Department of Civil and Structural Engineering

Environmental Costs and Environmental Benefits Analysis of Packaging Waste Recovery and Recycling Targets

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ABSTRACT

Society is faced with the growing problem of waste associated with mass consumption. The treatment and final disposal of waste is linked to a wide range of environmental problems, including loss and wastage of resources, atmospheric, aquatic and land pollution, as well as public health concerns. For these reasons, since the early 1990s there has been an emphasis on waste minimisation and recycling initiatives. The European Commission decided that packaging waste would be its first target in an aim to reduce waste in general - to be followed by several other producer responsibility type legislations. The landfill Directive came into force in 2002 – It reduces the amount of bio-degradable waste that can be landfilled and bans hazardous waste from most landfill sites. The End of Life Vehicle Directive came into force in 2003 and put the responsibility on the producer to organize recovery and recycling of vehicles. The Waste Electrical and Electronic Equipment (WEEE) came into force in 2004 and requires manufacturers of such products to finance their recovery and recycling.

This study looks at the UK Producer Responsibility Obligations (Packaging Waste) Regulations 1997 and the targets that have been chosen to enable the UK to fulfil the requirements of the European Directive (94/62/EC) on Packaging and Packaging Waste.

The aim of the research focuses on establishing target levels with maximum environmental benefits, specifically for recovering and recycling cardboard packaging waste in the UK. The methodology used is Life Cycle Assessment (LCA), which considers the whole life cycle of cardboard packaging, including the manufacture of packaging from raw (or recycled) fibres, its transport and use and waste management options.

A range of scenarios have been modelled to reflect present day achievements, the levels of recycling expected of Member States through the revised Directive targets, as well as extreme scenarios. The scenarios are:

- Base scenario: 53% recycling, 4.23% incineration and 42.77% landfill
- Scenario 2: 60% recycling with 37.2% landfill and 2.8% incineration
- Scenario 3: 70% recycling with 27.9% landfill and 2.1% incineration
- Scenario 4: 80% recycling with 18.6% landfill and 1.4% incineration
- Scenario 5: 35% recycling with 60.45% landfill and 4.55% incineration
- Scenario 6: 100% landfill
- Scenario 8: 100% incineration

It was found that significant reductions in global warming and carcinogens are associated with increasing levels of recycling (the highest level assessed was 80% recycling), but this comes at a cost of a slight increase in energy usage impacts. Global warming impacts fall by 20% with an increase in recycling from 53% to 80%. However, some of these potential benefits are compromised if waste cardboard needs to be exported to Europe for recycling.

This particular project is looking at waste related policy issues. However it needs to be acknowledged that the manufacturing of cardboard packaging accounts for a significant proportion of the total burdens associated with the cardboard-packaging life cycle. These burdens are not affected by waste management policies; instead they would require improvements in the manufacturing processes to be made.

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GLOSSARY – DEFINITIONS ABBREVIATIONS

CML: Center for Environmental Studies, at Leiden University, the Netherlands

DETR: Department of Environment, Transport and the Regions

DEFRA: Department of Environment, Food and Rural Affairs

EA: Environment Agency (England and Wales)

IPCC: Intergovernmental Panel on Climate Change

MSW: Municipal Solid Waste

PPIC: Confederation of Paper Industries

SETAC: Society of Environmental Toxicology and Chemistry

Definitions

Category endpoint: Variables which are of direct societal concern, such as human life span or incidence of illness, natural resources, valuable ecosystems or species, fossil fuels and mineral ores, monuments and landscapes, man-made materials.

Category midpoint: Variables in the environmental mechanism of an impact category between the environmental interventions and the category endpoints, such as the concentration of toxic substances, the deposition of acidifying substances, global temperature or sea level.

Characterisation: The step of life cycle impact assessment (LCIA) in which the contributions made by the environmental interventions to each impact category are assessed by quantitative or qualitative methods.

Eco Indicator 99: weighting methodology

Environmental effect: A consequence of an intervention in the environmental system.

Environmental impact: any adverse change to the environment including one or more environmental effects.

The Directive: The Packaging and Packaging Waste Directive (94/62/EC)

Recyclate: Secondary materials resulting from recycling processes

The Official Journal of the European Communities defines recovery and recycling as follow:

- **Recovery** means any activity that recovers either materials or energy from waste and includes recycling, composting and incineration with energy recovery.
- **Recycling** means the reprocessing in a production process of the waste materials for the original or other purposes, including organic recycling:
    - **Open-loop recycling** is when a product is recycled and undergoes a change of inherent properties, i.e. it is recycled into something completely different
    - **Closed-loop** recycling is when a product is recycled without altering its inherent properties. (The Official Journal of the European Communities, 1994)

The European Commission has suggested that the definition of recycling could be divided into two further categories in the future:

- **Mechanical recycling** is the reprocessing of materials for any purpose without changing the chemical structure of the processed materials.
- **Feedstock (chemical) recycling** is the reprocessing of materials for any purpose by changing the chemical structure of the processed materials. It involves the potential use of plastics as energy (as oil, energy is the main use). **(European Commission 2000)**

For the purpose of this thesis, recycling is defined as the reprocessing of waste materials for the original or other purposes, but without altering its inherent properties, i.e. the waste cardboard is recycled into pulp that can be used for packaging or other applications.

CHEMICAL SYMBOLS

As: Arsenic

Cd: Cadmium

CH_4: methane

Cl^- Chloride

CO_2: carbon dioxide

COD: Chemical Oxygen Demand

Cr: Chromium

Cu: Copper

F^-: Fluoride

H+: Hydrogen ions

HC: Hydrocarbon

HCl: Hydrogen chloride

HF: Hydrogen fluoride

Hg: Mercury

H_2S: Hydrogen sulphide

Mn: Manganese

N: Nitrogen

NH_3: Ammonia

NH_4: Ammonium

Ni: Nickel

NO_2: Nitrogen dioxide

NOX: Nitrogen Oxides

Pb: Lead

PM10: Particles with a diameter <10μm

PO_4: Phosphates

Sb: Antimony

Sn: Tin

SO_X: Sulphur Oxides

TSP: Total small particles

TSS: Total suspended solids

TOC: Total Organic Carbon

V: Vanadium

VC: Vinyl Chloride

VOC: Volatile organic compound

1 INTRODUCTION

1.1 Background

This research focuses on the issue of waste management. Society is faced with growing quantities of waste arising from mass consumption. The treatment and final disposal of waste through conventional means is associated with pollution and other environmental impacts; hence the present emphasis in improving measures to minimise and recycle waste materials. Waste management needs to become more sustainable: that is to be environmentally effective, economically affordable and socially acceptable (Warner Bulletin, 2000).

A waste hierarchy has been defined in which waste prevention is seen as being preferable to re-use, which is preferable to recycling, followed by energy recovery, incineration without energy recovery and finally landfill. The waste hierarchy was first introduced into European waste policy in the European Union's Waste Framework Directive of 1975. In 1989, it was formalised into a hierarchy of management options in the European Commission's Community Strategy for Waste Management (European Commission, 1989).

Waste minimisation at source is seen as the most important goal. If waste is reduced at source then less of it needs to be dealt with at the final stage of a product life cycle. However in term of legislation, only the Landfill Directive has so far set mandatory targets to reduce waste - the first target is to reduce biodegradable waste going to landfill to 75% of 1995 figures by 2010.

However, as technologies and understanding of pollution are getting better, there are increasing attempts to quantify the environmental, economics and social benefits and costs/impacts of all waste management options, regardless of their assumed place in the hierarchy (House of Lords, 1992-1993).

The European Commission decided that packaging waste would be its first target in an aim to reduce waste in general, as public perception towards that specific waste stream seems to be particularly significant. Several other producer responsibility type legislations were to follow. The landfill Directive came into force in 2002 – It reduces the amount of bio-degradable waste that can be landfilled and bans hazardous waste from most landfill sites. The End of Life Vehicle Directive came into force in 2003 and put the responsibility on the producer to organise recovery and recycling of vehicles. The Waste Electrical and Electronic Equipment (WEEE) came into force in 2004 and requires manufacturers of such products to finance their recovery and recycling.

The European Directive on Packaging and Packaging Waste (94/62/EC) set targets to recover 50%-65%, and recycle 25%-45% of all packaging waste, with a minimum of 15% for each material (paper/board, metals, plastics, and glass) category. Within the framework of the packaging directive recovery means any activity that recovers either materials or energy from waste and includes recycling, composting and incineration with energy recovery. Recycling means the reprocessing in a production process of the waste materials for the original or other purposes, including organic recycling.

The 15% minimum for each material was introduced to ensure that Member States would not recycle just a few materials to reach the targets such as glass or paper, but instead would have to deal with all materials. The low level of 15% was to acknowledge that some materials were better suited to recycling than others and that there are still some technological barriers for recycling others in large quantity (plastics for example). The targets were to be achieved by Member States by June 2001. However these targets, when decided in 1994, were the outcome of concessions on political grounds between the then twelve Members States, rather than being scientifically based, this is further discussed in section 2.1.2.2. The Directive's targets are to be reviewed every five years and the first review was in 2002. The present study is sponsored by the UK Department of Trade and Industry with the aim to develop new targets on a more scientific basis.

Several techniques exist for quantifying the environmental impacts of policy implementation. Life Cycle Assessment (LCA) is the most common, and it may be combined with Cost Benefit Analysis or Economic Valuation to present a fuller picture to decision-makers as discussed by (Pearce, 1999) and (Beukering, 1998). Although LCA is not a new method, interest in its use has increased since the early 1990s, resulting in both the development and the growing harmonisation of methodology. The first guidance on how to compile an LCA was published by the Society of Environmental Toxicology and Chemistry in 1991, and several guidelines were published afterwards. The International Standards Organisation (ISO) published a standard in 1997 and this is further discussed in section 2.2. LCA is largely being used as a supportive tool by governments for policy development and by Non Governmental Organisations (NGOs) and companies either to compare competitors' products or identify possible environmental improvements. Much is expected from the method, but at the same time its results are often criticised (Finnveden and Ekvall, 1998).

1.1.1 Sustainability and waste management

In order for waste management to become sustainable it needs to be environmentally efficient, economically affordable, and socially acceptable.

Environmental efficiency requires that the overall environmental burdens of managing waste be reduced, both in terms of consumption of resources (including energy) and the production of harmful emissions to air, water and land.

Economic affordability requires that the costs of the waste management system are acceptable to all sectors of the community served, including householders, commerce, industry, institutions and government.

Social acceptability requires that the waste management system meets the needs of the local community, and reflects the values and priorities of the society (White, 1998).

1.1.2 The need for a packaging targets optimisation model

Optimum targets can be defined as targets that would achieve the highest environmental improvements for a minimum economic cost and with the highest social benefits.

Optimising packaging waste recycling to balance environmental and social impacts, as well as economic costs, requires that these impacts and costs be predicted, hence the need to analyse packaging waste recycling systems. Indeed, very high levels of recycling are likely to prove unsustainable due to high transport distances, higher energy requirements as well as the economic costs of running separated waste collection. Hence the optimum level, in terms of sustainability, is likely to lie below 100%. A typical model to analyse a recycling system would have the following additional benefits:

- The process of building a model focuses attention on missing data.
- Once completed, the model will define the *status quo* of packaging waste recycling, both by describing the system, and by using life cycle assessment to calculate the overall environmental costs.
- Modelling allows 'what if...?' calculations to be made, which can then be used to define the points of greatest sensitivity in the system. This will show which changes will have the greatest effects in reducing environmental impacts and at what costs.

- The model can predict the likely environmental impacts and economic costs in the future. These are useful especially in such long-term processes as the development of markets for secondary materials. Market development is vital to ensure that higher levels of recycling can be sustained. Modelling packaging waste recycling will allow prediction of the likely amounts of reclaimed material available, which will in turn allow investment in the necessary equipment.

1.2 Aims and objectives

The research focuses on the targets set by the Producer Responsibility Obligations (Packaging Waste) Regulations 1997, which is the UK legislation to implement the European Directive (94/62/EC) on Packaging and Packaging Waste.

1.2.1 Original Aims and objectives

The original aim of the study was to establish sustainable target levels for recovering and recycling packaging waste in the UK. The following steps had been envisaged:

Use Life Cycle Analysis (LCA) techniques as part of a model to establish:

- The environmental benefits and environmental impacts associated with different levels of recycling targets for all type of materials;
- The level at which increased recycling ceases to make a noticeable difference to the overall environmental benefit of the process;
- The environmental benefits and environmental impacts associated with recycling of packaging waste compared with landfill and incineration.

It was originally intended to use Cost Benefit Analysis (CBA) techniques to:

- Establish a level at which the targets would be sustainable: maximise environmental benefits, minimise economic costs and impacts to society.

However, after presenting the project at the transfer viva, the examiners argued that the original aims of the projects were over ambitious and that concentrating first on establishing environmental impacts might be a more realistic goal. This was taken into consideration in addition to time limitations and access to relevant data. Furthermore, each type of packaging material has specific associated issues and data requirements associated with it. For this reason the decision was made to focus on cardboard waste. Cardboard was selected from amongst other relevant material types (plastics, metals, glass, etc) for the following reasons:

- High level of paper and board recycling is already taking place across Europe (up to 85%) and the UK (53% in 2001) (Commission, 2002)
- There is no scientific evidence of what environmental benefits are achieved (or not) at 53% recycling of cardboard packaging waste in the UK

- Cardboard packaging represents almost 60% of all paper and board packaging on the market (CEPI, 2005)

1.2.2 Revised aims and objectives

This study aims to assess the environmental benefits and environmental costs associated with different levels of recycling targets, specifically for cardboard packaging waste within the context of the UK. It will also seek to establish the level at which increased recycling ceases to make a noticeable difference to the overall environmental benefit of the process. Environmental benefits and environmental costs associated with cardboard packaging waste recycling will also be compared with 100% landfill and 100% incineration.

1.3 Structure of the thesis

The thesis is divided into six chapters, and the following is a brief outline of their contents.

Chapter one introduces the research, presents its aims and objectives. Chapter two is the literature review; the focus is on first presenting the political and regulatory context for this research, with an overview of environmental concerns and the general political agenda. It then presents the development of the European Directive (94/62/EC) and how it was implemented in the UK. The second part of the literature review concentrates on available methodologies to conduct the study, with first a detailed presentation of LCA principles and methodological issues, how it is being used in waste management as well as in other relevant political contexts. It then presents alternatives techniques that could have been used or are used in conjunction with LCA. The last part of the chapter presents the case of cardboard packaging waste.

Chapter three describes the paper and board manufacturing processes highlighting environmental impacts rising from specific steps.

Chapter four introduces the methodology and models developed for the study. LCA is used to assess the environmental impacts of several scenarios with different waste management options For the scenarios that include recycling, of the portion of waste not being recycling, 93% is landfilled and 7% incinerated, which represent the national average of final waste disposal. The selected scenarios are:

- Base scenario: 53% recycling, 4.23% incineration and 42.77% landfill
- Scenario 2: 60% recycling with 37.2% landfill and 2.8% incineration
- Scenario 3: 70% recycling with 27.9% landfill and 2.1% incineration
- Scenario 4: 80% recycling with 18.6% landfill and 1.4% incineration
- Scenario 5: 35% recycling with 60.45% landfill and 4.55% incineration
- Scenario 6: 100% landfill
- Scenario 8: 100% incineration

Chapter five presents the results for the modelled scenarios, and three sensitivity analyses. The first sensitivity analysis tests the assumption of all return transport made empty, the second one focuses on the assumption made for final disposal of waste, and lastly energy efficiency within cardboard manufacturing is tested. Chapter six discusses the findings of this project and looks at what other parameters should be taken into consideration when setting packaging waste recycling and recovery targets. Chapters seven and eight present the conclusion and suggestion for further work in the field of packaging waste recycling targets.

