impacts of different treatment options for the iron and steel fractions of municipal solid waste (waste collection not included).**
To obtain the total value for a column, the negative values should be subtracted from the positive.

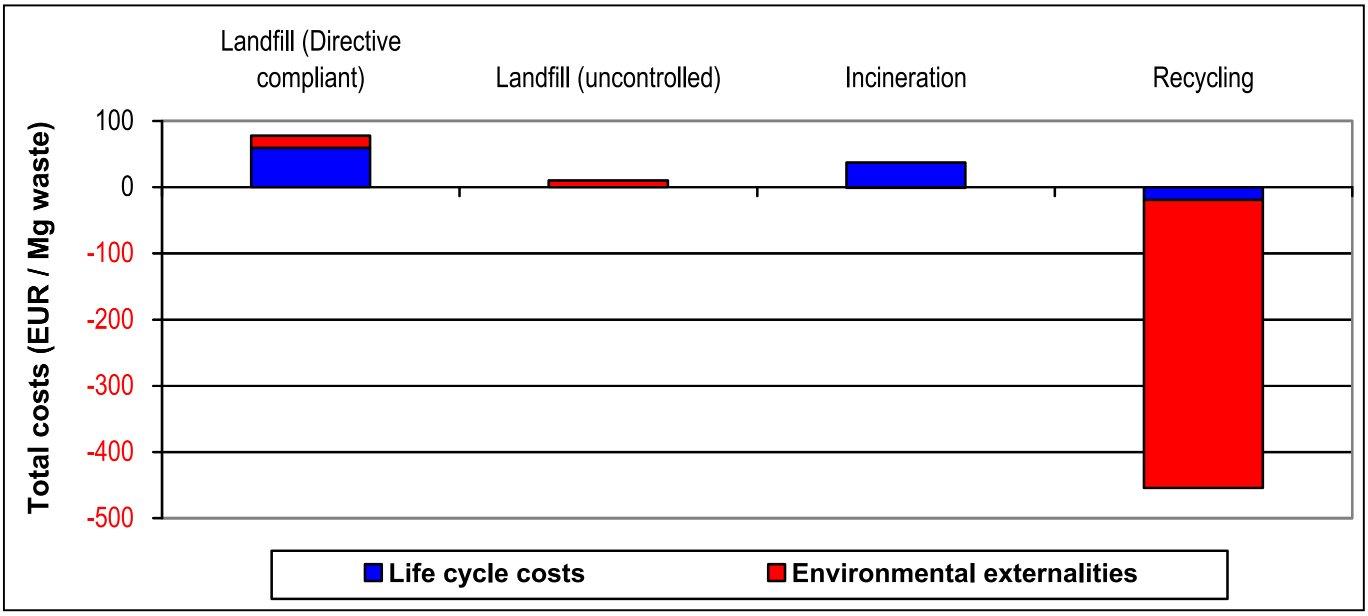

Figure 28 **Societal costs for different treatment options for the iron and steel fractions of MSW.**
A negative cost is a benefit. Waste collection not included, but an additional cost of separate collection of 29 Euro is added to the recycling option.

9.6.6 Comparing treatment options for the aluminium fractions

Figure 29 presents the environmental impacts for different treatment options for the aluminium fractions of municipal solid waste. It shows that recycling by far is the option with lowest net environmental impact.

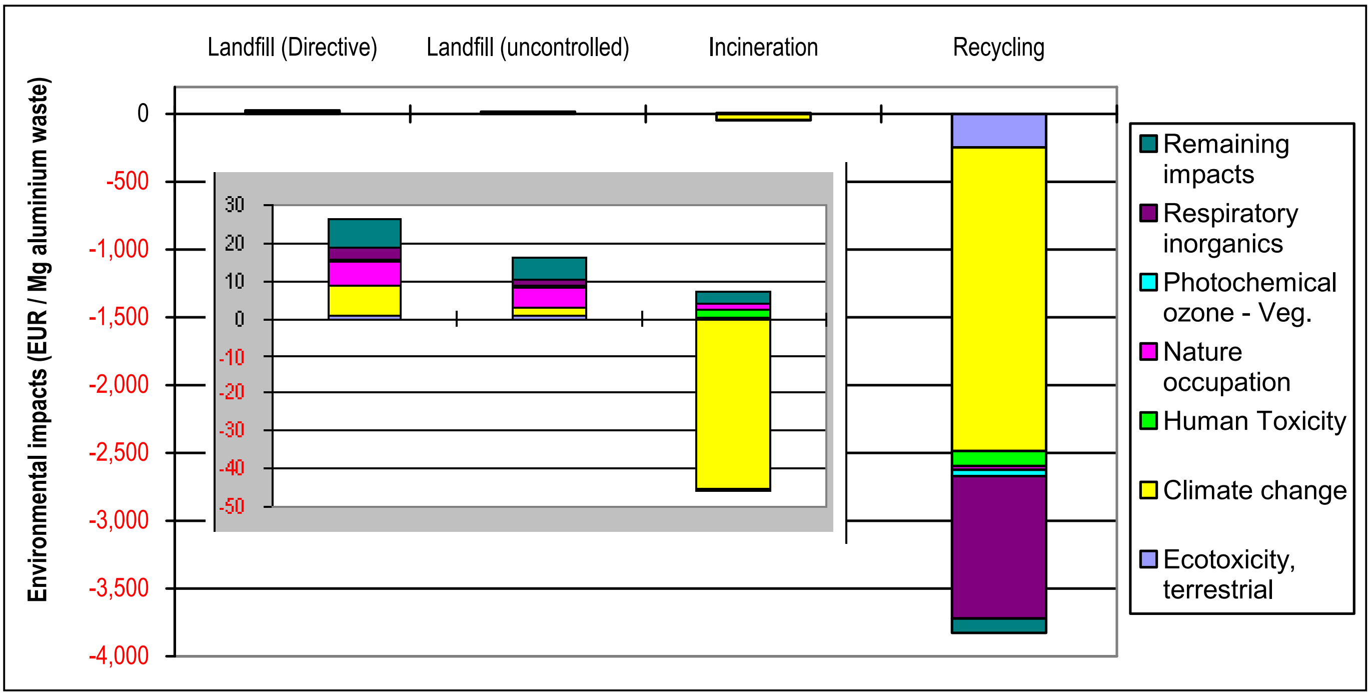

Figure 29 **Environmental impacts of different treatment options for the aluminium fractions of municipal solid waste (waste collection is not included).**
The smaller inserted graph is an enlarged version of the results for the first three columns. To obtain the total value for a column, the negative values should be subtracted from the positive.