2 LITERATURE REVIEW

This chapter presents the findings of an extensive literature review. Section 2.1 concentrates on the background of the environment in politics, and how it came to be such an important issue. It looks at how the European Packaging and Packaging Waste Directive (94/62/EC) was developed and the lengthy process for it to be implemented in the UK law. It finishes with an overview of the Directive revision consultation and proposal, and the actual new targets. Section 2.2 focuses on environmental assessment methodologies. It outlines Life Cycle Assessment (LCA) principles, methodological issues are discussed, and other techniques such as multi-criteria approach and ecological footprint are also presented. Section 2.3 is an overview of economic tools available to combine with LCA.

2.1 Packaging waste legislation

2.1.1 European environmental policy

The introduction of environmental considerations into European policy happened progressively. The first reference to waste policy at a European level was made in 1971, when the European Commission declared prevention and recovery of persistent waste as a policy target. In 1972, at the Paris Summit the idea of an Environmental Action Plan was launched, and this was put into action in October 1972 for the period 1973-1976. Since 1976 more and more actions have been taken through policy to safeguard our environment.

Looking back at the last three decades the political, industry and legislative attitude has greatly evolved. Indeed in the 1970's the approach to protect the environment was one of a 'command and control', actions were only taken after pollution occurred, i.e. clean up, liability actions and introducing adequate protective regulations. In the 1980's, the approach was to develop environmental prevention, through the use of market instruments such as environmental taxes and regulations. In the 1990's the approach was a hybrid one, integrating the 1970's and 1980's approaches, sustainable development with long term planning, regulatory and economic tools, as well as using life cycle analysis to evaluate the environmental impact of products (Allemby, 1999).

'One of the EU's main principles is subsidiarity[1]. European regulations have to have added value; policy that is best made at national level should be made at this level.

[1] Subsidiarity is the principle that decisions should always be taken at the lowest possible level or closest to where they will have their effect, for example in a local area rather than nationally, In the context of Europe that means some decisions are best made by individual Member States rather than by European Commission/ Parliament

Every European decision has to be based on the Treaties that form the European Union, however when is an issue of Community interest and when is it not?' (Environmental News from Netherlands, 1996) Regarding the environment issue, and especially the waste issue, it has been criticised that the latter vary greatly between Member States and within each country, so that Europe should only formulate guidance and not legislate on these matters. Table 2.1 summarises key events and policy changes in the historical development of environmental issues in Europe.

Table 2.1: Historical development of main environmental policies in Europe

1957	1972	1986	1992	1999
1957 Treaty of Rome No specific attention to environmental issues. 1967 a Directive harmonised the classification, packaging and labelling of dangerous substances.	1972 declaration of Paris Summit: 'Economic expansion had to result in an improvement in the quality of life, in particular by paying special attention to environmental protection'. The Summit also launched the idea of European Action Plan.	1986 The Single European Act Provided specific legal basis for EC environmental policy with a whole section devoted to the 'Environment' 1987 Brundtland Report: 'Our Common Inheritance' developed the idea of sustainable development as a means to reconcile environmental protection and economic growth.	1992 Treaty on European Union (Maastricht) Introduced several 'environmental' amendments due to the international context (Earth Summit at Rio 1992)	1999 Treaty of Amsterdam was ratified: Limited changes introduced, although it was then accepted that economic growth has to happen in a sustainable way. Several other changes were made to ensure environmental protection was taken into account when developing new legislation.

Source: (Glocker et al., 1998), (WAMTEC, 2000)

2.1.2 Packaging waste legislation

2.1.2.1 Development of waste policy in the EU

The principal Directive controlling waste management throughout the EU is the Framework Directive on Waste (75/442/EEC), as amended by 91/156/EEC and adapted by 96/350/EC. The provisions contained in the Framework Directive of 1975 were transposed into UK law mainly by the 1990 Environmental Protection Act and the amendment of 1991, by the 1995 Environment Act, together with a number of regulations on various aspects of waste management (DETR, 1999). The aims of the Directive (75/442/EEC) are:

- The requirement for Member States to implement national waste regulation,
- The obligation for the licensing of waste treatment plants,
- Waste management is defined as prevention of waste arising, re-use, recycling and safe disposal,
- Establish the polluter pays principle.

(The Official Journal of the European Communities, 1975)

In 1985 the Council Directive (85/339/EEC) covered beverage container waste as part of the European Community strategy for waste management. The Council Directive's success was deemed due to the variety of different approaches adopted by Member States.

In November 1989 targets for waste reduction were first mentioned by an official of the Commission's Environment Directorate during a conference in London. The aim was for the Community to become self-sufficient in the management and disposal of all types of wastes. Waste minimisation in turn would have to become a central element. The aim would be to eventually set targets for waste prevention, recycling or reuse and treatment where appropriate to diminish the amount of waste going to landfill (ENDS, 1989).

2.1.2.2 Development of the European Packaging and Packaging Waste Directive

In 1992 it was highlighted that beverage packaging only accounted for around 5% of household wastes and that there was a worldwide solid waste disposal crisis. A sector-by-sector or one-sided approach was no longer appropriate and a comprehensive approach for dealing with packaging and packaging waste was needed (Bulletin EC, 1992). It was at that time estimated that 50 million tonnes of packaging wastes were produced in the Community every year, of which only 9 million tonnes (18%) were recycled, with the proportion varying widely from one Member State to another and between materials. Thus, for example Germany implemented its "Ordinance on the minimisation of packaging waste" in 1991, setting targets of 60% increasing to 80% recycling of packaging waste (Michaelis, 1995). France, the Netherlands and Denmark were also in the process to implement packaging waste regulations. In contrast England, Spain, Portugal and Greece had no national regulations or recycling/recovery targets specific to packaging waste.

It was in that context that in July 1992 the Commission adopted a proposal for a Council Directive on Packaging and Packaging Waste (COM (92) 278). The proposal defined criteria to be met by packaging in terms of composition and suitability for reuse and recovery. The aim of the proposal was to reduce the adverse impact of packaging on the environment (by recovering or recycling packaging), and applied to all types of packaging waste whether from household or industrial sources. This was later amended (in 1993) to include minimisation, re use and then recycling as the preferred options Targets were set for the Members States to be met within 10 years of adopting the proposal:

- Recovery (recycling, composting, regeneration, energy recovery) of 90% of packaging waste;
- Recycling (including composting and regeneration) of 60% of each material in packaging waste.

(The Official Journal of The European Communities, 1992)

The proposal also planned for the development of a harmonised system of databases concerning packaging waste. Provision of measures relating to the provision of information for consumers were also outlined, to instruct them how to dispose of their used packaging, and for collectors of waste to tell them which materials have been used and thereby facilitate segregation, collection, sorting, and recycling.

An Advisory Committee of Representatives of the Member States (later known as Committee 21) was established to assist the Commission in adapting the provision of the future Directive to scientific and technical progress, and to formulate a waste hierarchy.

The UK government agreed with the view expressed by the UK Packaging Chain Forum [2]that 'identical recycling targets for each packaging material are not only arbitrary they also ignore the different contribution that different packaging materials can make to achieve the overall targets. Some materials are better suited to recycling, others to energy recovery and some to reducing overall quantity of packaging waste' (ENDS, 1992). It was later decided by different countries to set different targets to match their own national situation.

Thus the targets were the results of a lengthy consultation process between the Member States (twelve at the time). As mentioned earlier the first proposal of recovery and recycling targets of packaging waste was in 1992. Table 2.2 highlights most of the different targets and time frames proposed between 1992 and the final Directive in 1994. The final targets might appear weak compared to the initial ones, however different levels of recycling and reprocessing capacity in Member States had to be taken into consideration. By setting maximum recycling targets the European Commission wanted to prevent Member States already able to collect large quantities of packaging waste but with limited reprocessing capacities flooding other Member States reprocessing market. Thus these rates were the outcome of political debate rather than being scientifically based.

Table 2.2 Summary of EU packaging targets development

	Recovery	Recycling	Time requirement
October 1992	90% (min.)	60% (min.)	10 years
June 1993	60% (min.)	40% (min.)	5 years
October 1993	50% (min.)	30% (min.) of all materials	5 years
December 1993	50% (min.)- 65% (max)	25% (min.) - 45% (max) with 15% (min.) of each material	5 years

[2] UK Packaging Chain forum represented all industries involved in packing: raw materials manufacturers, Converters, Packers/Fillers and Sellers.

2.1.2.3 The European Packaging and Packaging Waste Directive

The Directive (94/62/EC) was brought into force on 31 December 1994 by publication in the Official Journal of the European Community, and covered all packaging marketed in the EU and all households, commercial and industrial packaging wastes. The aims of the Packaging and Packaging Waste Directive are to:

- Harmonise national measures so as to prevent or reduce the impact of packaging waste on the environment of all member states and of other countries,
- Remove obstacles to trade and distortion, and restriction of competition,
- Prevent the production of packaging waste,
- Reduce the amount of waste for final disposal through packaging reuse, recycling and other form of recovery.

(The Official Journal of the European Communities, 1994)

Member States had 18 months to implement the Directive, and five years to take appropriate action to reach the set targets and requirements. The targets are mandatory and should have been reached by July 2001. They were set at:

- Minimum 50% and maximum of 65% by weight of packaging waste should be recovered,
- Minimum of 25% and a maximum of 45% should be recycled, with a minimum of 15% for each material to be recycled.

As agreed by the Council of Ministers (The Official Journal of the European Communities, 1994). Each Member State had the freedom to choose the targets they wished to achieve within their country, although they officially had to announce to the European Commission what targets had been decided (see table 2.3) and how the Directive would be transposed into national law. The targets individually chosen by each Member State reflected their own national recycling achievements and their potential to achieve higher rates.

Table 2.3: Targets established by Member States to reach for 2001

	Recovery	Recycling	Recycling for each material	Recycling targets for packaging materials					
				Paper and cardboard	Aluminium	Steel	Glass	Plastics	Beverage composites
Directives Target	Minimum 50% - maximum 65%	Minimum 25% - maximum 45%	Minimum 15%	15	15	15	15	15	15
Austria		25	15	90	95	95	93	40	40
Belgium	80	50	15						
Denmark				55	15	15	65	15	
Finland	82 (reuse/ recovery)			53	25	25	48	15	
France	50-65	25-45	15						
Germany	65	45		70	60	70	75	36	60
Greece	50-65	25-45	15						
Ireland	50-65	25-45	15						
Italy	50-65	25-45	15	45	35	38	48	17	
Luxembourg	55	45	15						
Netherlands	65	45	15	85	80	80	90	27	
Portugal	25-50	25	15						
Spain	50-65	25-45	15						
Sweden				40	90	70	70	30	15
UK*	52	26	16						

Source: (EC, 1999)
*The targets apply to obligated businesses[3] only

[3] Obligated businesses in the UK are those with a turnover exceeding £1 million and handling more than 50 tonnes of packing a year

2.1.3 UK legislation

The UK was one of the few Member States in the early 1990's that had no official scheme or plan regarding recycling or reuse of waste. It only had a voluntary programme to meet the UK's obligations under the Beverage Containers Directive. The government's White Paper 'This Common Inheritance' in 1990 outlined that the government wanted to encourage 'industry to reduce unnecessary packaging of consumer goods' (HMG, 1990). In March 1990, Chris Patten (Environment Secretary) announced at a packaging conference the government's hope to recycle 50% of recyclable household waste, that is around 25% of all household waste by the end of the 1990s. This was the starting point of a lengthy consultation process between the government and industry.

2.1.3.1 The consultation process

The government initially left the packaging industry to organise and regulate itself to meet the forthcoming Directive targets. This was known as the Packing Chain, and included packaging raw material manufacturers, packaging manufacturers (converters), packers/fillers and wholesalers/retailers. Several consortiums and groups came into action with different plans. 1992 saw the first industry effort through a proposition by the Consortium of the Packaging Chain (COPAC). However, in February 1993, the government rejected their final report mainly on the ground that there was no clear indication of how the targets were to be met, and how the funding would be raised.

In July 1993, the Environment Secretary requested the industry to draw a new action plan including financial provision. The Producer Responsibility Industry Group (PRG) was then created, in response to the new challenge, representing more than a hundred businesses concerned with packaging issues. It replaced the COPAC organisation, and submitted a report in November 1994, the main points of which were:

- On the basis of current data, recovery of about 58% of the UK's packaging waste (as against 30 % currently) was achievable by 2000, but not on a voluntary basis. Underpinning legislation was required to ensure compliance and to provide the necessary incentive to create business operated schemes to organise recycling and recovery;
- All parts of the packaging chain needed to be involved, from raw material manufacturers to retailers, if effective co-operation was to be achieved and recycling and recovery costs minimised. It was essential that business sectors

co-operated to increase end-use markets for recyclate and to stimulate investment in new reprocessing capacity while retaining a market led approach.

- There was a need for renewed commitment to waste to energy which was more appropriate than recycling for some packaging waste,
- Development of incentives for minimisation, for example through material-specific charging,
- A commitment for continuing consumer awareness and participation,
- VALPAK (Value from Packaging) an organisation together with sector specific materials organisation would be set up to put the plan into action,
- The funding would be organised through VALPAK, via a levy on packaging to meet the costs of the target (new collection, sorting facilities, industry investment). It was estimated that around £100 million would be raised in the 2-3 first years, which could rise to £300-500 million by 2000.

However no actual measures were taken in respect of increasing demand for recycled products.

The plan was supposed to divert packing waste from landfill from 4.9 million tonnes in 1993 to 3.4 million tonnes in 2000. At that time PRG projected that packaging consumption would increase by 10% over the next seven years from 7.29 to 8.05 million tonnes (ENDS, 1994). However the financial issues were still not addressed. The packaging chain was made up of four sectors: raw material manufacturers, converters, packers/fillers and wholesalers/retailers. The disparate nature of the packaging chain meant that each sector had different interests to promote and protect within a policy-making process, which led to a need for legislation rather than relying on voluntary agreement (Nunan, 1999). VALPAK was then put into place to implement the PRG plan.

In July 1996, after a consultation with businesses, organisation and local authorities, the majority of business responses favoured a shared approach to the legal obligation for the reasons set out by the PRG. This is known as the percentage activity obligation, which defines how the legal burdens is shared depending on the level of responsibilities attached that specific activities within the packaging chain. That action by downstream businesses (packer/fillers and retailers) was necessary to modify buying specifications and consumer attitudes, while upstream (converters and raw material manufacturers) needed to co-operate on providing processing capacity and outlets for collection material. As a result of the consultation the percentage activity obligation were adjusted to the shares in table 2.4.

Table 2.4: Shared responsibility in the packaging chain

	Percentage Activity Obligation	
	1995	1996
Raw material Manufacturer	5.5 %	6%
Converters	14.5 %	11%
Pack/fillers	35 %	36%
Sellers	45 %	47%

Source: (DoE, 1995); (DoE, 1996)

A threshold was also proposed during the consultation to exclude businesses handling 50 tonnes or less of packaging and packaging material per annum. This decision was based on a preliminary assessment: - A threshold at 50 tonnes excluded 90% of businesses from the obligation to recover and recycle - The 10% of business that would be obligated actually handled 94% of all packaging and packaging materials put on the market (DoE, 1996). The activity obligation only applies to obligated businesses (i.e. above the threshold).

The December 1995 agreement was modified in July 1996, the system proposed by the government was based on four guiding principles:

- Using the market to reflect the cost of dealing with waste in the price the consumer pays,
- A shared producer responsibility involving all parts of the packaging chain,
- The lightest regulatory approach giving maximum scope for obligated businesses to self-certify and self-assess their obligations and perform those obligations through business run schemes, or individually subject to monitoring by the Environment Agencies,
- Continuing industry involvement in monitoring, reviewing and proposing changes to the system through the Advisory Committee on Packaging.

In his review the Environment Secretary responded to the request of businesses to plan a slower build up to the Directive targets by amending the interim targets on individual businesses as in table 2.5. The government also decided to take a staged approach. The threshold was set such that businesses with a turnover below £5 million and who handled less than 50 tonnes of packaging would not be obligated in 1997-99. However from 1 January 2000 it would change and only businesses with a turnover below £1 million and who handled less than 50 tonnes of packaging would be excluded from the obligation; at the same date an obligation would apply to wholesalers. The Targets then, apply only to obligated businesses. To prove that industry was acting to reach the targets, businesses would have to register either with a compliance scheme i.e. VALPAK or could choose the individual route and then register with the relevant Environment Agency.