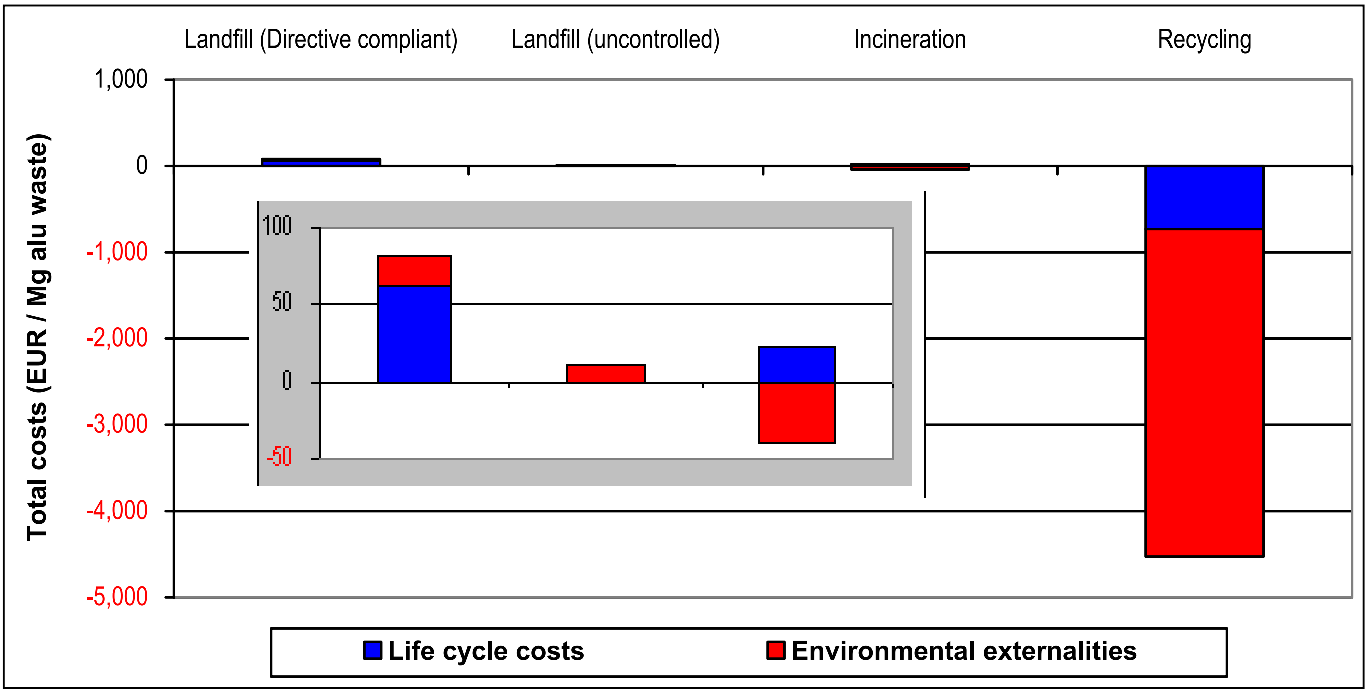

Figure 30 **Societal costs for different treatment options for the aluminium fractions of MSW.**
The smaller inserted graph is an enlarged version of the results for the first three columns. Waste collection not included, but an additional cost of separate collection of 29 Euro is added to the recycling option. A negative cost is a benefit. To obtain the total value for a column, the benefits should be subtracted from the costs.

9.6.7 Comparing treatment options for residual waste fractions

Figure 31 presents the environmental impacts for different treatment options for the residual waste ("Other wastes") fractions of municipal solid waste. Incineration is the option with the lowest environmental impacts. The results are shown for the Krakow composition of "Other wastes" only. Although the composition of "Other wastes" is very different for Malta (see Chapter 5.1.2), the results for the Maltese composition are practically the same.

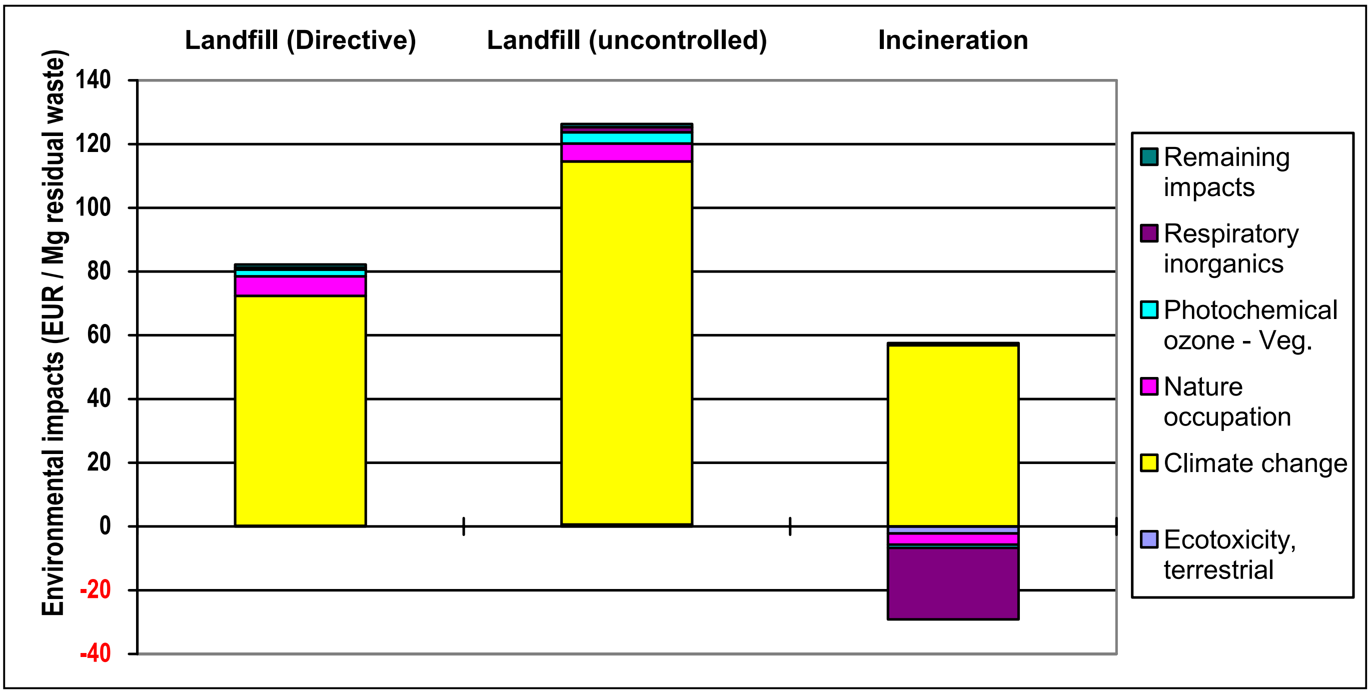

Figure 31 **Environmental impacts of different treatment options for the residual waste ("Other wastes") fractions of municipal solid waste in Krakow (waste collection is not included).**
To obtain the total value for a column, the negative values should be subtracted from the positive.

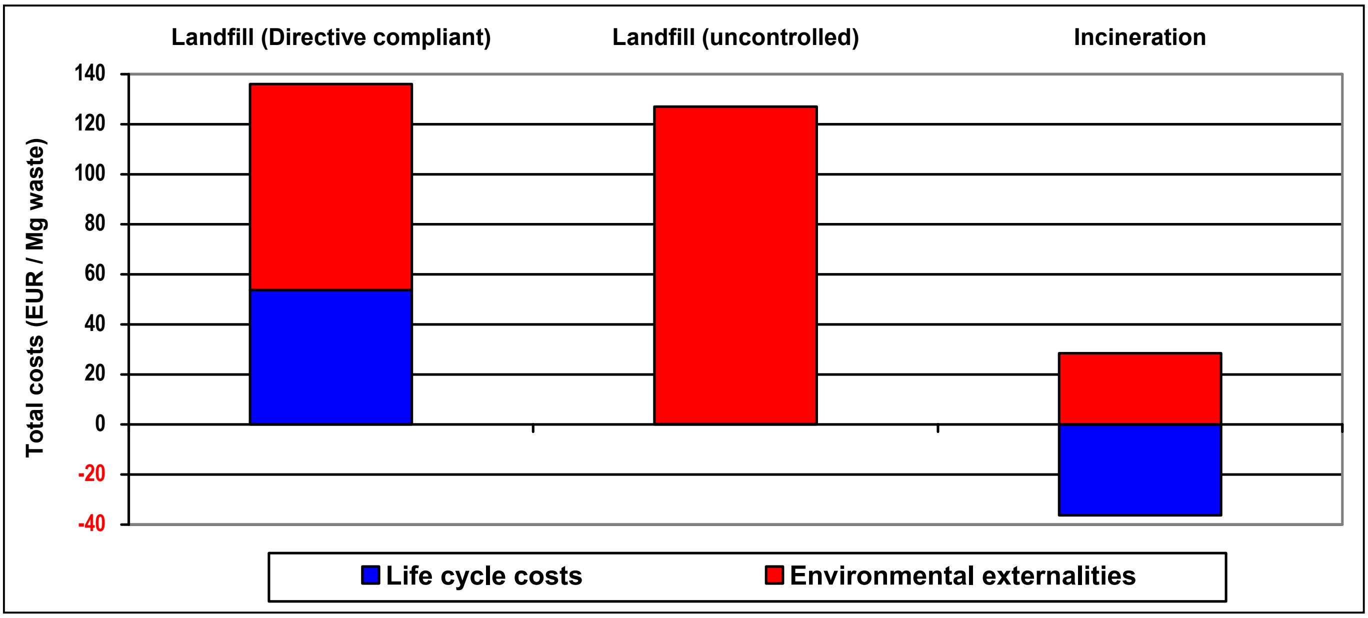

Figure 32 **Societal costs for different treatment options for the residual waste ("Other wastes") fractions of municipal solid waste in Krakow (waste collection not included).**
A negative cost is a benefit. To obtain the total value for a column, the benefits should be subtracted from the costs.