As seen earlier several targets were proposed during that time and these are summarised in table 2.5.

Table 2.5: Summary of the target development through consultations in the UK regulations

	Recovery	Recycling	Time requirement
1990[1]		25% of recyclable household waste	By the end of the 1990's
October 1992[2]		42% of all used packaging	By 1999
July 1993[3]	50%-75%		By 2000
November 1994[4]	58%		By 2000
Proposed in December 1995, and accepted for the final Directive in 1996 for the regulations[5&6]	98/99: 38% 99/00: 43% 00/01: 52%	7% 11% 16%	By the end of each year

(Targets only apply to obligated businesses)

1(ENDS, 1990), 2 (ENDS, 1992), 3(ENDS, 1993), 4 (Producer Responsibility Group, 1994), 5 (DoE, 1995), 6 (DoE, 1996)

After seven years of consultations and proposals the Directive was finally implemented through the Producer Responsibility Obligations (Packaging Waste) Regulation. Producer Responsibility is an extension of the 'polluter pays' principle, aimed at ensuring that businesses take responsibility for the products they have placed on the market once those products reach the end of their life. Producer responsibility is an alternative to taxation or traditional regulation. By placing at least some of the costs of their products on producers, Producer Responsibility schemes give producers an incentive to design products that minimise waste, or that can be reused or recycled (DTI, 2000). Several other producer responsibility obligations are coming into force in the UK such as the End Life of Vehicle, Batteries waste, Electronic and Electrical Waste.

2.1.3.2 The Producer Responsibility Obligations (Packaging Waste) Regulation

The Producer Responsibility Obligations (Packaging Waste) Regulation was implemented in March 1997. Its set the recovery target at 52% and the recycling targets at 16% for each materials. Interim targets were agreed (see table 2.5 above) to ensure that recovery and recycling activities gradually increased, but also to allow time to develop new facilities to cope with waste collection to ensure targets would be reached by 2001. Shared responsibility was the main principle of the UK legislation, ensuring that all participants of the packaging chain were taking responsibility for the packaging they were putting on the market.

The Regulation put four main obligations on obligated businesses:

- Register with the relevant Environment Agency or compliance scheme and provide relevant data on packaging flows,
- Recover the obligated tonnage of packaging waste calculated on the previous year of weight of packaging handled by the company. If registered with a compliance scheme then the business's obligation is fulfilled,
- Have proof that the obligations have been met,
- Retailers have a duty to increase consumers' awareness about their role in increasing recovery and recycling.

(Statutory Instruments, 1997)

By May 1997, plans for a number of other collective schemes besides VALPAK had emerged, some specialised in one material and others would develop dealing mainly with smaller business after 2000 when the threshold for turnover would change.

The Environment Agency (EA) introduced the Packaging Waste Recovery Notes (PRNs) system in May 1998. PRNs are issued by the EA to approved reprocessors (Cooper, 2000). Any obligated businesses or compliance scheme that deliver packaging waste to an approved reprocessor are given PRNs to confirm tonnages that have been reprocess. This is the only way an obligated business or compliance scheme can prove it has fulfil its obligation under the Producer Responsibility Obligations (Packaging Waste) Regulation. However if an obligated business or compliance scheme recover more packaging waste than its obligation, then the extra PRN can be sold to other businesses or compliance. Obligated businesses and compliance scheme on behalf of their members, must buy sufficient PRNs to meet their recovery obligation for the year. The revenues generated by the PRNs are to be invested in new waste collection, sorting and reprocessing capacity to ensure that the UK businesses meet the EC targets (Bailey, 1999)

The targets to be reached by 2001 were partially achieved by the UK, the 25% overall recycling was fulfilled, however the overall recovery level felt short by 2% at 48%.

2.1.4 Revision of the Directive

By 1999 most Member States had transposed the Directive into national law. Although one of the aims of the Directive was to harmonise the European approach to waste management, in practise this was not achieved. Indeed, Member States have implemented the Directive at different pace and in ways best suited to their national conditions. Thus the recovery schemes for used packaging differed in the expectations with regards to funding and ongoing cost. At an early stage of consultation Member States had very divergent views of how the revision should head.

As per article 6.3(b) no later than six months before the end of the first five years phase referred to in paragraph 1(a) the European Council shall by qualified majority and on a proposal from the European Commission, fix targets for the second five years phase referred to in paragraph 1(c). This shall be repeated every five years thereafter (The Official Journal of the European Communities, 1994). The proposal of the European Commission on the revision of the Packaging and Packaging Waste Directive in December 1999 included two options:

Option 1: - 90% by weight for total recovery
- 60% by weight for recycling of each material of packaging waste

Option 2: - No recovery target
- A minimum of 60% by weight of total packaging waste to be recycled
- 75% by weight for glass
- 65% by weight for paper and board
- 55% by weight for metals
- 22% by weight for plastics

The problem with option one was the same as when it was first proposed in 1992 in the initial proposal. It would be hard and expensive for certain materials i.e. plastics to reach such a target, and that some materials are better suited to incineration than recycling. In addition, there was still no scientific demonstration of the environmental benefits of the targets.

In option 2, the no recovery target was due to experience showing that high recovery targets resulted in the promotion of incineration.

There was also the debate on the financial implication with ever-higher targets. A Pricewaterhouse Cooper study stated that packaging waste only represented 3% of total solid waste in the EU. Recovering half of those packaging waste was estimated to cost 10 billion Euros: 'Is it worth raising the recovery target and maybe doubling the bill?' (European Packaging & Law, 2000)

One of the main changes is that the material specific targets are now different for each material. This highlights the fact that different materials can be recycled at different levels due to technological limitations or the level of environmental impacts.. A summary of the different proposals between 1999 and 2003 is presented in table 2.6.

Table 2.6: Summary of the proposed revised targets

	Deadline	Overall Recycling	Overall Recovery	Glass	Paper	Metals	Plastics	Wood
1994 Directive	2001	50-65%	25-45%	15%	15%	15%	15%	none
Commission proposal	2006	60-75%	55-7%0	60%	55%	50%	20%	none
1st reading Common	2008	60-75%	65%	60%	55%	50%	20%	none
Position revised	2008	60%	50-80%	60%	60%	50%	22.5%	15%
Commission Proposal	2008	60%	50-80%	60%	60%	50%	22.5%	15.5%
2nd EP[4] reading	2008	60%	55-80%	60%	60%	50%	22.5%	15%

The UK position on the revision of the Directive was that no change should happen for 2002-2006. The UK argued that no real study had taken place yet to demonstrate the validity of the targets regarding diminishing the environmental impact packaging waste had. The UK considered it best to commission proper scientific research to help evaluate what had been reached so far and how the different waste management methods compared to each other and whether real improvements in reducing environmental impact was taking place (Materials Recycling Week, 2000)

[4] EP: European Parliament

The Directive's targets were reviewed and agreed in July in 2003 with the aim being to increase recycling levels. The new targets are to be reached by December 2008 and are:

- Minimum 60% recovery,
- Minimum 55% recycling with:
    - 60% for glass
    - 60% for paper and board
    - 50% metals
    - 22.5% plastics
    - 15% wood

2.2 Life Cycle Assessment methodology

This section focuses on the environmental assessment methodology used to conduct the modelling of selected scenarios. Life Cycle Assessment (LCA) was selected by the research sponsor as the most appropriate methodology. Its concept and principles are largely explained in this section, followed by a discussion on methodological issues discussed in the literature. The second part of this section deals with environmental economics and valuation techniques that can be used with LCA.

2.2.1 Definition

Life Cycle Assessment (LCA) was first used in the 60s. Then an LCA was more about energy analysis that the all input-output analysis. Some of the very first published studies focused on energy requirements for alternative packaging in the early seventies, for examples see Boustead (1972) and Sundstrom (1973).

The concept is based on the ability to study the whole life cycle of a product, service or policy from 'cradle to grave'. This includes accounting for all input and output crossing a defined system, including energy, materials, resources and emissions to air, water and land, including waste disposal. The input and output are then related to environmental impacts. The International Standard Organisation (ISO) developed an international standard for the LCA technique in 1997 (ISO 14040). Prior to that several organisations had published general guidelines to apply the method, the most recognised being 'A Technical Framework for Life Cycle Assessment' published by SETAC in 1991. It defined LCA as:

'A process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and materials used and wastes released to the environment; to assess the impacts of those energy and material uses and releases to the environment; and to identify and evaluate opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing extracting and processing raw materials; manufacturing; distribution and transportation, use, re-use, maintenance; recycling and final disposal' (SETAC, 1991).

The ISO standard 14040 now prevails over any other guidelines as the method to use to develop LCA. It defines LCA as:

'The assessment that includes the entire life cycle of the product, process or activity, encompassing extracting and processing raw materials, manufacturing, transportation and distribution, re-use, maintenance, recycling and final disposal' (ISO, 1997).

2.2.2 Development of LCA

LCA was developed in the 1960's; it has mainly been used in "design for the environment" allowing engineers and designers to improve products without shifting environmental burdens along their life cycle. At the turn of the millennium LCA is being used in the development of products, activities and policies.

Waste management practices aim to become more sustainable, environmentally aware and to apply the Best Practical Environmental Option (BPEO) wherever possible. Yet, it can be a daunting process to evaluate all aspects of waste management activities. In order to determine the total environmental, economic and social impacts of waste management systems, it is possible to use the technique of LCA. Since the early 1990's LCA has been increasingly used in waste management policy development by local authorities, central government and the European Commission (Environment Agency, 1997).

Public LCA studies are used to support the development of environmental legislation and regulation, development of criteria for environmental taxes, standards, or Eco-labelling programmes, or to provide consumer information. In the private sector, companies can use LCA results to support product development or marketing, to enhance the credibility of the company's environmental policy, or to guide suppliers to act in more environmentally friendly manners (Miettinen and Hamalainen, 1997). Two types of LCA can be conducted:

- Awareness-raising or exploratory LCA is conducted with the aim of an increased understanding of the complex system investigated and its environmental burdens,
- Prospective or comparative LCA is conducted with the purpose to compare two products for improvement or to understand different scenarios and where their environmental burdens lie (Ekvall and Finnveden, 2000).

The prime objectives of carrying out an LCA are to:

- Contribute to the understanding of the overall and interdependent nature of the environmental consequences of human activities (Life cycle impact assessment, LCIA). This is the most important and most difficult part of an LCA. LCIA focuses on emissions (to air, water, waste) and their contribution to particular environmental impacts such as global warming.

- Provide as complete a picture as possible of the interactions of an activity with the environment (life cycle inventory, LCI). LCI identify and quantify all inputs and output of a system.
- Supply decision-makers with information, which defines the environmental effects of these activities and identifies opportunities for environmental improvement (Interpretation), (Consoli et al., 1993).

Within an LCA, the LCI offers a clear and comprehensive picture of the flows of energy and materials through a system and gives a holistic and objective basis for comparisons. The LCIA quantify all potential environmental impacts (to land, water, air) of a product system over its life cycle. The results can identify opportunities for product improvement and indicate environmentally friendly options where a comparison is made. The results may contribute in targeting the more significant environmental impacts and the stage of the life cycle, which they relate to.

It is however very important to emphasize that an LCA will not give the answer to a problem, but will inform decision-makers of where and to what level environmental impacts are occurring.

2.2.3 Structure of LCA

LCA is a time and data intensive method. It takes place in four defined phases: Goal and Scope definition, Life Cycle Inventory (LCI), Life Cycle Impact Assessment (LCIA) and Interpretation. Each step has a methodology outlined in ISO 14040. Figure 2.1 outlines briefly the interaction between the different stages.

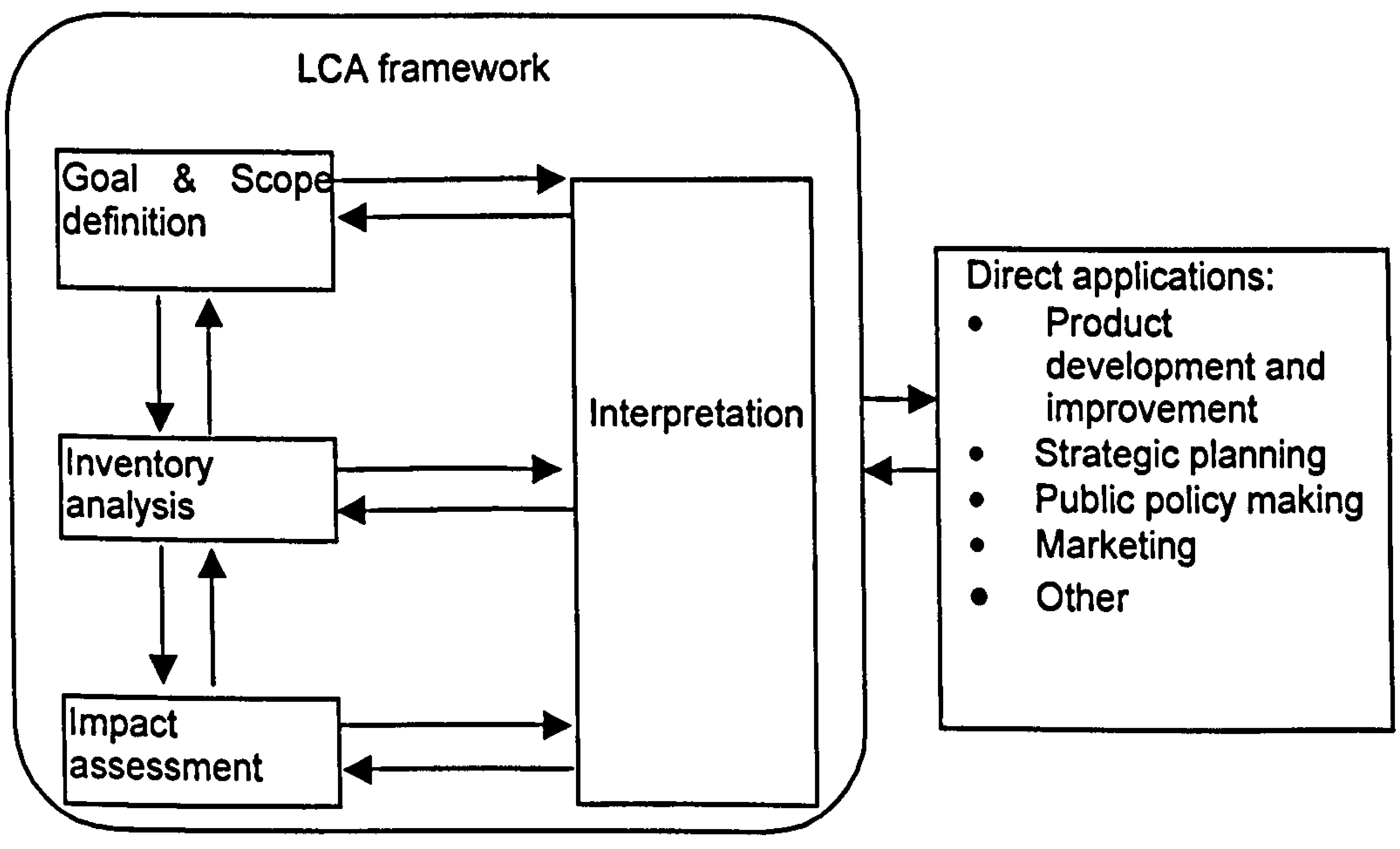

Figure 2.1: Structure of LCA, source (ISO, 1997)

2.2.4 Goal and scope definition

This is the first step of an LCA and will state the intended application for conducting the LCA, the reason for carrying out the study and the intended audience. The scope of the study has to be well defined to ensure that the breadth and depth of the analysis are compatible with the stated goal, and sufficient to address it (Consoli et al., 1993). In addition three other aspects need to be defined accurately (but might be amended later on) at this stage:

- The functional unit: This is the unit of the product/service whose environmental impacts will be compared. It has to be clearly defined, measurable, and relevant to input and output data (Guinee, 2001). It is often expressed in terms of the amount of product, such as a tonne of municipal solid waste.
- The system boundaries: define what is included within the assessment, and what is omitted. As shown in figure 2.2 the outline of the box denotes the 'system boundaries' and includes the 'foreground system' (what is studied) and its surroundings called 'the background system'. The background system is the source of all inputs to the system and the sink for all outputs from the system (Beukering et al., 1998). In the case of packaging waste management the foreground would be the processes within waste management (collecting, sorting and disposal). The background system would include processes such as grid electricity production and raw materials. The boundaries exclude the use of packaging before it became waste. The system boundaries need to reflect the inputs and outputs that will be included in the inventory (defined below). These will be determined by the environmental issues a study wishes to address. The following boundaries need to be considered: geographical, life cycle (should the mining of the raw materials be included (cradle) and should the emission from final disposal (landfill) be considered (grave)), technosphere and biosphere. Due to the subjectivity of the process, the transparency of the defined process and the assumptions made need to be clearly documented (ISO, 1997), (Lindfors et al., 1995).
- Data quality requirement: shall be defined to enable the goals and scope of the LCA study to be met. The data quality requirement should address: time, geography and technology coverage; precision, completeness and representativeness of the data, as well as consistency and reproducibility of the methods used throughout the LCA. Data sources should also be acknowledged with their representativeness and/or uncertainty (ISO, 1997).