9.7 Comparison of the Krakow and Malta results

Figure 33 places side by side the endpoint results for the Krakow and Malta scenarios to allow easy comparison. This alignment shows that the Malta scenarios generally have much more contribution to climate change and less displaced emissions than the same scenarios for Krakow.

Since the results per waste fraction and waste treatment method are very similar for the two study areas (which is why only the results for one of the study areas were presented in Chapter 9.6), the main explanation for the difference is the difference in waste composition (see Chapter 5.1.1). Malta has a much larger fraction of wet biodegradable wastes, which is a main contributor to climate change (see Figure 17), and at the same time this larger fraction means that materials for recycling take up a proportionally smaller share of the municipal waste.

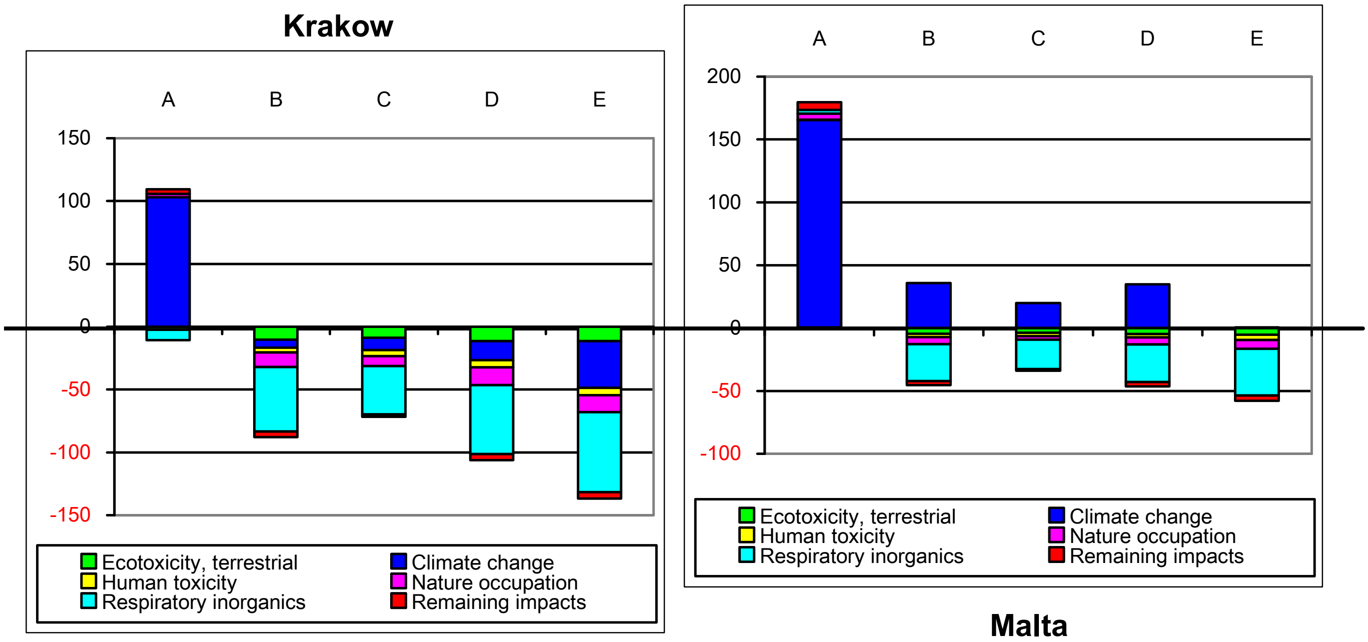

Figure 33 **The endpoint results for the Krakow and Malta scenarios from Figure 13 and Figure 15, but aligned to allow easy comparison (results are "Total environmental impacts in Euro/Mg waste").**

9.8 Importance of site-dependent modelling of emissions / environments

The results presented in Chapters 9.3 and 9.4 are produced with the site-dependent characterisation factors for ecotoxicity, human toxicity (see Annex II), acidification, eutrophication and photochemical ozone formation (see Annex III) applied to the processes that could be geographically identified to take place in Malta and Krakow, respectively, as shown in the figures in Chapter 8. Figure 34 and Figure 35 prove the importance of this site-dependent modelling for the midpoint results.

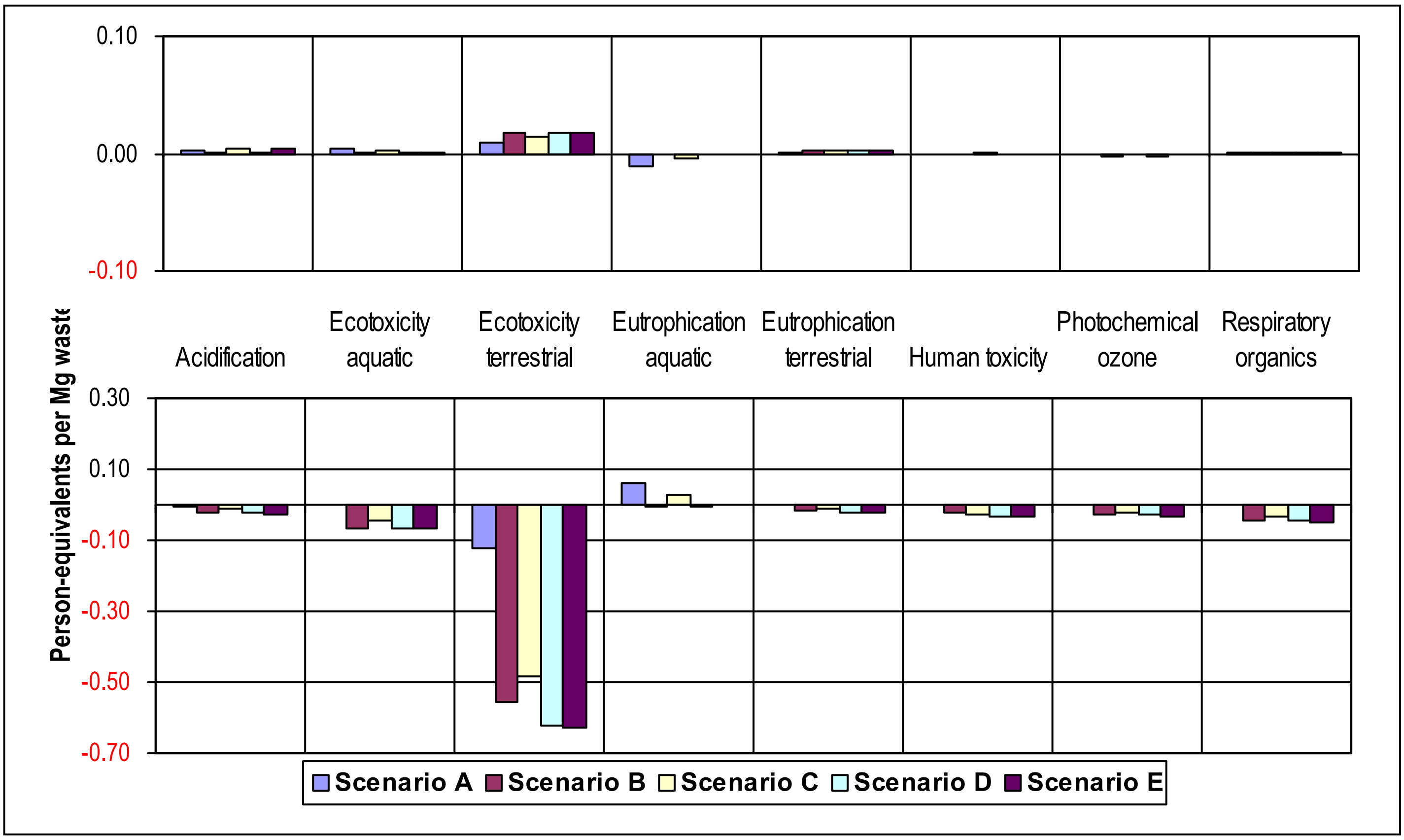

Figure 34 **Normalised results from Figure 11 (lower part) and the differences when applying site-generic characterisation factors (upper part of the figure) – Krakow case.**
To obtain the site-generic result, the results of the two graphs should be added.

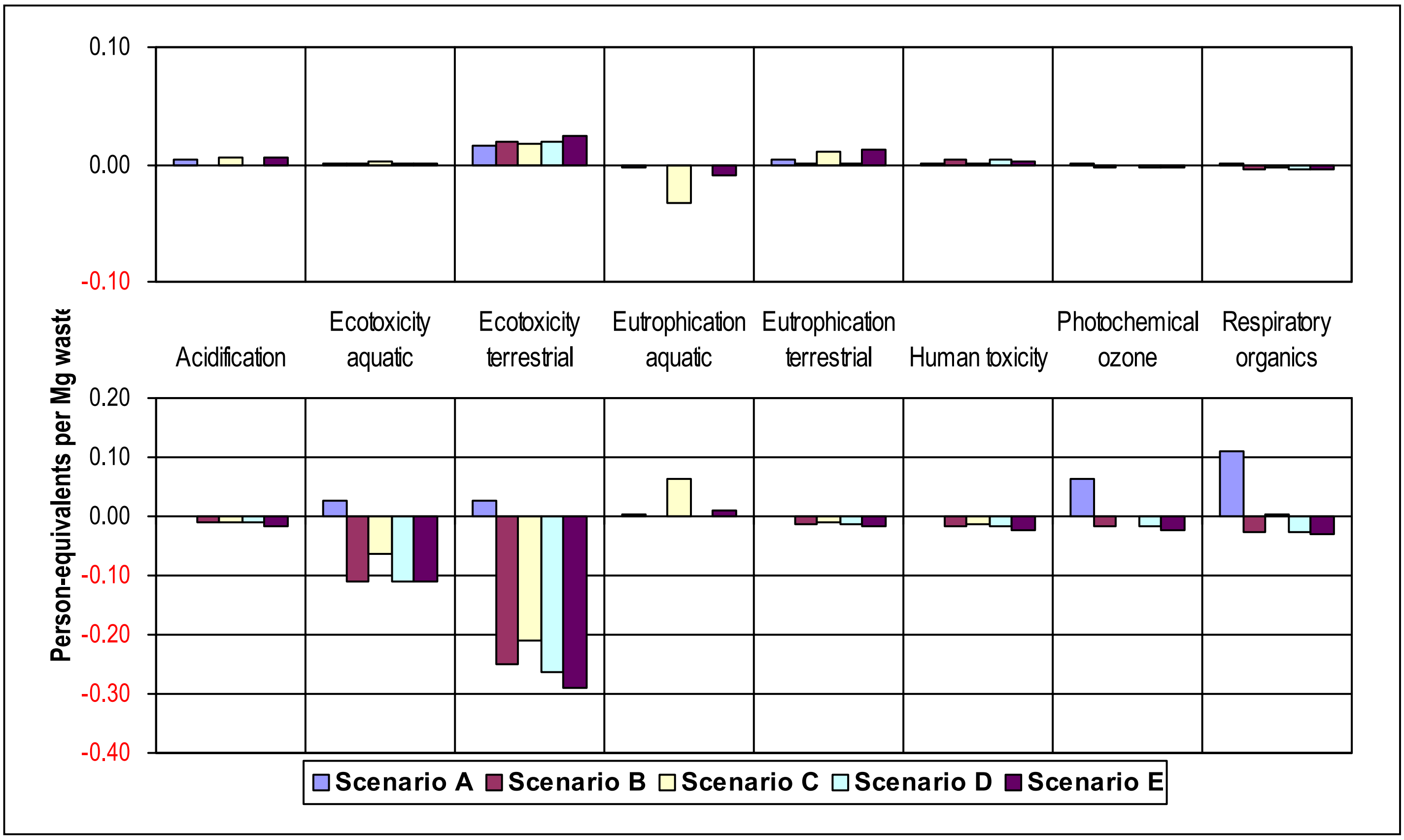

Figure 35 **Normalised results from Figure 12 (lower part) and the differences when applying site-generic characterisation factors (upper part of the figure) – Malta case.**
To obtain the site-generic result, the results of the two graphs should be added.

The largest differences can be found for aquatic eutrophication and terrestrial ecotoxicity. The difference is larger for Malta as a result of the particular geographic conditions (small island) that result in very different characterisation factors. The generally small differences here can be explained by the relatively large part of the emissions that are not geographically specified (see the Figures in Chapter 8) and therefore do not have site-dependent characterisation factors.

Table 57 and Table 58 present the importance of the site-dependent impact assessment modelling for the endpoint results.

Table 57 Differences in the endpoint result for Krakow when applying site-generic characterisation factors.

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	in $Euro_{2003}$ per 1 Mg waste (Krakow)				
Acidification	0.09	0.04	0.16	0.04	0.15
Ecotoxicity, aquatic	0.04	0.00	0.02	0.00	0.00
Ecotoxicity, terrestrial	0.20	0.33	0.26	0.30	0.30
Eutrophication, terrestrial	0.06	0.12	0.17	0.12	0.20
Human toxicity	-0.15	-0.17	0.02	-0.15	0.00
Photochemical ozone – Vegetation	-0.06	-0.17	-0.05	-0.17	-0.11
Respiratory organics	0.00	0.00	0.00	0.00	0.00
All differences	*0.18*	*0.15*	*0.58*	*0.15*	*0.54*
Relative change to results in Table 49	*0.2%*	*-0.2%*	*-0.8%*	*-0.1%*	*-0.4%*

Table 58 Difference in the endpoint result for Malta when applying site-generic characterisation factors.

Impact category	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
	in $Euro_{2003}$ per 1 Mg waste (Malta)				
Acidification	0.18	-0.07	0.28	-0.07	0.25
Ecotoxicity, aquatic	0.00	0.01	0.03	0.01	0.01
Ecotoxicity, terrestrial	0.29	0.37	0.33	0.37	0.48
Eutrophication, terrestrial	0.33	0.11	0.67	0.11	0.83
Human toxicity	0.09	0.67	0.07	0.67	0.37
Photochemical ozone – Vegetation	0.11	-0.18	-0.12	-0.17	-0.17
Respiratory organics	0.00	-0.01	-0.01	-0.01	-0.01
All differences	*1.00*	*0.90*	*1.26*	*0.91*	*1.76*
Relative change to results in Table 53	*0.6%*	*-9.3%*	*-9.1%*	*-8.0%*	*-3.1%*

Also for the endpoint results, as these are analogous to the midpoint results, the differences are small. The relatively large percentage changes for Malta in Table 58, especially for scenarios B to D, are where large impacts are counterweighted by large displaced emissions. This results in a net value close to zero. A small deviation, however, can appear large when expressed in percentages of the net result, as in Table 58.

The largest difference in the endpoint results for Krakow is due to terrestrial ecotoxicity, which also contributes to the difference for Malta, as already expected from the differences in the midpoint results. For Malta there are also important differences for terrestrial eutrophication, due to the very low site-dependent characterisation factors, and for human toxicity, especially in the "incineration scenarios" B and D, due to the lower characterisation factor for dioxin emissions. For Krakow, the characterisation factor for dioxin emissions is slightly higher than the site-generic factor, which explains the slightly lower impacts for human toxicity in the site-generic endpoint result.