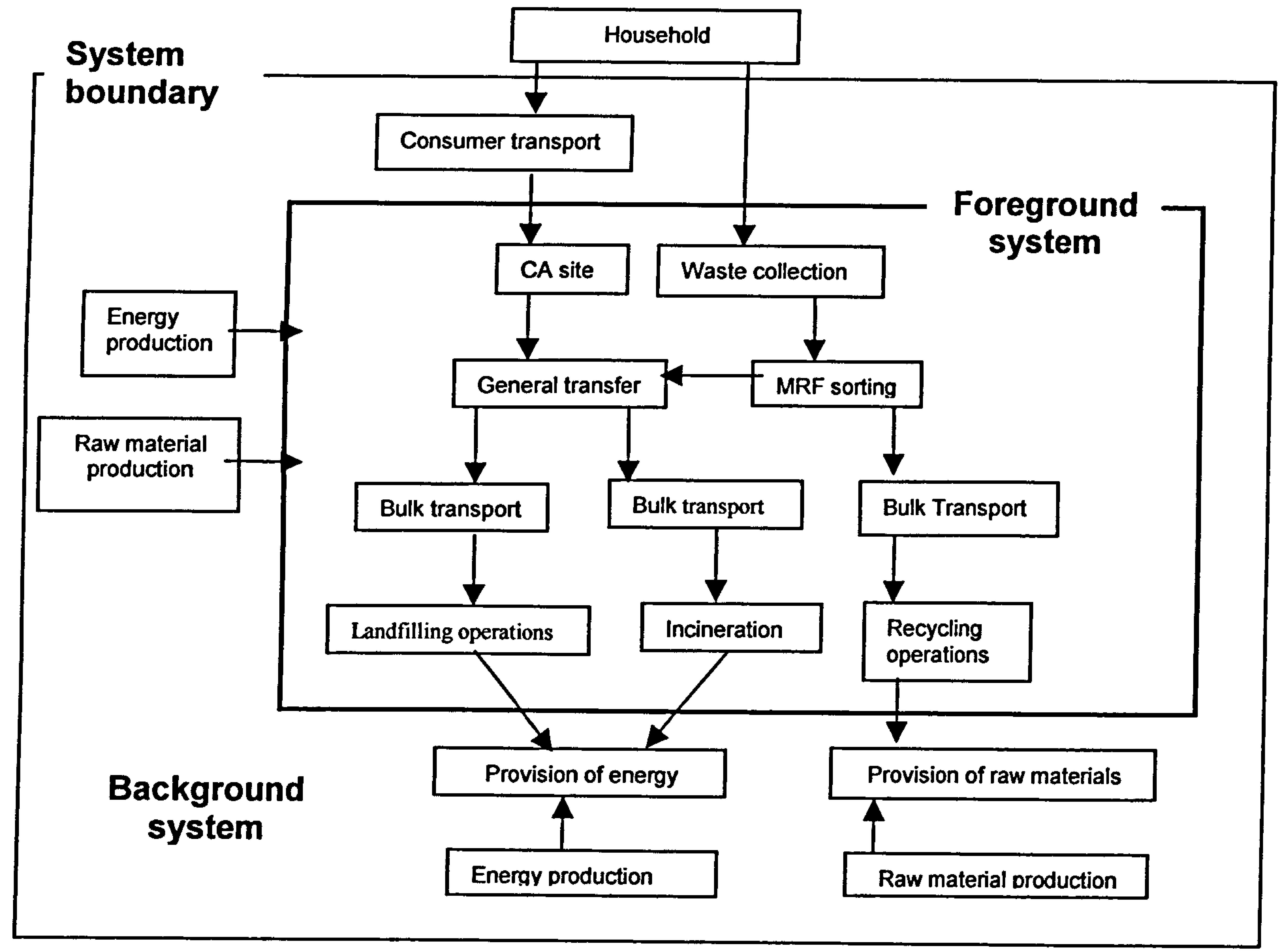

Figure 2.2: System boundary for waste management, source: (Beukering et al., 1998).
CA = Civic Amenities
MRF = Material Recycling Facilities

2.2.5 Life Cycle Inventory (LCI)

Once the system boundaries and the functional unit have been defined, there is a need for all inputs (raw materials and energy) entering the system and all outputs (emissions to air, water, land and waste) leaving the system to be quantified. The inventory will account for all these along the life cycle of the system under study, by dividing the life cycle into a series of steps or process trees, and then calculating the inputs and outputs for each of the steps/trees (White et al., 1995). Another issue the inventory will look at is the predicted amount of useful products that might arise from waste, such as compost, secondary materials and useful energy, which will then replace raw materials or national grid energy. The total life cycle inventory is thus:

- Direct burdens: arise from the operations being studied (foreground system), and could be directly affected by the decisions based on the study.
- **Plus** indirect burdens: arising in the supply chains of materials and energy provided to the activity being study (background system) and are not directly affected by the outcome of the study.
- **Minus** avoided burdens: associated with economic activities displaced by material and/or energy recovered. (Clift et al., 2000).

2.2.6 Life Cycle Impact Assessment (LCIA)

Life cycle impact assessment is the third step of LCA and is still controversial. It associates each input and output with a particular environmental issue (e. g. global warming), and evaluate the significance of potential (not actual) environmental impacts. The level of detail, choice of impacts evaluated and methodologies used, depend on the goal and scope of the study. There are no generally accepted methodologies for consistently and accurately associating inventory data with specific potential environmental impacts.

The need for impact assessment depends on the purpose and results of the study. In a comparative study, it may be that one alternative is better than all other alternatives on all environmental burdens in the inventory, and thus the conclusion is easy to reach. However, more often than not, one scenario will do better on some environmental burdens but worse on others, thus it might be desirable to attach some degree of importance to the environmental burdens.

The impact assessment method is described in ISO 14042, and a distinction is made between:

- Obligatory elements: such as classification and characterisation
- Optional elements: such as normalisation, and valuation (also known as weighting).

The last step of impact assessment, valuation is still under development and no recognised methodology has been accepted yet. Still several techniques are available and being widely used.

According to the ISO standard if characterisation and classification do not take place then it is a Life Cycle Inventory and not an LCA.

Life cycle impact assessment is divided into four stages, for which several methodologies are available and discussed in this section, each step is an analysis process of the inventory data.

2.2.6.1 Environmental impact category definition

General categories of environmental impacts needing consideration include resource use, human health and ecological considerations (Clift et al., 2000). In the literature several studies were found to concentrate on the same range of impacts, namely global warming – acidification - eutrophication – human toxicity – ecotoxicity and photo-oxidant formation - Bloemhof-Ruwaard et al. (1996); Johnson (1993), Hanssen (1998) and Heijungs (1992). However these are also fairly standard set of mainstream impacts studied through LCA.

2.2.6.2 Classification and characterisation

Classification assigns the inventory data results to relevant impacts to which it may potentially contribute to (global warming, acidification), the scale (local, regional or global), and media (water, air or land).

Characterisation quantifies the relative contribution of individual environmental emissions to an impact category, by using pre-defined characterisation factors. This is achieved by grouping together all potential contributable emissions to an impact, and by multiplying the measures of the relative equivalence with the pollutant load the contribution of each pollutant to the impact category is calculated (see figure 2.3) (Beukering et al., 1998a). For example, the reference emission for global warming is CO_2. The weighting applied to CO_2 is therefore one. All other emissions, which contribute to global warming, are weighted relative to their CO_2 equivalence. The effect of global warming caused by 1kg emission of methane is 21 times greater than the one caused by 1 kg of CO_2. Therefore methane is given an indicator factor of 21. For each impact category there can be several methodologies available to characterise burdens into impacts, for example the most commonly used methodology for greenhouse gases effect is the one developed by the Intergovernmental Panel on Climate Change, which entails the calculation of equivalent values of different contributing emissions to CO_2. Table 2.7 presents environmental impacts most relevant to waste management (Jensen et al., 1997).

2.2.6.3 Normalisation

Normalisation can be used to relate the results of the classification/ characterisation steps to total emissions in a certain area over a given period. This procedure shows to what extent an impact category has a significant contribution to overall environmental problems. Normalisation increases the comparability of data from different impact categories, and provides a better basis for weighting options (Beukering et al., 1998) by relating the magnitude of impacts in different categories to reference values; an example of a reference value could be the total contribution to an impact category by a nation (Clift et al., 2000). However normalisation only reveals which effects are large in real terms, it does not provide any information about the relative importance of these effects.

2.2.6.4 Valuation methodologies

Characterisation of the results is only a partial aggregation of the results. It classifies the inventory results enabling decision-makers to have an understanding of the potential environmental effects of the emissions quantified during the inventory. Yet the ability to base decision-making on characterised results alone can be limited by the need to determine which impacts carry most importance (weight) in the particular decision.

The valuation/weighing step rates the importance of different impact categories against each other. This is achieved by converting indicator results for selected impact categories to a common scale by using numerical factors based on values. This can potentially include a final aggregation to a single indicator. The impact assessment data may then be converted to monetary values through the application of economic valuation to each environmental impact category. Weighting has always been a controversial issue, in large because it is a value laden expression of relative severity requiring social, political and ethical values, whereas the preceding steps are based on traditional natural sciences (Clift et al., 2000). There are still no agreed methodologies for this last step.

Several methodologies are available to complete the valuation step. The two most widely used are based on two different approached: the mid point and end point approach. The two approaches are discussed here and are the one used in the modelling for the present study.

2.2.6.4.1 Mid – point approach

The midpoint approach is characterised by the type of categories and final indicators used for the impact assessment stage. It will generally result in categories such as global warming, eutrophication, and ozone depletion. The indicator unit will be relative to the flow potential equivalence, such as that of CO_2 to global warming. This allows drawing a very detailed picture of the environmental interaction of the system being studied (Hertwich et al. 2002). However, one drawback of the methodology is that for decision makers it might be hard to visualise or understand what the CO_2 equivalent might entail. Thus, stakeholders are left with a detailed list of potential environmental impacts and how they compare to each other but without real interpretation in terms of potential damage. The methodology that is used in the present study is CML developed by the Centre for Environmental Studies (CML) at Leiden University of the Netherlands, and is based on a scientific understanding of the effects of pollutants on the environment (Heijung et al, 2003).

- The CML method uses the midpoint approach and aggregates the results to give a single indicator value based on the scientific knowledge. It gives a weight to impact category based on their concentration in the environment and the severity of their potential impact (Heijung et al, 2003). The results after valuation are still presented as environmental effects such as global warming, eutrophication. This approach is seen as 'more accurate' than the endpoint techniques as it is calculated using scientific understanding of emissions effects to the environment (Goedkoop et al, 2002). However the results stay relatively abstract with regard to the units they are reported in, which can make it difficult for stakeholders to reach a decision.

Other methodologies using midpoint are: Ecoscaricity is a valuation system where emissions are multiplied by eco-factors, these are calculated from the annual consumption of a given flow in a specific area, and the maximum acceptable yearly flow for the same area. The EDIP works on a similar basis, but handles environmental impacts and resources consumption on work environment as separate categories (Pira, 2000)

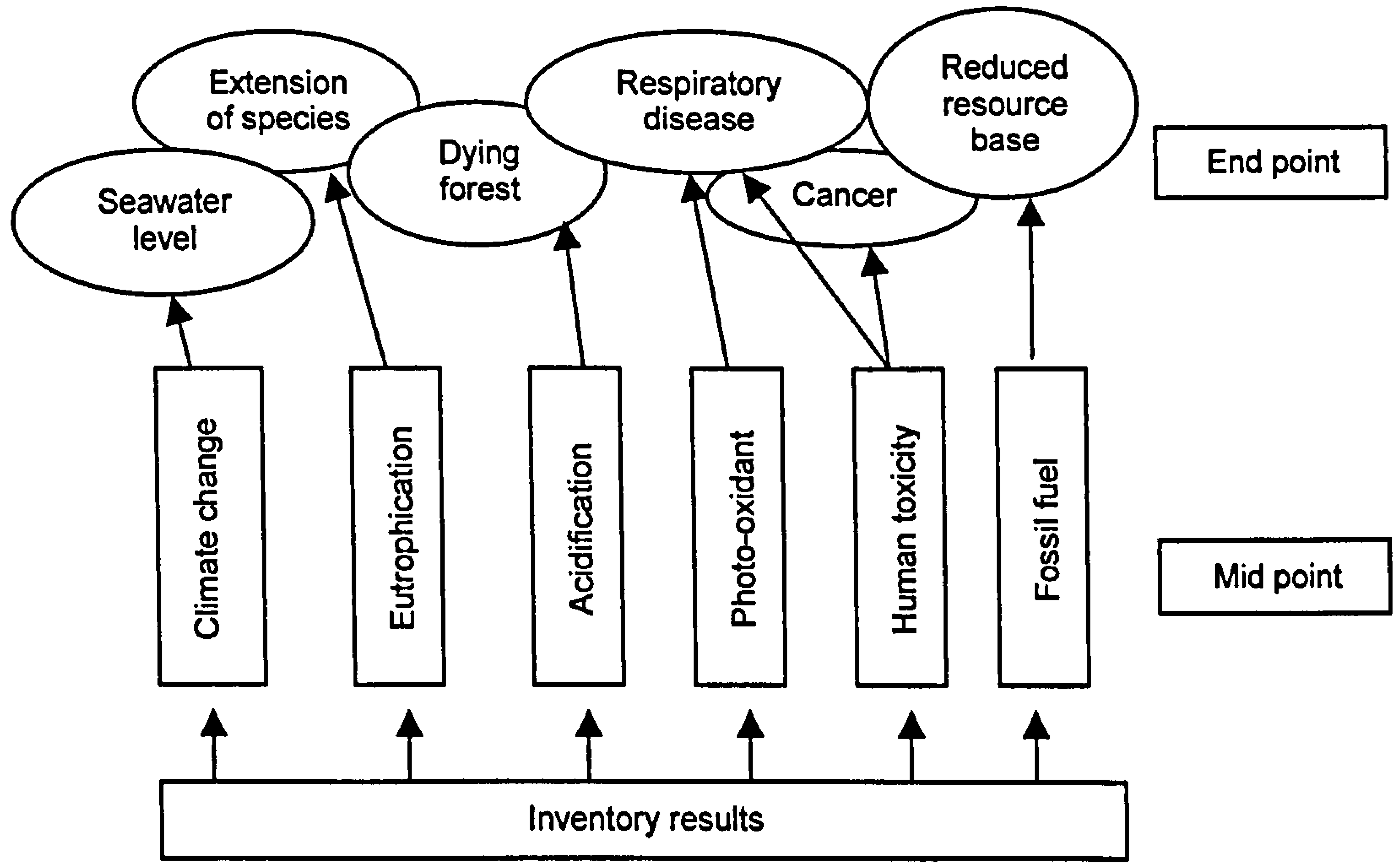

Figure 2.3: Emissions pathway from inventory to midpoint and endpoint stages, source (Goedkoop and Michiel, 2001)

2.2.6.4.2 Endpoint methodology

Endpoint methodology attempts to predict damages and then aggregates predicted damages in terms of a single indicator. This is achieved through the use of a panel by applying economic valuation, as well as societal and ethical values of environmental damages. The damages it includes are those to human health, ecosystem quality and resources. These damages can be the result of one or several impacts at a time (Goedkoop and Spriensma. 1999). Thus, while the midpoint methodology will stop at quantifying global warming potential or eutrophication potential, the endpoint methodology will translate these into potential damages to health or the ecosystem. The units used as measures of health damage are such as years lived disabled (YLD), years of life lost (YLL) or disability adjusted life years (DALY). For ecosystem quality, this is as a percentage of species that have disappeared in a certain area due to environmental load. For resources extraction, it is as a parameter indicating the quality of the remaining mineral and fossil resources. For example, human health impacts associated with climate change can be compared with those of ozone depletion using a common basis such as disability adjusted life years (DALY) (Hertwich and Hammit, 2001).

However (Heijung et al., 2003) stress that damage approach methodologies are at present, unable to comprehensively quantify all impacts with high levels of certainty. Existing data may allow a prediction of potential impacts, such as skin cancer or cataracts, but the data does not currently support the inclusion effects such as crop damage, and marine life change. Yet this does not mean that they should not be dealt with. Indeed the 'precautionary principle' (UNEP, 1992) requires that *'Where there are threats of serious or irreversible environmental damage, lack of full scientific certainty shall not be used as a reason for postponing cost effective measures to prevent environmental degradation.* The endpoint methodology chosen for the study is Eco-Indicator 99 for which data are mainly relevant to Europe (Goedkoop and Michiel, 2001).

- Eco-Indicator 99 is a damage-orientated methodology. The classification and characterisation of the inventory calculates potential damages to human health, potential damages to ecosystem quality and potential damages to resources. The characterisation results are typically in units such as loss of human life, disability adjusted years. The weighting applied to aggregate the results as one value is based on expert panel. The weights were developed by using a questionnaire sent to 365 identified (82 responses) experts who have an interest in LCA. The aim of the questionnaire was to obtain "statistically significant differences between damage categories" (Goedkoop and Spriensma, 1999). The outcome of the panel was to establish for a specific damage category (characterisation) relations such as "human health is more important than ecosystem quality and that ecosystem quality is more important than resources" (Goedkoop and Spriensma, 1999). The panel also had to assign a weight to the damage categories. By aggregating all the answers of the panel the weightings were set. While the midpoint methodology will stop at quantifying global warming potential or eutrophication potential, the endpoint methodology will translate these into potential endpoint damage to health or the ecosystem (Goedkoop et al, 2002).

Another well-known endpoint method is the EPS 2.0 (Swedish Environmental Priority Strategies). This valuation technique is based on willingness to pay to restore five safeguard subjects to their normal status. These are biodiversity, production capacity, human health, resources and aesthetic values. Emissions are then valued according to their estimated contribution to the changes in the safeguarded subjects. The damage models are based on world averages (Bengtsson and Steen, 2000).

2.2.6.5 Interpretation

This is the final phase of an LCA, which analyses the interaction and results obtained from previous phases, with the aim of reaching conclusions and providing recommendations. LCA results can be applied in different situations such as:

- Identify major differences in potential environmental impacts between systems.
- Select the types of impacts caused by a system that can potentially be improved when compared with a reference system.
- Evaluate the environmental options given by the choice between different alternatives, e.g. identifying potential environmental benefits of an alternative.
- Identify impact categories that are not significantly affected by the choice between potential alternatives.

(Lindfors et al., 1995)

2.2.6.6 Sensitivity analysis

As for any modelling sensitivity analysis needs to be conducted. A Sensitivity analysis will involve testing the robustness of some or all or the assumptions underlying a model (Guinee, 2001). This type of analysis will give a better understanding of the magnitude of the effect the assumptions made. The outcome of an LCA can be heavily dependent on the assumption. This should not be a problem as long as the conclusions of the LCA are stable, i.e. the same option is still identified as the most favourable (Goedkoop and Michiel, 2001). The sensitivity analysis can also be used to test the impact of data source onto the results i.e. if generic databases were used, how are the results affected if more local data were to be used.

Table 2.7: Environmental impact categories relevant to waste mangement

Impacts categories	Scale of effects
Energy Depletion Potential: Extraction of non-renewable energy carriers; can be included in Abiotic Depletion Potential.	Global
Global Warming Potential: Contribution to atmospheric absorption of radiation leading to increase in global temperature.	Global
Ozone Depletion Potential: Contribution to increase in ultraviolet radiation reaching earth's surface through depletion of stratospheric ozone.	Global
Aquatic/Terrestrial Ecotoxicity: Contribution to health problems in flora and fauna caused by exposure to toxic substances.	Continental/ Regional
Acidification Potential: Contribution to acid deposition onto soil and into water.	Continental/ Regional
Human Toxicity: Contribution to human health problems through exposure to toxic substances via air. Water or soil (especially through the food chain).	Continental/ Regional
Photochemical Oxidant Creation Potential: Contribution to formation of atmospheric aerosol particles forming photochemical smog.	Regional/Loc al
Eutrophication Potential: Contribution to reduction of oxygen concentration in water or soil through providing nutrients, which increase production of biomass.	Regional

Source: adapted from (Beukering et al, 1998), (Clift et al, 2000), (SAFEL, 1998a) and (McDougall, 2001)

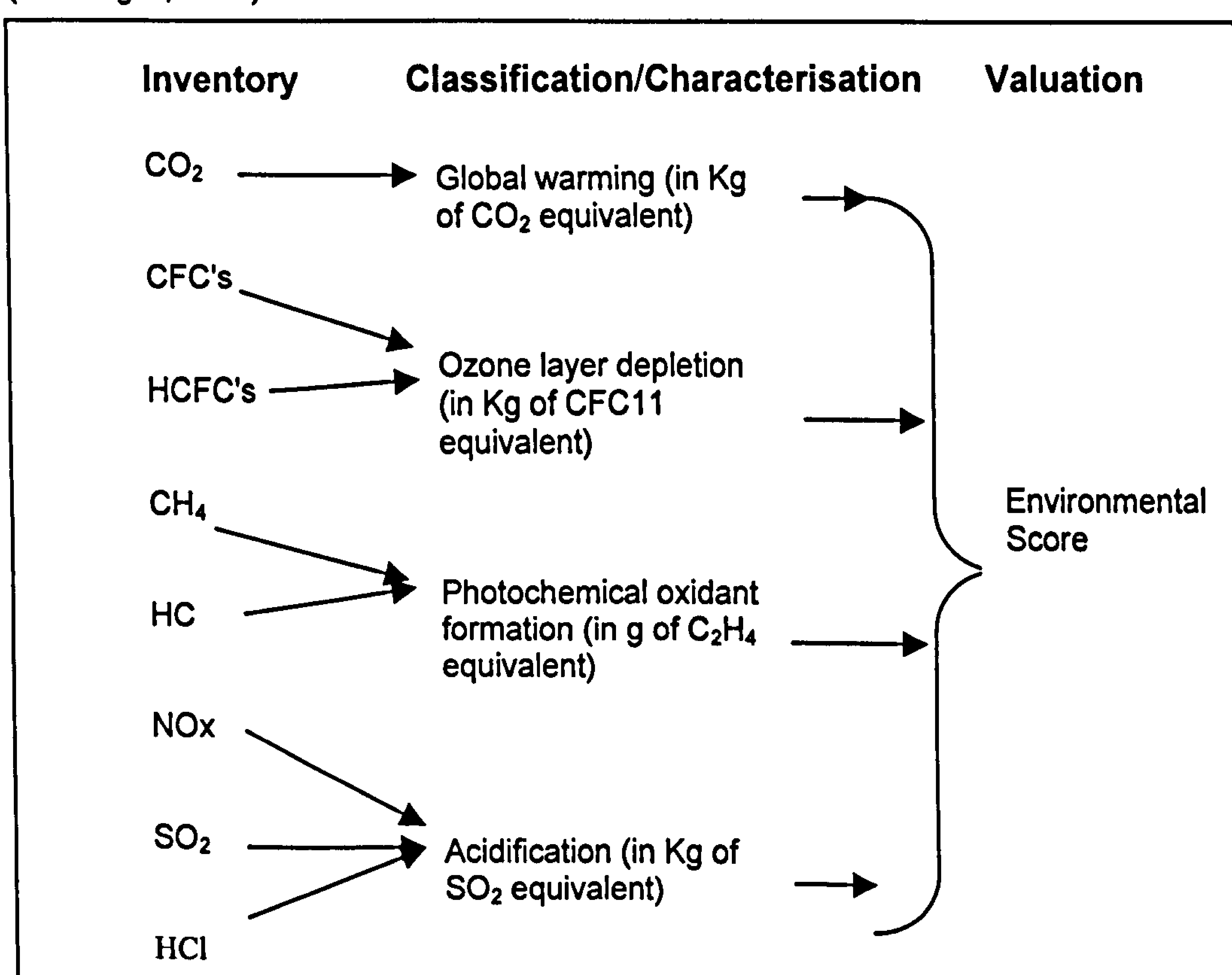

Figure 2.4: Impact assessment stages in LCA (Dobson et al., 1998)

2.2.7 Methodological issues associated with the use of LCA

A large amount of literature exists on the use of LCA, and for the purpose of this research the review concentrated on the use of LCA as a decision tool in waste management planning and recycling. Many publications were found that outlined the principles of LCA and how to use the results, but did not approach methodological issues, such as Jensen et al. (1997), Aumonier et al (1997), Clift et al. (2000), Environment Agency (1997), Miettinen et al. (1997) and Powell (2000).

The literature review found that methodological problems concentrated on boundary systems such as Finnveden (1999), McDougall et al. (2001) and Hanssen (1998), allocation methodology within LCI were discussed by Guinee et al. (2004), Huppes (1994) and Lindfors et al. (1995), weighting factors and the results applications by Bengtsson et al. (2000), Hertwich et al. (2002) and (Goedkoop et al. 1999). These are discussed below.

2.2.7.1 Boundary setting

Finnveden (1999), McDougall et al. (2001) and the European Environmental Bureau (2000) discussed the issue of boundary setting when dealing with waste. They focused on where the cradle and grave should be set when conducting a LCA of waste. The general approach of LCA is to identify and quantify flows from cradle to grave, that is flows that are drawn for the environment and discarded to the environment without subsequent human transformation. Finnveden's argument is that for a LCA on waste, this is typically not done. Instead, the inputs are solid wastes as they appear i.e. from household stream, and become the cradle for the LCA. However, this entails that all waste have to be originated from a similar source otherwise enlarging the boundary will be necessary to include the differences. In the case of a waste based system, the previous stages of a 'product' are not included in the system.

Hanssen (1998) reviewed eighteen product based LCAs with the intention to identify the main environmental problem related to types of products and the life cycle stages that are the most important sources of environmental impacts. One of his conclusions was that comparing LCAs results could be rendered impossible because even if the products are like for like, the boundary of the system and its assumptions are different. For example in the case of a light bulb, whether the use phase is within the boundary or not, will completely change the outcome of a study. Another typical example being the LCAs conducted on behalf of Procter and Gamble (Little, 1990) and the Women's Environmental Network (Landbank Consultancy, 1991), both on disposal vs. washable nappies which reached opposite conclusions to which one was best for the environment. This would be due for example to whether energy use for washing and transporting re-usable nappies are included or not, what waste disposal is chosen for the disposal nappies, if packaging is included or not.

Porteous (1997) discusses how an LCA's results should be used and how sensitive conclusions are to what is included or not within a system boundary. He also emphasizes the danger of using weighting factors, and the conclusion that can be drawn from their application, referring to two famous case studies (described above) on disposable versus washable nappies.

2.2.7.2 Allocation

Once all inputs and outputs for one system have been compiled there is a need to allocate which output is due to which process. In reality it is the case that one industrial process rarely yields one product only. Thus the environmental impacts of one process need to be allocated to all relevant products/system it is part of. . ISO 14040 (ISO, 1997) set a methodology to conduct allocations:

- Step 1: Where possible, the use of allocation should be avoided by:
  - Dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes;
  - Expanding the product system to include the additional functions related to the co-products.
- Step 2: Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned based on physical causality such as mass or energy content. Lindfors et al. (1995) favour such approach and argue that if the mass flow of a product A is ten times higher than the mass flow of a product B then ten times more of environmental impacts are allocated to product A compared to product B.
- Step 3: Where physical relationships alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way which reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products. For example Guinee et al. (2004) and Huppes (1994) argue that the economic value of the product justifies its existence, i.e. if two product (A and B) results from a same process and A is worth only 25% of product B value, then product B should bare 75% of the environmental impacts of the process they share..

However, the ISO procedure has been criticised because it does not take into account the fact that different approaches to the allocation problem result in different types of information, nor does it take into account the relation between the method and the study goal.

Allocation procedures have to be applied in three cases: multi-output processes, multi-input processes and open loop recycling (see definition in glossary).

Where no changes occur in the inherent properties of the recycled material in closed loop recycling, the need for allocation is avoided as the use of secondary material displaces the use of virgin (primary) materials (SETAC, 1994). However in an open loop recycling (see figure 2.4) where the material is recycled into other products and undergoes a change of inherent properties there are two choices. Either the system boundary should be expanded to include all products resulting from a process/activity. Or by allocating shared unit processes, based on physical properties, economic values and the number of subsequent uses of the recycled material (Finnveden and Ekvall., 1998).

Bystrom and Lonnstedt. (2000) prefer the options of system expansion rather than allocation within an LCA. When comparing incineration to recycling of waste paper, they argue that by using a larger amount of waste paper as raw material, the paper industry has modified its requirement in materials and energy. Higher use of waste paper means less biomass for energy and thus requires the process to depend on more polluting energy from the national grid, however if waste paper is incinerated the energy purchased by society can be reduced, which will have a positive effect on CO_2 emissions. All these processes should be included in the system rather than being dealt with through allocation which will then not credit the system with energy generated from the incineration process. Ekvall and Tillman. (1997) as well as Finnveden (1999) also discussed the issue and what should be the key factors to ensure a proper allocation procedure. They all concluded that whenever possible allocation should be avoided (as mentioned in ISO 14040) and system boundary expansion preferred. However, where allocation cannot be avoided the following issues should be considered:

- What raw/virgin material is being replaced by recycled one,
- What recycling process is being used,
- How are the recycled materials being used and how many times will it be re-recycled,

- Which type of energy (fossil fuel, renewable) is being used during the recycling processes,
- How the raw material saved, by using recycled materials, is being used (or not).

Newell and Field (1998) consider similar issues and have developed an accounting-based solution using a dimensionless parameter "*k*" that can be defined as the ratio of load allocated per kg of the primary material, for when allocation cannot be avoided. There does not seem to be any procedure that proves that any specific method is the 'correct' one. Instead arguments are usually based on what intuitively seems reasonable or fair.