The relatively small differences in the overall Endpoint indicator results could be expected, since the impact categories climate change and respiratory inorganics, which dominate the endpoint results, have no site-dependent characterisation models. While this is understandable for climate change, the lack of a site-dependent characterisation model for respiratory inorganics is seen as a major shortcoming for the site-specific impact assessment. Especially for Malta, a site-dependent characterisation model for respiratory inorganics could have led to significant reductions for the local emissions to this impact category. However, it is only a small part of the overall respiratory inorganics that arise from the geographically specified processes, and the amounts of these geographically specified emissions are quite similar between the five scenarios. Thus, a site-dependent modelling for respiratory inorganics would not affect the overall result significantly.

It should be noted that while site-dependent modelling has little importance for the waste treatment scenarios this cannot be taken as an argument for ignoring site-dependency in other contexts, i.e. for comparisons of other types of human activities or for assessments conducted in a regulatory context, addressing e.g. the exceedance of regulatory thresholds for individuals due to peak exposures.

9.9 Comparison of midpoint and endpoint results

When midpoint results show the same ordering of alternatives for all impact categories, there is no need for endpoint modelling. However, it is seldom the case that one option is best in all impact categories. Even though the same ordering of the five scenarios from best to worst: E > D > B > C > A, is found for most impact categories in Figure 11 and Figure 12, there are some impact categories (aquatic ecotoxicity, climate change, human toxicity and aquatic eutrophication) that deviate from this pattern. The endpoint indicator results can, however, be directly compared across impact categories.

When endpoint modelling is not performed, i.e. when decisions need to be based on the midpoint results alone, there can be an inherent psychological tendency to weight all normalised indicator results equally (see also Chapter 6.1.17), even when this is explicitly acknowledged that normalised results do not express any statement of importance. Other approaches also exist for cross-comparing normalised midpoint indicator results with weighting factors.

An equal weighting of the normalised results for all impact categories is the same as stacking the columns of Figure 11 and Figure 12 for each scenario. This is scientifically not justifiable in general. Figure 36 presents such a stacking of the midpoint results for Krakow, for the purpose of comparison to the endpoint result and explaining why this is inappropriate.

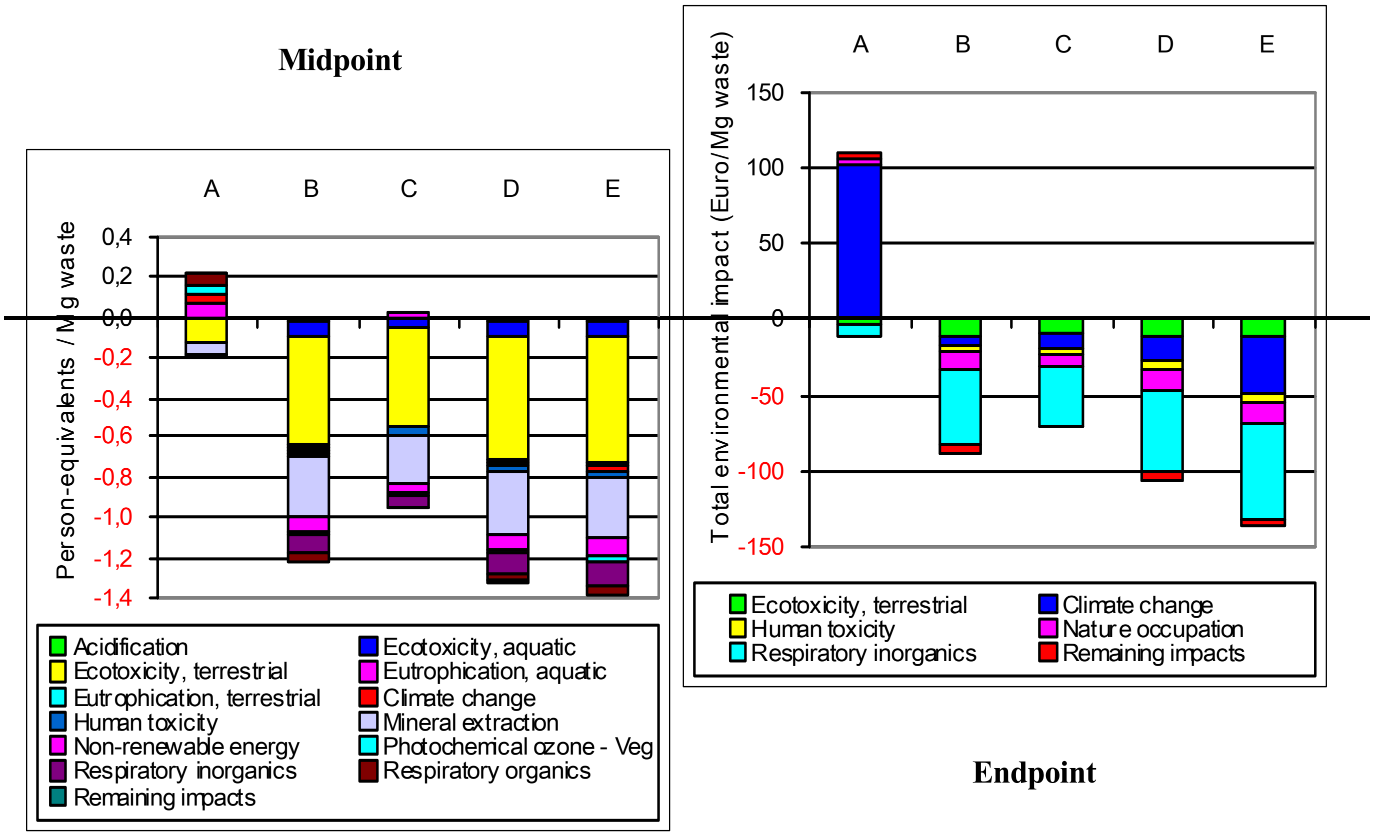

Figure 36 **Comparison of midpoint and endpoint results, when applying a 1:1 weighting of the midpoint results (not recommended as a general procedure).**

The ordering E > D > B > C > A is found both for the stacked midpoint result and the endpoint result. However, this is more a coincidence here than an inherent feature of correspondence in the two approaches. This can be seen from the differences in the dominant impact categories in each set of results.

One may therefore say that the midpoint method produces the “right” overall result (in terms of the same ordering of the scenarios as in the endpoint results) but with the “wrong” arguments (i.e. placing the emphasis on the two impact categories “Terrestrial ecotoxicity” and “Mineral extraction”, that are not particularly important from an endpoint perspective, while downplaying the role of climate change and respiratory inorganics that dominate the endpoint result).

9.10 Results from application of input-output data

From Figure 37 and Figure 38, the input-output-based inventory data suggest that there are more impacts for waste collection (see Chapter 5.2.3) and for the upstream processes of waste incineration (Chapter 5.4.2) than using the process-based inventory alone. A deeper analysis of the methods and underlying data is however required, as higher numbers does not automatically mean that they are more complete.

Generally, the impact categories have the same relative importance in the results, with the exception of nature occupation, which is nearly missing in the process-based results for waste collection, and aquatic ecotoxicity, where the process-based data in contrast include some more emissions. This can either be caused by different quality of the used process-based data or by distortions of the results due to methodological assumptions when using the economic input-output data.

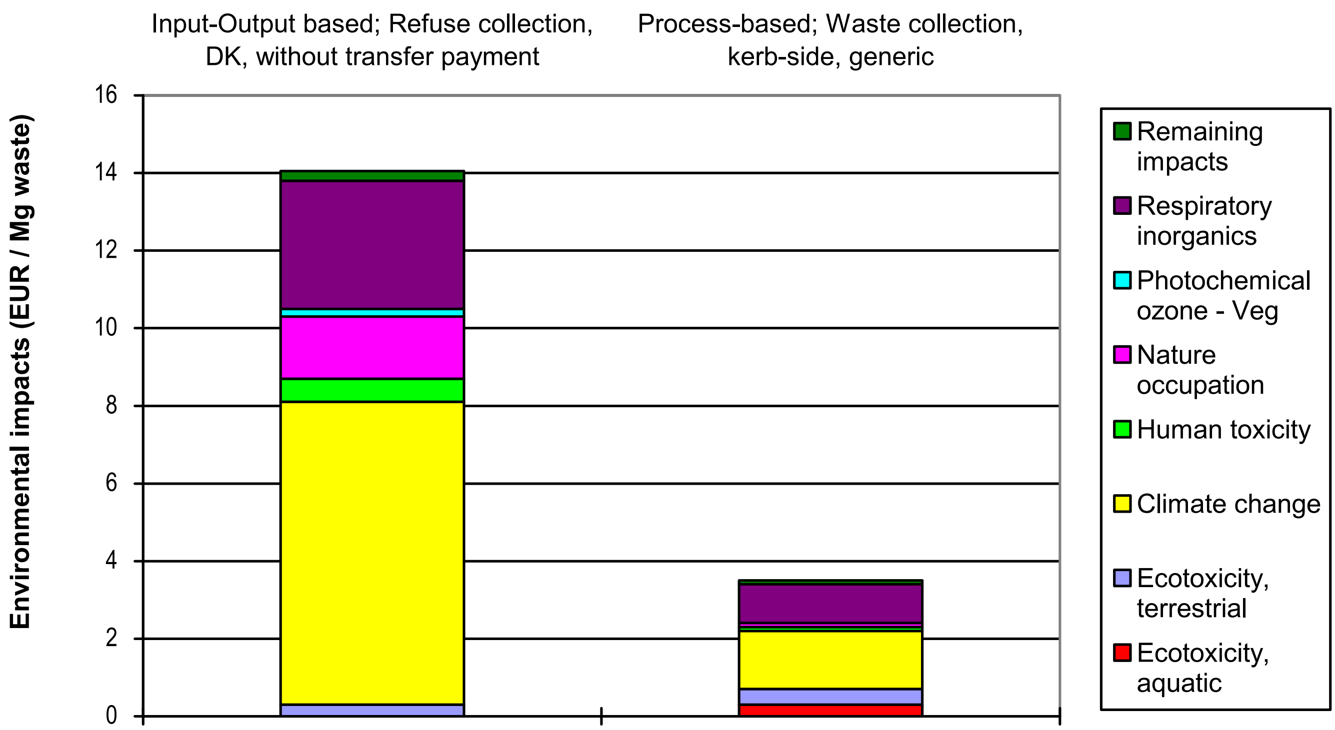

Figure 37 **Comparison of endpoint results for waste collection, using input-output based data versus process-based data.**

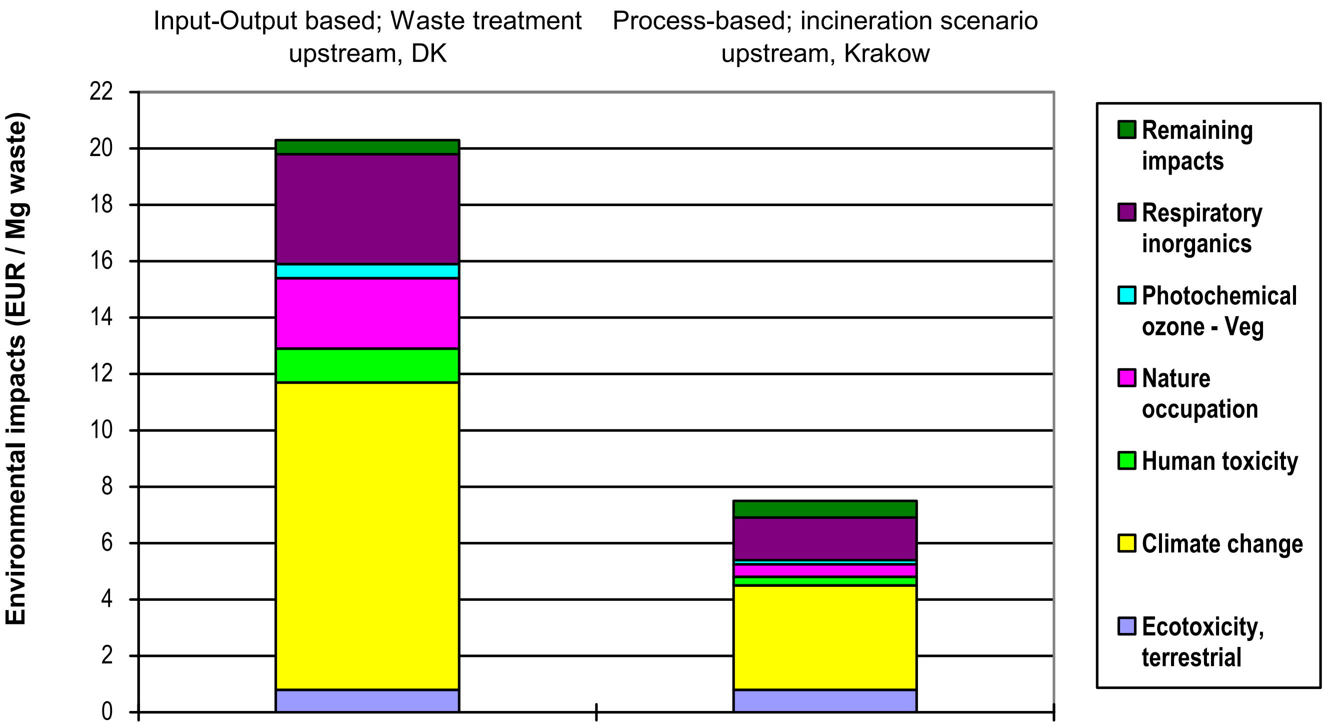

Figure 38 **Comparison of endpoint results for waste incineration, using input-output based data versus process-based data.**

To explain the differences in more detail, an analysis was conducted of the processes contributing to the endpoint results in Figure 37 and Figure 38. While environmentally extended input-output databases report the emissions for each complete industry branch, the Ecoinvent database reports combustion of fuels in unit processes. Therefore, the comparison of input-output data with process data is not entirely easy, as can be seen in Table 59 and Table 60.

Table 59 **Processes contributing to the total environmental impact (in Euro$_{2003}$) of waste collection, using input-output based data versus process-based data.**

Process	Input-Output-based refuse collection, DK, without transfer payment	Process-based waste collection, kerb-side, generic
	Euro per Mg waste	Euro per Mg waste
Direct emissions	1.69	1.84
Electricity and district heat	1.32	0.16
Machinery	0.94	-
Motor vehicles	0.77	0.01
Basic non-ferrous metals	0.64	0.61
Detergents & other chemical products	0.59	-

Process	Input-Output-based refuse collection, DK, without transfer payment	Process-based waste collection, kerb-side, generic
	Euro per Mg waste	Euro per Mg waste
Freight transport by road	0.54	0.03
Food	0.53	-
Refined petroleum products etc.	0.52	-
Radio and communication equipment	0.40	-
Air transport	0.36	-
Transport by ship	0.35	0.04
Concrete, asphalt and rockwool products	0.31	0.01
Wood products	0.30	-
Ferrous metals	0.29	0.13
Vegetable and animal oils and fats	0.22	-
Office machinery and computers	0.22	-
Fertilizers etc.	0.20	-
Wholesale trade	0.18	-
Rubber products, plastic packing etc.	0.18	0.02
Dyes, pigments, organic basic chemicals	0.18	0.01
Coal, crude petroleum, natural gas etc.	0.17	0.02
Cement, bricks, tiles, flags etc.	0.16	-
Construction materials of metal etc.	0.15	-
Hand tools, metal packaging etc.	0.13	-
Basic plastics and synthetic rubber	0.11	-
Tobacco products	0.11	-
Furniture	0.09	-
Agricultural services and landscape gardeners	0.08	-
Medical & optical instruments etc.	0.08	-
Civil engineering	0.08	0.03

Process	Input-Output-based refuse collection, DK, without transfer payment	Process-based waste collection, kerb-side, generic
	Euro per Mg waste	Euro per Mg waste
Printing activities etc.	0.08	-
Toys, gold & silver articles etc.	0.08	-
Restaurants and other catering	0.08	-
Paints and printing ink	0.07	-
Broadwoven cotton	0.07	-
Transport via railways	0.07	0.01
Fuel combustion in various industries	-	0.33
Remaining processes	1.74	0.13
Total	**14.10**	**3.37**

In Table 59, all industries are lower in the process data, including items such as electricity, machinery, motor vehicles, chemicals, air and ship transport, radio and communication equipment, and wood products, which also appear on the list with no entries of Table 60.