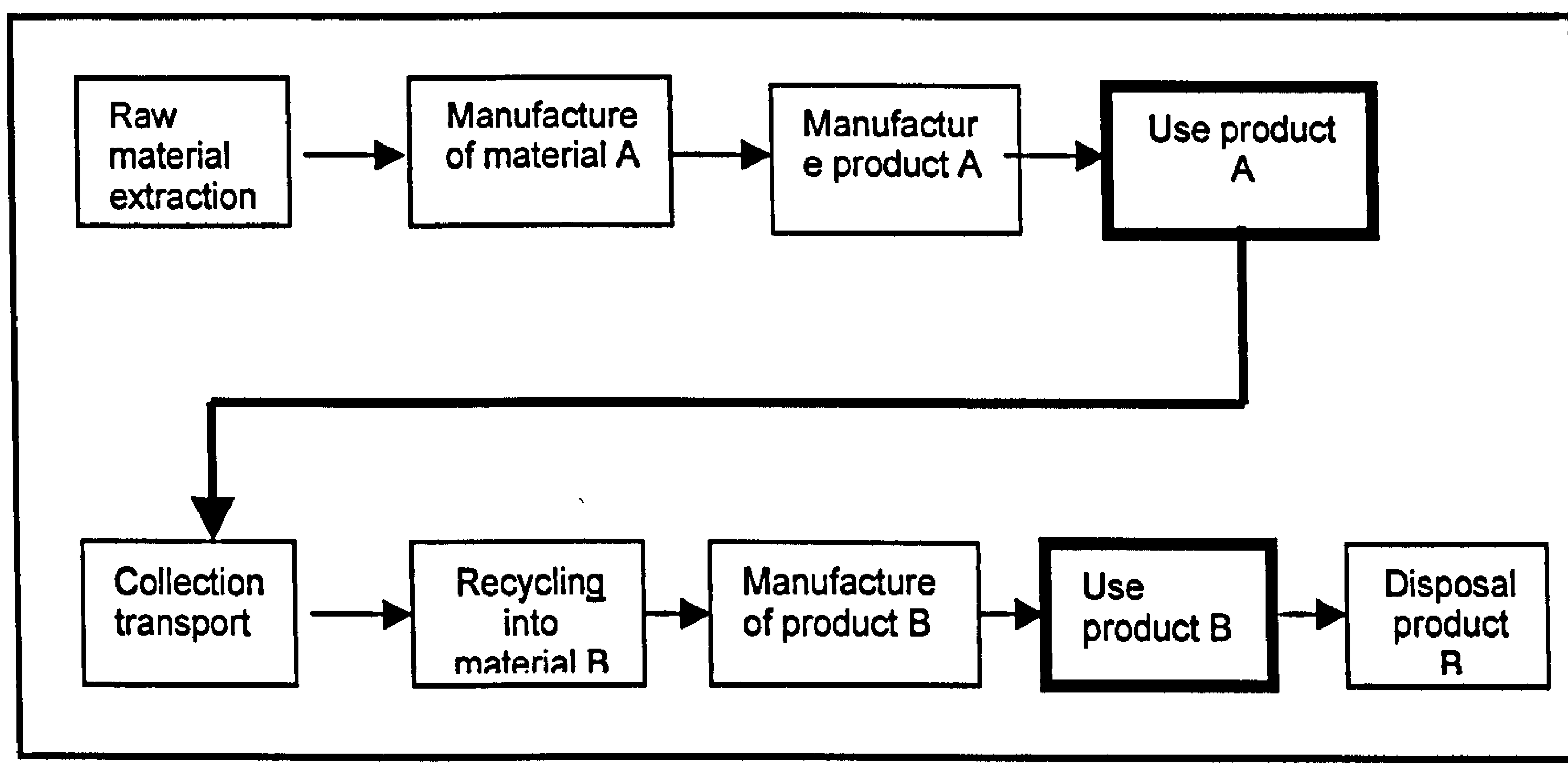

Figure 2.5: An open loop recycling System, source (SETAC, 1994)

2.2.7.3 Weighting/valuation

This is the most subjective stage of an LCA, it aims to simplify the results by converting them into a homogeneous unit. It is based on value judgment and is not scientifically based. ISO 14042 (life cycle impact assessment standards) cautions that different individuals or organisations may have different preferences or values, therefore different parties are likely to reach different weighting results based on the same characterisation indicator results. Any scheme that weights or aggregates the impact categories to a single score may appear to ease the decision making process. However, the assumptions and priorities upon which the decision will be based are obscured and only the values or opinions of one group are used. Thus, broad acceptance of the outcome of the decision is uncertain. Furthermore ISO 14042 highlights that impact assessment is designed to support better decision-making, but Life Cycle Impact Assessment does not replace the decision-making process. Bengtsson and Steen (2000) justify the use of weighting because while we can expect a reasonable consensus to form for e.g. the acidifying potential of SO_2 in a certain region, we cannot hope for such a consensus when it comes to the relative importance of e.g. threats to biodiversity as compared to threats to human health. Such a comparison of effects that are fundamentally different, inherently involve an element of subjectivity. Yet, this does not mean that they are completely random or arbitrary.

Several methodologies have been developed and the weights applied are all different in their scope and the value judgement used to implement weights. The two most distinguished types of valuation methods used in today's LCA field are the midpoint approach (also known as the environmental themes or impact orientated approach) and the endpoint method (also known as the damage approach). Figure 2.5 illustrates the pathway of emissions from inventory to midpoint and endpoint stages. The most clearly identified differences between them are the approach to the assessment of impact, and how valuation is integrated into the analysis (Goedkoop and Michiel, 2001).

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2.2.8 LCA and waste management

Within the waste management industry and the political agenda on handling waste, LCA is being used at many levels. The industry employs it as a management tool to form decisions. Politicians (EUROPEN, 2002) use it to develop awareness such as the Environment Agency's programme on LCA (Environment Agency, 1997), but also to inform policy development and target settings. Many authors have discussed the application of LCA specifically to waste management, either as an activity or for strategy and policy development, such as McDougall et al. (2002), Newell and Field (1998), Aumonier et al. (1997) and Barlaz et al. (1995)

Some studies concentrate on the wider issue of waste management; McDougall et al. (2002) published an updated version of White et al. (1995) integrated waste management model, which is primarily based on life cycle inventory (LCI), to be used to develop waste management strategies including all aspect of waste treatment from collection and sorting to recycling, landfill, thermal treatment and biological treatment. Newell and Field (1998) address the problem of recycling and how it is adjusted in LCI when a material is recycled several times or used in different products at the same time. The Environment Agency (1997) launched a Life Cycle Inventory for waste management research programme in 1994, with the main aim being to develop software. WISARD (Waste Integrated Systems Assessment for Recovery and Disposal) was launched in December 1999 with its main target uses being Local authorities. Aumonier and Coleman, (1997) reviewed the EA program and emphasized the fact that waste management needs to be integrated (i.e. no one single method of waste disposal can deal with all materials in waste in an environmentally sustainable way) and aware of cost-benefits of decisions. They concluded that LCA offers a robust framework for examining the options that might be the Best Practicable Environmental Option (BPEO) for individual waste types, and in specific circumstances for mixed wastes, but also to compare the performances of alternatives strategies. Barlaz et al. (1995) present the American Environmental Protection Agency project on integrated waste management using LCA which is very similar to that of the UK's EA.

Waste management today is following the waste hierarchy, of prioritising minimisation, followed by re-use and recycling, the final options being incineration with energy, incineration, and lastly landfill. However it can be argued that the hierarchy might not always represent the Best Practicable Environmental Option (BPEO). Indeed, as discussed by (White, 1998), some options are less appropriate for dealing with certain waste types, and, in practice a mix of management techniques will always be required to manage a range of wastes. It may also prove environmentally beneficial to manage all wastes together in a manner different from those which appear optimal when each waste stream is considered individually. Politicians have acknowledged these limitations. Aumonier and Coleman (1997) mentioned that in the drafting of the Packaging Directive, the original prescriptive '...recycling is preferable to incineration with energy recovery...' is now qualified by allowing modification of targets '... if LCA shows other processes to have environmental advantages...' (Article 4).

A few authors concentrate on the issues of waste transport and type of collection; Craighill and Powell (1996) looked at the environmental impact of kerbside collection and subsequent use of recycled materials versus waste disposal systems where wastes are landfilled and primary raw materials are used in manufacturing. They further incorporated an economic valuation of the identified environmental impact. Recycling came as the better option. Bell (2000) presents a study that investigated through the use of LCA and CBA, the alternative transport of waste available for Greater London concentrating principally on Inland waterways. This relates to a more holistic approach of transport planning at a local and national level, and the congestion problems faced by UK drivers. His conclusions were that water transport would significantly reduce environmental impacts linked to road transport but also help in easing congestion in urban areas. He also concludes that LCA and CBA are useful tools to inform decision-makers.

LCA could be used to determine the optimal recycling rate (i.e. achieve the highest environmental improvements for a minimum economic costs but with the highest social benefits) to meet defined environmental objectives, as it would allow a full picture of what is happening and where, and so identify where in the life cycle the most environmental benefits can be achieved. With respect to the target of recycling, 50% of the recyclable household waste, announced by the UK Government in 1990, (HMG, 1990), Pearce and Warford (1993) asked "why 50%? Why not 100% or 20%?" Pearce et al's views are that 50% was the outcome of a balancing act designed to consider both the views of environmental groups and 'green consumers' on one side, and industry and business on the other, rather than a clear attempt at identifying the best or 'optimal' level. To determine the optimal level of recycling it is necessary to consider costs and benefits associated with recycling. Contrary to common belief, not all recycling is beneficial. Figure 2.6 shows the quantification of environmental impacts of collecting and sorting of materials in an Integrated Waste Management (IWM) system (A), and converting the recovered materials from the IWM system into recycled material for further use (B). A+B gives the value of environmental impacts of recovering materials from IWM. Though, when recycled material is used virgin materials usage is reduced and thus saving the environmental impacts (C). Therefore the net overall impact of waste management including reprocessing = A+B-C.

There are cases where the associated environmental impacts exceed the environmental benefits, Speaking about recycling targets, White et al. (1995) said that they are useful for measuring progress towards a goal in general. However when evaluating recycling goals, targets may not always measure progress towards environmental objectives. Recycling is a way to reduce and recover raw material and energy consumption. Rigidly set recycling targets may not produce the greatest environmental benefits. Environmental benefits do not increase with recycling rate. Indeed a European study (European Commission, 1996) suggested that, if the volume of recycling were to increase, the net environmental benefits would fall in many Member States because the average tonne of 'future' recycled materials will contain less of the materials that are easiest to recycle. Thus, the marginal environmental benefits decline as the quantity of recycling increases. The marginal environmental benefits would be the extra level of environmental reduction per extra tonne of materials collected. However, for Member States with low recycling levels, will at first reduce environmental impact by increasing recycling until it reaches an optimum. Indeed, at high levels of recovery, proportionally more energy is needed to collect used materials from diffuse sources, so there is little, if any environmental gain.

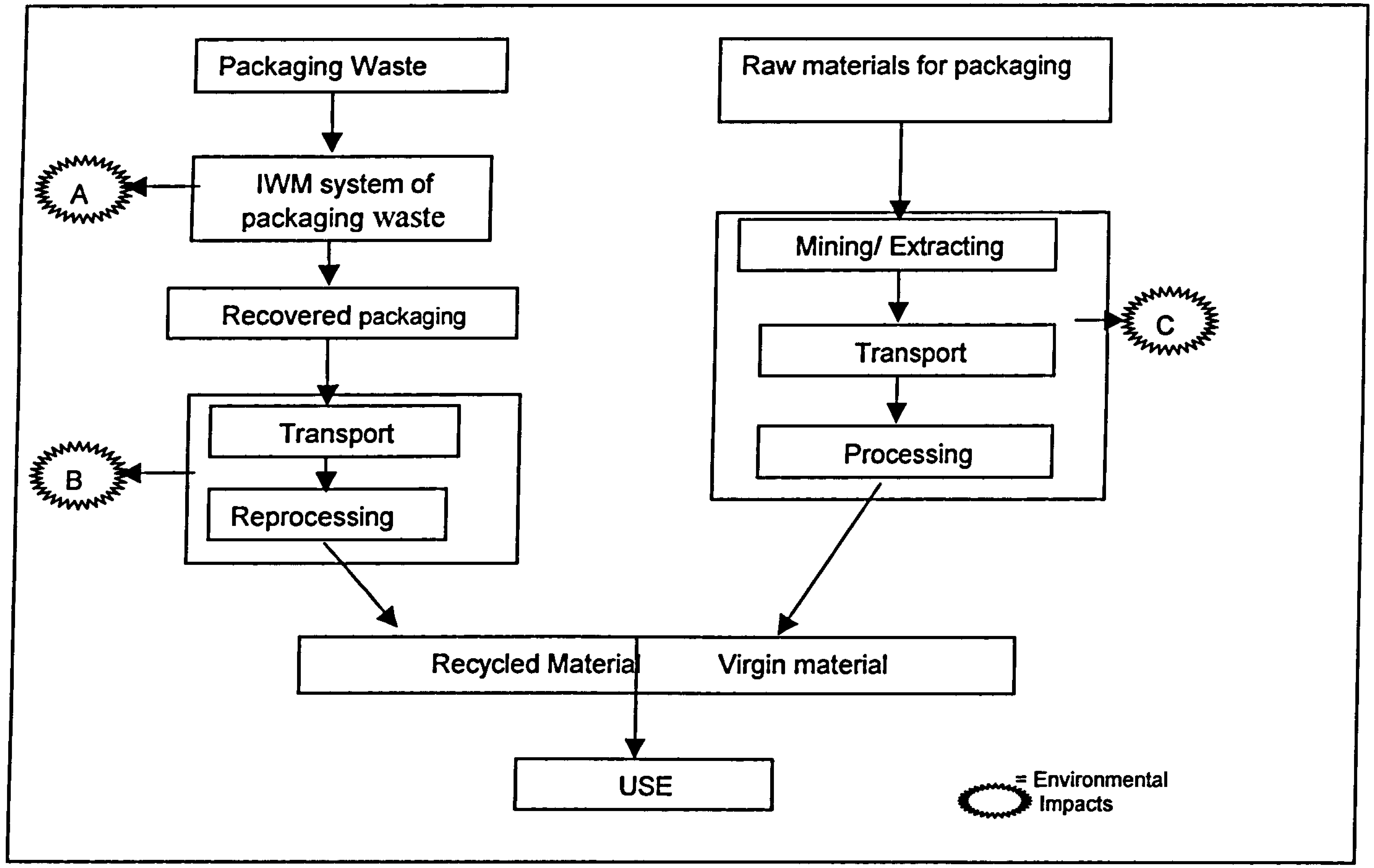

Figure 2.6: Recycled material life cycle versus virgin materials life cycle, adapted from (White et al., 1995).

2.2.9 LCA in the context of waste paper and the setting of targets

An ample literature exists on paper, pulp and newsprint, some of which focus on waste paper. Yet, very little relevant publications were found specifically regarding paper packaging waste.

An article by Leach (1998) provides a swift overview of the issue of using LCA for waste management, but is also valid for other applications, such as boundary setting, assumptions concerning technology and processes.

Finnveden and Ekvall both work extensively on using LCA in the context of paper and pulp, and waste paper. They reviewed several studies highlighting the methodological problem within LCA, explaining why different studies have reached different results when comparing recycling versus incineration of waste paper (Finnveden and Ekvall, 1998). The divergences in methodological approaches are mainly based on the modelling of energy production such as energy (from national grid) replacement by energy resulting from incineration included as well as modelling of land-use related impacts of forestry and the alternative use of forest area. They presented the results of one study, which modelled 100% incineration vs. 70% recycling with 30% incineration of newsprint waste (in the context of Sweden). Here waste incineration was assumed to replace energy from oil combustion and the alternative use of land was non-commercial. With these assumptions, the CO_2 and SO_2 emissions were 25% and 50% lower in the recycling case than for 100% incineration. However if the energy production is excluded from the analysis, the ranking is reversed. Another study they reviewed concentrated on energy sources. In one recycling scenario, the competing energy source was assumed to be 50% industrial waste and 50% average Swedish district heat. The net total methane emissions were negative because methane from industrial waste at landfills are reduced in this scenario. Although the fossil emissions CO_2 were much higher than in the case without recycling, the total global warming potential (GWP) is less than half the GWP without recycling. In the other recycling case studied, the competing energy source was assumed to be oil. The net total methane emissions were much higher than in the first recycling case, and the total GWP results were slightly higher than in the case without recycling. Another issue they identify is related to the selected methodology for allocation in the case of open loop recycling and incineration with energy recovery. Johnson's (1993) study asks the same questions but concentrates specifically on newsprint waste. One of her conclusions was also that assumptions and boundary settings largely influence the outcome of a LCA study.

Hunt (1995) focuses on paper and plastics waste and uses LCA to assess their impacts if landfilled, incinerated or used in composting. The main issues he identified are greenhouse gas emissions and leachate, and the problem of integrating emissions release over time (such as for landfill) in a model. Virtanen and Nilson (1993) approaches wastepaper in particular, and the idea of integrated waste management as the way forward. Indeed a single view approach focusing on one material or one type of waste treatment be it recycling, incineration or landfill, is limited and unrealistic. Today's approach needs to calculate what level of integration between the different waste treatment options need to be implemented. Landfill and incineration play an important role as much as recycling and should not just be seen as last options. They suggest that the objective of an efficient material production and recycling scheme should not only be to recycle, but should rather be to minimise the resource utilisation and emission of all streams of materials from cradle to grave. To identify such optimal schemes, it is necessary to consider many different alternative arrangements for material management, because the advantages gained in one respect might easily be lost in another. One of their conclusions was that recycling wastepaper does not reduce environmental impact per se. Rather, the type of environmental damages occurring change, such as fossil fuel increase and the emissions linked with it, as more energy is required if recycling is taking place rather than landfilling. Bystrom and Lonnstedt (2000) also mentioned that a policy focusing primarily on recycling might have wider consequences such as a change in the waste mix. This would influence the overall waste management approach, and the fact that due to higher paper recycling, paper manufacturers would use more energy from the grid, as biomass energy might no longer be readily available. This implies that recycling would have an impact on greenhouse gas emissions, whereas incineration with energy recovery would reduce the reliance on fossil fuel. Porteous (1997) concentrated on using LCA to establish a waste management hierarchy, and the fact that today politicians are reluctant to promote energy from waste even though the regulations on emissions are very strict.

LCA is becoming widely accepted as a support tool to inform for policies and strategies development, and a number of studies have been commissioned to better understand how policies might best be implemented. Thus, in 1996 the European Commission contracted Coopers & Lybrand (1997) to "Evaluate the potential of state of the art techniques such as eco-balances and LCA for policy making purposes and in particular to establish a global hierarchy on packaging and packaging waste". The study concentrated on establishing the best hierarchy for packaging waste but did not specify to what level each option should be used. The outcome for paper was that recycling should be promoted only if high demand for the recycled fibres existed. The study's geographical application was the EC. Powell (2000) presents the experience of Gloucestershire waste collection authorities in using LCA to establish their local strategy. The main finding was that LCA helped in understanding the environmental impact of options used for the strategy and to ensure that the best one was implemented first.

Materials organizations are also interested in using LCA to promote levels of recycling they believe are optimum. APME (Association of Plastics Manufacturers in Europe) used an LCA study (Eggels et al., 2001) to try to demonstrate what is the maximum value of recycling for plastic wastes (15%). However the study was widely criticised, as it was not made fully available to the public. Weaver et al. (1997) studied the European Pulp and Paper sector and considered how environmental policy might influence or limit its development. He observed that environmental policies are dictating holistic industry development paths, competitive structures, and international paths. However there is a risk that more stringent and holistic environmental policy might stop or reduce technology and environmental improvement, by locking in wrong technologies, i.e. by encouraging ever higher recycling levels no investment and research would be done in incineration or primary manufacturing. The conclusion is that recycling should be increased as far as possible, but hinted that if incineration technology were to improve then perhaps a more integrated policy would be more advantageous in environmental terms.

Although a large number of studies were found that dealt with waste paper, very few specifically dealt with paper packaging and optimum level of recycling. The EC commissioned a study to evaluate the costs and benefits for the achievement of reuse and recycling targets for the different materials in the context of Directive 94/962/EC (RDC and Pira, 2003), which used LCA and Cost Benefit Analysis (CBA). The purpose of the study was to investigate the technical feasibility, the environmental impacts and economics costs and the benefits of various options for reuse and recycling targets in the framework of the Directive. The study analysed a number of packaging applications with respect to the costs of reuse and recycling as well as the environmental benefits on the basis of life cycle.

The main conclusions were:

- For the European targets:
    - Overall recycling rates should be between 50% to 68%
    - Paper and Board: 61% to 71% in low population density areas, 55% to 65% in high population density areas.

- For the UK targets:
    - Overall recycling: rates 49% to 69%
    - Paper and Board 61% to 71% in low population density areas; 55% to 65%, in high population density areas and 80% for industrial waste. This gives a national average of 60% to 75%.

This study concentrated on packaging once it has become waste, whereas the present study looks at the whole life cycle of paper packaging.

All the above studies are useful in their own right, but none have looked at the whole life cycle of packaging including the waste management activities. Clift (1993) points out that 'If you look at the whole Life Cycle, you may reach conclusions which are quite different from those you reach if you only look at the waste'. This is the aim of the present study as explained in chapter one. Sometimes, looking at just the waste management or the product life cycle, does not allow the part of the life cycle of a product that is actually responsible for most of the environmental impacts to be readily identified.

2.2.10 Limitations of LCA

LCA is still evolving as a methodology. It is a very resourceful approach to evaluate environmental impact related to a specific system, however as with any other methodology, limitations exist. Following is an overview of some of LCA's limitations.

One of the main drawbacks of LCA is the transferability of the results. Indeed any LCA study is underpinned by its own assumptions, as discussed for example by McDougall et all (2001) (Hanssen, 1998) and Porteous, (1997). This restricts the comparison between similar LCA studies, as the assumptions to develop the model might have been different. For example the assumption used to set the boundary system of an LCA influences the outcome of a study. Thus depending what is included or not within the system being studied different conclusion can be reached. An LCA on any electrical appliance such as a television set or a washing machine, the inclusion of the 'use phase' or not will dramatically change the outcome (Ecobalance and DMG, 1999). When comparing products or services these have to be the same types otherwise the results are biased (Otto, 2002)

Norris (2003) discussed the issues of the data requirements within LCA. The data quality is perhaps the largest limitation and uncertainty of LCA. Thus if a database is used to model the energy input in a model, how relevant are these data to the specific model under study? Databases use averages figure based on a nation or Europe (for example), how different are the emissions of the specific power plant included in the study to that of a database.

LCA only predicts potential not actual impacts. It employs an overall system balance and functional unit to aggregate resource use, solid waste and emissions overtime and space. It is not however able to assess the actual environmental effects of a studied system, nor does it assess safety, risk or whether thresholds are exceeded (Guinee, 2001).

The LCIA is based on set of general conversion factors, developed from best scientific knowledge in chemistry, toxicology and physics. Our knowledge of the impacts of human activity with the environment is improving daily. The current knowledge in equivalency factor is still developing (Udo de Haes, 1993). Some impact can not be converted at present. This is true for some aspect of biodiversity. Thus the current weighting methodology can be seen as incomplete due to a lack of conversion factors (Hanssen, 1998).

The last stage of LCIA, valuation, is still very controversial. Before the valuation stage results are presented in non-comparable units, which make it hard for decision-makers to reach a conclusion. There is a need to convert and aggregate the results. However, as of today there is still no agreed methodology on how to do this but a range of tools are available (Bengtsson and Steen, 2000). The use of a weighting methodology can obscure the transparency of the results and increase the uncertainty of the results as weighting approaches rely on different values, judgements and methodologies of aggregation.

Huijbregts et al (2003) discuss the limitation of LCA in the uncertainty of the model. "*An important source of uncertainty is the complete lack of data for several emissions*". They go on to discuss that this type of uncertainty is most severe for the toxic impact categories and radiation. The addition of one single substance emission may cause a substantial increase in the total score for these impacts.

2.2.11 Alternatives to LCA

The sponsor for this project specified the use of LCA as the methodology for the research. However, several other methodologies exist that can also assess environmental impacts few of which include other parameters than just the environment such as hazard in 'Risk Assessment Analysis'. Some of these methodology are being widely used for policy development and product understanding, each with their own their advantages and drawbacks. Here is a short overview of some of these techniques.

- **Environmental impact assessment** (EIA) is defined as an assessment of the impact of a planned activity on the environment. It is a systematic process whereby information about the environmental effects of an action is collected and evaluated, with the conclusion being used as a tool in decision-making (Pearce et al., 1999). EIA gives a systematic and objective account of the significant environmental effects to which an action is likely to give rise.

- **Risk assessment** (RA) estimates the probability and severity of hazards to human health, safety and ecosystem functioning. Any hazardous substances have the potential to cause harm. Yet, the chances that they will, depend on the circumstances of their introduction to the environment, i.e. on the exposure of humans and the natural environment to the hazard. This risk assessment needs to assess the potential hazard, the likelihood of that hazard being realised and the severity of the impacts for any given level of exposure (Pearce and Hett, 1999).

- **Ecological footprint** (EF) is a term developed by the Canadian William Rees. Rees who defined (EF) as "*the maximum rate of resource consumption and waste discharge that can be sustained indefinitely without progressively impairing the functional integrity and productivity of relevant ecosystems wherever they may be*". (Robins, 1995). Ecological footprint calculates an index of sustainability and resource use by referencing the consumption of resources and the associated environmental impact to a common unit of geographical area. The land areas associated with the use of each resource or impact are then aggregated to give an 'Ecological Footprint' (EF), which represent the area of land required to provide the resources and mitigate the effects of the impacts on the environment (Chambers et al., 2000). A common technique for comparing the relative sustainability of nations is calculating the per capita EF, which gives rise to the idea of an individual's fair share of the earth's resources (Wackernagel, 2001). The aim of EF is the overall sustainability of a nation, town or person, not that of a product, activity or policy

- **Mass balance** (also known as materials flow analysis) studies follow and quantify the flow of a material or materials in a defined situation and over a period of time. This allows the identification of the points in the life cycle where resource use is most inefficient and tracks the types and quantities of wastes produced. Mass balance data, therefore can be used to maximise resource efficiency. The underlying principle is the fundamental physical law that within a closed system the total mass is constant. There may be movement of mass and transformation of mass but it is not created or destroyed (Linstead et al., 2000).

- **Multi-criteria Analysis** (MCA) identifies a set of goals or objectives and then seeks to identify the trade-offs between those objectives for different policies or ways of achieving a given policy. It then seeks to identify the 'best' policy option by attaching weights (scores) to the various objectives. MCA has attracted attention in the waste management policy field Powell (1996); Chung et al. (1996). MCA is very useful where there is a need to combine quantitative and qualitative data, which is not feasible with LCA. The main drawbacks of MCA is the arbitrariness that might happen in ordinal scoring of qualitative impacts, and in weighting overall impacts for relative importance as no clear methodologies exists. Ordinal scoring outcomes are generally number (index) which can be but in order of magnitude, i.e. option A score a 3 and option B score a 5. However the difference in terms of impacts between 3 and 5 is not known.

2.3 Environmental economics

An LCA identifies potential environmental impacts; these can be combined with an element of economic cost based evaluation, especially when looking at policy and strategy development. One main criticism of this combination is that the results are often presented in terms of economics and the environmental dimension comes secondary. Some of the economic assessment techniques that can be used with LCA are Cost Benefit Analysis (CBA) and Cost Effectiveness Analysis (CEA).

Methodologies such as LCA are informative tools to be used by decision-makers to better understand the potential environmental impacts of policies. Yet not only environmental impacts need to be considered, there is also a need to ensure a balance between what is environmentally reachable and what the economic and social acceptability of achieving the goals would be. Article 174(3) of the Amsterdam Treaty foresees that 'in preparing its policy on the environment, the Community shall take account of [...] the potential benefits and costs of action or lack of action' (European Commission, 2000). Cost-Benefit Analysis (CBA) methodology revolves around how economic benefits and economic costs are defined, and how they are measured. The valuation of environmental impacts can be achieved by the assignment of monetary values to quantified emissions identified through environmental impact assessment, based on estimates of the likely environmental and human health impacts associated with those emissions (Craighill, 1996). However, monetary valuation of environmental impacts is the focus of considerable debate, particularly with regard to ethics and methodology as discussed for example by Wightman et all (1999), Dobson (1998) and European Commission (200).

2.3.1 Cost Effectiveness Analysis (CEA)

Cost Effectiveness Analysis seeks to identify the least-cost (economic) option to achieve a specific reduction in environmental impact, without any explicit attempt to specify what the benefit of that level of impact reduction might be. Tietenberg (1992) defined CEA as "identifying the cheapest manner of achieving an objective". Pearce and Hett (1999) concludes that it is an aid to choosing between options, but, that it does not say whether a given option is intrinsically worthwhile, merely whether the option is better than some other option.

CEA starts from a pre-determined goal and identifies those technical and policy options which achieve these goals at the lowest monetary cost. For each alternative option, the economic costs as well as the level of environmental impact reduction that can be achieved in using this option can be identified and reviewed, allowing the option that will enable to reach the goal at the lowest economic cost to be identified (European Commission, 2000).

CEA applications are mainly encountered in environmental regulation, where various options to achieve specified air or water quality standards will be compared to find those that impose the least economic cost in progressing toward a fixed goal (Zerbe and Dively, 1994).

CEA combines non-monetary indicators of achievement with monetary estimates of costs. Thus a policy that secures a reduction of X tonnes of carbon dioxide emissions can be compared directly with one securing 0.8X tonnes of emissions reduction. If the cost of the former is £100 and the latter is £50, then simple quotients (rates per tonne) reveal the second policy to be more cost effective: £50/0.8 Xt = £62.5 per Xt compared with £100/1Xt = £100 per Xt.

2.3.2 Cost Benefit Analysis (CBA)

Cost-Benefit Analysis is a method used in the development of projects and policies to evaluate their social impacts. CBA establishes whether or not a project or policy is worthwhile from a social welfare perspective. This is achieved by comparing the monetary value of benefits with the monetary value of costs. An action is considered worthwhile or justified if the social benefits outweigh the costs. CBA tries to determine the best level of an environmental objective and/or whether a pre-defined policy mix is justified by its environmental benefits. CBA can be used to examine the justification of a single action in terms of the relative costs and benefits, or can be used to compare the relative advantages and disadvantages of a series of options (European Commission, 1999).

The use of CBA in environmental policies poses the question of which levels of targets should be envisaged? This question is much more difficult to answer since, additionally to the costs, another parameter needs to be valued in monetary terms, i.e. the environmental impacts of a measure. This is measured by using an economic valuation of the environmental impacts identified in an LCA, for example. The following questions characterise the discussion around benefit valuation: can we know how much a cubic metre of clean air is worth? Does a life saved have a price tag? Is it ethical to price the environment? (European Commission, 2000)

Figure 2.7 represents the results of a CBA for hypothetical targets. It takes into account environmental impacts and economic benefit/cost to reach each of the targets. The environmental impacts would have been identified through, for example, an LCA and monetary values have been established using economic valuation, thus enabling them to be included in the economic cost and the benefits of reaching the targets. In this case it can be seen that economic benefits outweigh economic costs at target levels up to and including 65%, at 70% costs exceed benefits. Decision-makers can therefore concentrate on maximising the targets knowing that it will be in the region between 55% and 65% (for this particular example).