Table 60 **Processes contributing to the total environmental impact (in Euro$_{2003}$) of the upstream processes providing input to waste incineration, using input-output based data vs. process-based data.**

Process	Input-Output-based Waste treatment upstream, DK	Process-based Incineration scenario upstream, Krakow
	Euro per Mg waste	Euro per Mg waste
Machinery	2.88	-
Radio and communication equipment	1.62	-
Electricity and district heat	1.50	0.62
Ferrous metals	1.43	1.31
Basic non-ferrous metals	1.24	0.24
Cement, concrete, asphalt and rockwool products	0.79	0.93
Construction materials of metal etc.	0.75	-
Detergents & other chemical products	0.62	-
Office machinery and computers	0.55	-

Process	**Input-Output-based Waste treatment upstream, DK**	**Process-based Incineration scenario upstream, Krakow**
	Euro per Mg waste	**Euro per Mg waste**
Wood products	0.52	-
Freight transport by road	0.52	2.04
Refined petroleum products etc.	0.49	-
Air transport	0.42	-
Pulp, paper and paper products	0.39	-
Transport by ship	0.37	0.10
Motor vehicles	0.32	-
Furniture	0.26	-
Vegetable and animal oils and fats	0.25	-
Wholesale trade	0.25	-
Dyes, pigments, organic basic chemicals	0.25	-
Fertilizers etc.	0.21	0.19
Toys, gold & silver articles etc.	0.21	-
Rubber products, plastic packing etc.	0.20	0.07
Medical & optical instruments etc.	0.20	-
Starch, chocolate and sugar products	0.18	-
Meat and meat products	0.17	-
Basic plastics and syntethic rubber	0.17	-
Civil engineering	0.16	-
Coal, crude petroleum, natural gas etc.	0.14	0.07
Printing activities etc.	0.13	-
Paints and printing ink	0.12	-
Broadwoven cotton	0.12	-
Gravel, clay, stone and salt etc.	0.11	0.06
Slag and building waste disposal	-	0.38
Fuel combustion in various industries	-	0.84
Remaining processes	2.67	0.75
Total	**20.2**	**7.60**

Comparing results industry by industry, it is obvious that in the given comparison the process-based data are generally lower, with “Freight transport by road” in Table 60 as a notable exception, and for many input-output industries the process data simply lack a comparable entry and vice versa. However, it should be noted that the process “Fuel combustion in various industries” at the bottom of the two tables should be distributed over all the industries above it. Nevertheless, the size of this process is not even large enough to fill the gap of the first industry on the list in which there is no entry (“Machinery”).

It might be preferable to use input-output based data in combination with the more specific process-based data (hybrid approaches), although this needs to be established considering the underlying methodological merits and limitations of using such economic input-output data for environmental assessments. Unfortunately, this hybrid approach could not be followed in this project, due to the missing detail of the input-output data with respect to material recycling; see Chapter 5.3.5. Applying input-output data for landfilling and incineration, but not for material recycling, would give a bias in the assessment in favour of the former options.

It may be argued that the potential data gaps identified in the Ecoinvent data raise questions about the reliability of any conclusions drawn from applying these process data. This would especially be the case if the data gaps were proportionally larger for one scenario than for another and if input-output methods were not without limitation. However, it is not clear here whether there is incompleteness or whether the differences are caused by methodological problems of the input-output approach. Also, as many upstream processes appear in all of the five scenarios, thus a high degree of covariance in the completeness between them is expected, so that much of this potential incompleteness will cancel out in a relative comparison. Furthermore, a large part of the total emissions of the analysed systems come from the waste treatment processes, rather than from the upstream processes that are affected by the potential data gaps.

10 Effect of discounting

The above results were all derived from undiscounted data. The economic cost data include financing costs, but assume constant cost over time, i.e. that capital investments are made on a continuous basis.

Discounting implies that the importance (cost) of environmental impacts occurring in future is reduced by a factor, known as the discount rate, for each unit of time that the impact is removed from the present.

Discounting of future costs and benefits would therefore mainly affect the weight of environmental impacts relative to the economic costs, i.e. it would favour the "economic optimum" scenario D at the expense of the other more expensive scenarios. This is of particular relevance when comparing to the "societal optimum" scenario E. The effect of discounting increases with the size and uniformity of the discount rate.

Furthermore, discounting would reduce the importance of impact categories with long-term impacts (such as climate change and nature occupation) more than impact categories with more immediate impacts (such as toxicity and eutrophication). However, this would not affect the ordering of the five analysed scenarios, since most impact categories give the same ordering of the scenarios; see Chapters 9.1 and 9.2.

It should equally be noted that discounting environmental impacts that may occur on future generations is not inline with the fundamental principles of sustainability.

11 Interpretation and recommendations for waste management strategy

11.1 Strategic recommendations

There are large economic and environmental advantages, even benefits from avoided impacts, in a strategy that completely avoids landfilling of municipal wastes. This is particularly the case in the context of climate change.

For all separately collected waste fractions, recycling (including composting with energy recovery) is usually the waste treatment option with the lowest environmental impact, and for the remaining wastes ("Other wastes" and the residuals that are not separately collected) incineration is the option with the lowest environmental impact.

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 has the lowest societal cost even for these waste fractions.

The results in this study are adequately clear to support general waste management decisions, and are not influenced significantly by local conditions. The study has been extensively peer reviewed.

The results help point to the following strategic recommendations:

- Initiatives are required to overcome any financial, technical and psychological barriers for increased recycling of separately collected waste fractions.
- Government intervention may be necessary to ensure recycling also of some of the waste fractions with a high heating value, since on a purely economic basis incineration appears to be 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 and types under increasing rates of recycling and composting, to avoid over-investment in capacity and consequent technological lock-in.
- Government waste management interventions might most efficiently be made at the EU level, due to what appears to be the low importance of geographic variations and the disperse nature of impacts/benefits of the regional/global scale when considering a life cycle perspective. However, this will not replace the additional need to consider variations from a local impact perspective in relation to choosing the location of facilities including the local need for e.g. heat produced or compost, meeting legislative requirements, etc.

11.2 Limitations

The scope of this study was limited to alternative waste treatment options (landfilling, incineration, composting and material recycling), considering wastes already generated. In many cases, in accordance with the principles of the waste hierarchy, the prevention of waste generation through more sustainable consumption and production can prove a more cost-efficient and environmentally sound management strategy than waste treatment.

This study does not investigate *reuse* as an alternative to material recycling. The environmental merits of reuse systems are very dependent on local transport distances, and the cost is often decisive.

The scenarios applied in this study, as well as the associated emissions and results, are not actual predictions of future situations, as these can be influenced by changes including in waste composition (which was kept constant in this study).

This study has been based on specifically described current best available technologies, and that other - both current and future technologies - may have different performances to those described in this study.

While the impact assessment methods applied cover many important environmental (biophysical) impact categories related to waste management activities, the methods are not complete. Omissions that were covered in other studies include:

- Disamenities (related to the localisation of waste treatment plants)
- Noise, and time lost due to traffic congestion (both closely related to amount of transport and will therefore in relative importance between scenarios follow other included impact categories, notably injuries)
- Impacts of air pollution on buildings, fertiliser effects of nitrogen and sulphur emissions, several minor economic production impacts of climate change (all excluded due to their low importance).

It should be noted that there are many likely data gaps in the emissions and resource consumption inventory and possibly in the impact assessment. Nevertheless, these studies are based on current state-of-the-art information and practice. Preliminary approaches were adopted to highlight uncertainties associated with available data, suggesting the overall conclusions and main findings are likely to remain robust. As climate change is a dominant impact category in determining the societal optimum solution, uncertainties associated with the emission of greenhouse gases are important.

11.3 Similarity and differences to previous studies

In general, the conclusions of this study 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 the potentials offered by composting with energy recovery.

This study applies more recent environmental and costing data, representative of the best available technology. Especially for the composting option, this is important for the results.

The study assumes 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 options.

12 Methodological observations

12.1 Inventory methodology

By considering the use of conventional process-based life cycle assessment complimented with environmentally-extended economic input/output data (NAMEA matrices), this study identified potential advantages and disadvantages. Further critical investigations are necessary to identify the advantages and the limitations of the two approaches, particularly the potential merits of using a combination of the two in a hybrid method.

In the mainstream or "conventional" life cycle assessment method, where emissions and resource consumption data are based on clearly defined unit processes, expertise and experience based cut-off rules used to identify where the life cycles of various minor inputs no longer need to be considered, may lead to significant data gaps, if not properly done. For waste collection and the upstream inputs to waste incineration, data gaps of 76% and 62% of the total environmental impacts, respectively, were suggested through comparisons of the Ecoinvent data with so-called input-output data-based approaches (Chapter 9.10), while the completeness and correctness of the input-output data remains to be established.

In attempting to complete the data from the mainstream life cycle method with data from input-output based NAMEA matrices, problems were encountered in obtaining data at an adequately disaggregated level for material recycling. Material recycling is an example of a special problem in input-output-tables, since the processing of primary and secondary raw materials are taking place in the same aggregated industries, thus blurring the important environmental differences between these processing routes.

For situations of co-production, the preferred ISO procedure (1) subdivision of unit processes and collection of the separate inventory data or 2) system expansion) is not yet consistently applied in standard, commercial LCA databases. Partly it is also not or not easily applicable as data on a further differentiated unit processes is not available, or as system expansion would result in extended system boundaries that would not fit anymore together with the goal of the study, resulting in the need for allocation. This may lead to inconsistencies when using such databases to provide background data for an LCA of specific processes for which co-products are treated through either system expansion or allocation. This was especially problematic because recycling is an important part of the analysed systems, and it was necessary to adjust some of the background processes to avoid such inconsistencies that would influence the results.

12.2 Impact assessment methodology

Indicator results at both the midpoint and endpoint in a common framework provide complimentary insights and information. The advantages and disadvantages of the two approaches have been widely discussed elsewhere, including the additional step considered in this study of taking endpoint results a further step to external costs and comparing these with economic costs.

In theory, midpoint indicators provide a point at which equivalence in impacts between different substances or other inventory indicators can be established. At the same time, the indicators are not comparable across impact categories such as climate change and ecotoxicological effects. In practice, however, equivalence at the midpoint only exists for impact categories such as climate change and other midpoint indicators may not account for all steps, hence differences, in cause-effect mechanisms.

It could be argued that the use of midpoint methods should be generally dissuaded for decision support, while maintaining the important role of midpoints as important calibration points in the impact pathways. On the other hand, the importance of impact categories such as climate change may be underestimated in terms of likely damages and others such as certain non-cancer human health effects may be overestimated due to unquantifiable uncertainties such as whether biological thresholds will be exceeded or not in the complex reality of human exposure to mixtures of contaminants. Interpretation at both the midpoint and endpoint indicator level are therefore recommended, considering all available information including qualitative knowledge and the precautionary principle.

While site-specific impact assessment methods were adopted in this study for some emissions, the importance of site-dependency was found here to be low. However, characterisation factors may differ significantly for some types of chemical emissions and for some locations. These distinctions should be further investigated to also provide more guidance based on likely reductions in uncertainty attained using site-dependent factors in some cases. In general, such considerations may not be necessary for disperse sources of emissions associated with background inventory data.

In the attempt to combine the better of two existing impact assessment methods, and expand on missing areas, some obstacles were encountered that require further elaboration:

- Better consideration of the speciation of metal emissions, including in the inventory.
- The need for an impact characterisation model for emissions to groundwater.

- The characterisation models for e.g. metals and persistent organic chemicals in the context of toxicological effects may not adequately reflect irreversible binding and bioavailability over time in different environmental media.
- The endpoint characterisation models for ecotoxicity should be checked/calibrated to reflect the overall importance of ecotoxicity relative to other impacts on ecosystems
- There is a need to provide consistent endpoint indicators for ecotoxicological effects with those of other ecosystem impact categories.
- An endpoint characterisation model for aquatic eutrophication is missing.
- An endpoint characterisation model for tropospheric ozone impacts on vegetation is missing. This affects both the assessment of ecosystem impacts and impacts on agricultural crop production.
- A separate impact category for agricultural crop production should be created, which should include both the impact of ozone and the impacts of other ecotoxic substances on crop yields, the fertilisation effect of CO_2 and the different mineral nutrients in emissions, as well as soil losses through erosion. It could also include the non-fertiliser effect of adding compost to soil (e.g. reduced erosion, impacts on soil pathogens, improved soil workability and water retention capacity).
- A characterisation model for ecosystem impacts during relaxation after deforestation and climate impacts is missing.
- The lack of a site-dependent characterisation model for respiratory inorganics is seen as a potential shortcoming for the site-specific impact assessment.
- The available normalisation reference for Europe is from 1995. Its usefulness should be investigated and updates made, if warranted, on a continuous basis.
- The endpoint characterisation model for climate change should be updated, improved and better documented.
- As the endpoint method includes a number of additional assumptions that may be controversial, a wider scientific and stakeholder review procedure is needed to approach consensus on the procedures and values used.

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Annex I.

Peer review comments and replies

Downloadable at http://viso.jrc.it/lca-waste-partII-annexI.pdf

Annex II.

Human toxicity and ecotoxicity characterization and normalisation factors using a new multi-continental version of IMPACT 2002: Application to emissions in Krakow, Malta and a European uniform emission

Downloadable at http://viso.jrc.it/lca-waste-partII-annexII.pdf

Annex III.

Site-dependent midpoint characterisation, normalisation and damage assessment factors for the impact categories acidification, terrestrial eutrophication, aquatic eutrophication and photochemical ozone formation.

Downloadable at http://viso.jrc.it/lca-waste-partII-annexIII.pdf

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Abstract
The European Commission's Strategy on the Prevention and Recycling of Waste outlines why life cycle thinking is essential in the move towards more sustainable consumption and production. The importance of life cycle thinking is further highlighted in the Commission's complimentary Strategy on the Sustainable Use of Natural Resources, in its Integrated Product Policy, as well as in the proposed revisions to the European Waste Framework Directive and the up-coming Sustainable Consumption and Production Action Plan.
In 2004, following its international workshop and conference on life cycle assessment and waste management, the Institute for Environment and Sustainability (IES) of the European Commission's Joint Research Centre (JRC) launched a series of regional pilot case studies in collaboration with representatives of the European Union's new member states, acceding countries, and associated countries. The representatives selected, and provided, statistical data for nine waste management regions. The life cycle assessments took into account the situation around 2003 in each region and example management scenarios that achieve Directive compliance and beyond (ref. Koneczny K., Dragusanu V., Bersani R., Pennington D.W. Environmental Assessment of Municipal Waste Management Scenarios: Part I – Data collection and preliminary environmental assessments for life cycle thinking pilot studies, European Commission, JRC-IES, 2007).
This report, based on a study carried out on behalf of the JRC by 2.-0 LCA Consultants, considers in further detail the waste management options for the island nation of Malta and the central European city of Krakow, Poland. The life cycle assessments use more robust data, apply cutting edge methodologies, and take into account the waste management costs.
The resultant life cycle impact indicators provide a basis to compare the emissions and resources consumed attributable to each waste management option in terms of their contributions to e.g. different environment and human health burdens. One of the methods furthermore highlights how some of the trade-offs between environment, health, and the waste management costs might be partially considered in a single life cycle based cost-benefit framework, as a support to other decision-making information.

The mission of the JRC is to provide customer-driven scientific and technical support for the conception, development, implementation and monitoring of EU policies. As a service of the European Commission, the JRC functions as a reference centre of science and technology for the Union. Close to the policy-making process, it serves the common interest of the Member States, while being independent of special interests, whether private or national.

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