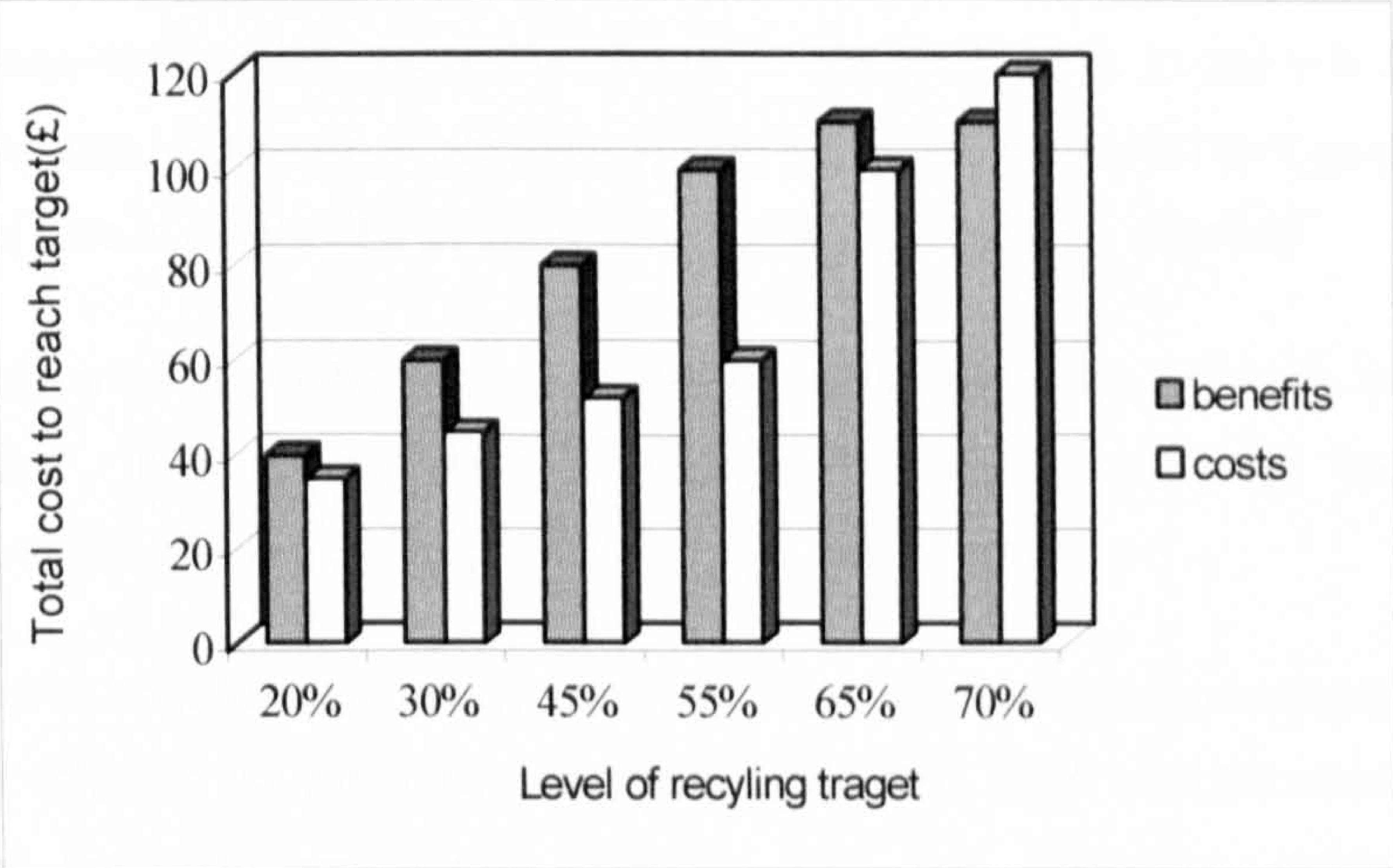

Figure 2.7 Cost benefit of reaching targets adapted from (Linher, 2000)

2.3.3 Discounting

The practice of discounting arises because individuals attach less weight to a benefit or cost received in the future to one received now. Discounting reflects this preference by giving costs and benefits in the future a lower 'weight' (Pearce and Warford, 1993). The two main justifications for discounting are:

- Capital productivity: money today can be used for productive purposes to generate additional income. Therefore, £1 taken today used to buy capital may be worth e.g. £1.20 in a year's time, and
- Impatience: individuals may not want to delay the chance of having money in the current period, with underlying reasons being the risk of death, uncertainty over the future, and diminishing marginal utility[5] of money (i.e. the cost of life could be much higher) in the future due to individual's expectations of being better off in the future.

(European Commission, 1999).

[5] Marginal utility: The marginal utility of any item (or money) depends on how much of it you already have, therefore the greater the supply of a product (money) the smaller its marginal utility (Easterlin, 2005).

Discounting is a method that enables the element of time to be incorporated into CBA and CEA. Throughout the life of a policy, costs and benefits will occur at different times and the use of discount rate provides a method for dealing with such differences and attempts to reflect individual's time preferences. In the context of recycling processes the future may be relevant since many external effects will only be felt in the future (e.g. soil contamination from landfill sites, depletion of natural resources, and global warming from carbon dioxide and methane emissions from landfill sites). The usual way to deal with temporal effects in the analysis is to apply a discount rate to future impacts. Suppose a damage of the value €X will occur in t years time, and the discount rate is r, then the present value of the damage is: $\mathbf{X/(1+r)^t}$

This present value is smaller than X. From the equation it can be seen the higher the discount rate r and the higher t, the lower the relative value of future damage (Beukering et al., 1998).

The costs and benefits of a policy must be considered across the whole life cycle of the system affected by the policy decision. Therefore, CBA can be used to combine aspects of financial costs and benefits with the economic valuation of the environmental impacts identified by LCA.

2.3.4 Economic valuation techniques

Economic valuation techniques are based on how much (in monetary term) people are willing to pay or give up or for good/services/taxes or to protect the environment, in that sense it characterise people preference and values. From that point of view economic valuation can be used as weighting methodology in LCA. Indeed one recurring problem in LCA is the aggregation of resulting environmental impacts, which usually are in non-comparable units. The valuation part of the impact assessment requires the assignment of relative values or weights to the various impacts. Research groups across Europe and the United States have developed several methodologies for the valuation stage in LCA, based on either: social opinion, political decisions, expert ranking, sustainability indicators or economic valuation, all of which are developed through the use of panels. A set of valuation factors that is widely acceptable has not been established, as there is a great deal of controversy about the various weighting techniques and valuation methodologies. Any consensus that might emerge will be artificial: there is no 'true' set of weights because there is no single objective function to which all will agree (Powell et al., 1998).

However, economic valuation can be one answer to the problem. Indeed, whilst some subjectivity cannot be avoided, economic valuation at least allows the comparison of impacts on a common scale, and the determinants of values can be made clear (Powell et al., 1996). Economic valuation is considered to be less arbitrary weighting procedure for environmental impacts (Beukering et al., 1998).

Economic valuation methodologies are concerned with estimating the value that individuals place on non-market goods and services as well as individuals' preferences for environmental improvement or conservation, or individuals' loss of well being because of environmental degradation or from losing an environmental asset (Pearce, 1994). A 'value' can be revealed by a consumer's behaviour derived from their stated 'willingness to pay' (WTP) or 'willingness to accept compensation' (WTA) (DoE, 1991).

The basic justifications for the use of economic values are:

- Economic values are based on individuals' preferences. As such they claim to measure what society, as the aggregate of individuals, prefers. By using such measures, WTP and WTA, is therefore responsive to social concerns rather than to the often self-selecting expert groups.
- LCA needs to be integrated with other equally valid social concerns, the main one being cost. Indeed money spent on one policy or action will not be spent

on something else: waste policy will take money that could have been used in schools. Hence, cost is a surrogate for the sacrifice that society makes when making policy choices.

- The economic approach forces a focus on damage, rather than for example, emissions, since it is damage that relates to the loss of human well-being, not emissions.
- Finally, the economic approach permits the integration of impacts that can not be measured in term of environmental impact such as visual impact on residents of waste treatment plant into LCA (Powell et al., 1998). Thus, an economic value can be applied such as hedonic pricing (see 2.3.3.3) to evaluate that specific impact.

However, limits exist to the application of economic valuation. Weighting and valuation methodologies often lack transparency in that the determinants of weights remain hidden. Furthermore Powell et al. (1996) argue that it is difficult to tell what is based on empirical knowledge of the environment and what is based on arbitrary political targets, which are often rooted in what is politically achievable rather than in what is environmentally sustainable. The important point is that LCA without the link to costs could easily lead to major distortions. It amounts to saying, for example, that a policy (product) A has less environmental impacts than a policy (product) B regardless of the cost of making the switch.

Conflicts between the conservation of environmental assets and traditional patterns of economic development exist, as demonstrated by the case of tropical forest clearance for agriculture, versus forest conservation. If environmental assets have inviolable 'intrinsic rights', then much economic development is morally unsound. If the rights of the people whose livelihood is subsequently put at stake are allowed, then there is a conflict of rights and no clear decision rules that enables us to choose the 'right' course of action. The pervasive problem is that as so many environmental assets are not marketed, there are no values to compare with those from economic development, and hence there is an asymmetry of valuation (Pearce, 1994).

A range of economic valuation techniques has been developed to assist in assigning the monetary value attached to environmental goods and services.. These techniques attempt to derive an individual's willingness to pay for an improvement [or willingness to be compensated for (willingness to accept) an environmental loss], as revealed in the marketplace, through an individual's actions or as directly expressed through surveys. The general aim of these is to determine the direct and indirect trade-offs that individuals might make. Thus, direct trade-offs might take place in supermarkets by choosing product A over product B based on Product A claims to be better for the environment, whereas indirect trade-offs take place in labour, housing and other markets.

The techniques that are most commonly used are briefly discussed below.

2.3.4.1 Conventional market price

Conventional market price or effect on production approaches, relies on the use of market prices to value the cost/benefits associated with a change in environmental quality. One approach in this category is the **dose response** technique, which determines the economic value of changes in environmental quality by estimating the market value of the corresponding changes in output of an associated good. As an example, changes in crop yield are linked to changes in atmospheric pollutant concentrations and deposition. The value of the lost yields is calculated by multiplying the yield loss quantity by the market price of crop (Beukering et al., 1998). An alternative technique would be the **replacement costs** approach, which calculates the cost of replacing/restoring an environmental asset after it has been impacted. Replacement cost will only give a minimum figure indicating only the engineering and other cost of re-creation, such as that of a deteriorating building due to acid rain or atmospheric pollution (Pearce, 1993).

2.3.4.2 Household production function

Household production function approaches use expenditure on activities or goods which are substitutes for, or complements to, an environmental good to value changes in the level of the environmental related good. The **avertive expenditure** approach relies on estimation of expenditure on substitute goods. It is based on determining the amount people are willing to spend on measures which mitigate impacts: the installation of double glazing windows is a substitute for reduced noise impacts, and expenditure on this provides an indication of willingness to pay for policies aimed at reducing noise levels (Powell et al., 1998). However it may be questioned whether people at large understand the level of environmental protection they are paying for, and the degree to which purchase of the item is considered a 'second' best option. The **travel cost method** is mainly used for estimating the recreational value of a specific area. It is based on the concept that people spend time and money travelling to a recreational site and that these expenditures, or costs, can be treated as revealing the demand for the site.

2.3.4.3 Hedonic pricing methods (HPM)

This approach is mainly used to analyse house prices. House prices are seen as a function of characteristics of the house itself (number of bedroom, heating system, etc), neighbourhood characteristics and also environmental variables such as air quality, nearness to a forest, etc. (Beukering et al., 1998). The hedonic price method is based on the concept that the price paid for a complementary good implicitly reflects the buyer's willingness to pay for a particular environmental attribute, or his willingness to accept an increased risk. These methods determine an implicit price for a good by examining the 'real' markets in which the asset is effectively traded. **Hedonic property (land) prices** have been used in the valuation of characteristics such as air quality, noise, fishery quality and amenity characteristics associated with residential and other properties. It is still commonly used to assess amenity effects, although many analysts have argued that the technique is not reliable in the valuation of environmental effects which are not readily perceptible in physical terms (European Commission, 2000)

2.3.4.4 Experimental (Hypothetical) Markets

The two key techniques that involve the use of experimental or hypothetical markets are the contingent valuation method and the contingent ranking method. Under the **contingent valuation method (CVM),** individuals are surveyed to determine their willingness to pay for a specified change in the quality or quantity of environmental goods, which often are not valued in conventional markets, (or how much compensation they would expect for an increase in risk or in environmental damages) (Powell et al., 1998). The mean willingness to pay value across all valid values is then used to provide an indication of the economic value of the specified changes. Difficulties with this approach include: problems in understanding the concept of risk and in particular marginal changes in risk, and individuals acting strategically when responding to questions (or indeed respondents giving random answers in that numbers are pulled out of the air). One potential advantage of CVM is that it is a way of estimating non-use values, by surveying the general population and asking what the respondent would be willing to pay to avoid some hypothetical incident or damage to a resource (Montgomery, 1998). For example a Contingent Valuation might take place on how much people would be willing to pay to protect the Grand Canyon (USA) from environmental impact of higher transport due to tourism, or how much they would be willing to pay for a sewage system to be implemented to avoid the pollution of a nearby lake. **The contingent ranking (or stated preferences) method** involves the elicitation of individual's ranking of preferences amongst a bundle or 'basket' of different environmental outcomes. Values for changes in environmental goods are derived by 'anchoring' preferences to either a money sum or the real market price of one of the goods included in the bundle/basket of outcomes. An example would be to ask residents of a Local Authority their willingness to pay extra council tax to increase recycling rate by improving kerbside waste collection. Thus giving them a choice such as:

- 20% recycling rate for X council tax a year
- 30% recycling rate for Y council tax a year
- 40% recycling rate for Z council tax a year

(European Commission, 1999).

Willingness To Pay (WTP) is familiar in the conventional market place: each decision to purchase a commodity records a vote for that commodity. The economist's construction of a demand curve[6] is a graphic illustration depicting the relationship between quantity demanded and price when all other economic variables are held constant. In general the lower the price, the higher is the demand. Thus more people will be willing to pay a given price for something the lower that price is. But, willingness to pay is obviously influenced by ability to pay: income and wealth (Powell et al., 1998). WTP measures will depend upon income, and care must be taken to consider how these values should be adjusted in moving between countries or regions with different income levels. Several different approaches have been suggested, such as adjustment according to relative income, according to purchasing power and/or environmental awareness. However, using such approaches assumes that WTP for environmental quality varies proportionately with income; but damage costs are not necessarily constant across countries in terms of income. Damage costs, in the context of economic valuation, are a (monetary) measure of society's loss of well-being. However the monetary value place on that well-being will be affected by the level of income available.

Monetary valuation techniques are controversial. They represent a convenient way of putting a value to the environment, but the results may be very misleading. The assumption that the environment only has a value in respect to human preferences is clearly open to debatable.

[6] Demand curve is a graphic illustration depicting the relationship between quantity demanded and price when all other economic variables are held constant, (King and Mazzota, 2005).

2.3.5 Economic Value

In environmental economics one usually distinguishes between two main sources of value: use-value and non-use value[7]. Together, they are called the total economic value (Tietenberg, 1992).

Use-values arise form individual's use of environmental goods or services. One can distinguish between direct and indirect use values:

- Direct use-values refer to an individual's direct use or interaction with environmental goods or services such as fruit and vegetable
- Indirect use-values refer to indirect support and protection provided to economic activities by the environment: e.g. water is of value indirectly because it is essential to agriculture. This is indirect use-value as the individual is not the farmer but benefits from what is cultivated..

Non-use values, on the other hand, are derived neither from current direct nor from indirect use of the environment. If, for example paper recycling can preserve old growth forests from being cut down, this may generate benefits for some individuals simply because they appreciate the existence of a pristine environment, even if they do not intend to use it for recreation or other purposes. Non-use values have, amongst others, been termed 'existence values', intrinsic values' or 'bequest values', all having a slightly different interpretation. These terms should be best seen as giving possible motivations underlying the existence of non-use value. Non-use value is used to distinguish a value that is placed on environmental damage by parties who do not themselves make use of the resource or property affected from the damages or loss suffered from actual users (Montgomery, 1998).

[7] Use Value: This is the value based by individual's on the resource they use from the 'environment'
Non-use value: This is the value individual puts on resource they do not use but believe should be protected as they might use them in the future (Pearce, 1993).

2.3.6 Economic assessment and LCA

In order to be able to compare environmental impact with the economic costs/benefits for example of a policy, all parameters considered need to be in comparable units. As reviewed in the previous sections, several methodologies can be used so as to translate environmental impact value into a monetary value. Once this is achieved then by using a CBA approach all parameters can be model. A CBA will identify all economic costs and benefits for all options. A project is desirable when the economic benefits outweigh the costs.

Very little has been published on the combined use of LCA and CBA and, where it has been reported, it has not been associated with waste, recycling or environmental policy. However the existing studies give some understanding of the potential advantages and limitations of using this combined approach.

Salter and Ford (2001) developed a methodology for a holistic environmental assessment in the offshore oil field business. A holistic approach includes environmental, economic and social aspects. Their starting point was that LCA may be used to identify and quantify environmental burdens, and it is often seen as a holistic approach in itself. However, LCA does not include accidental emissions, indirect, secondary or cumulative impacts, and is limited to environmental impacts. CBA informs us whether a project is financially desirable or not. However, it has its weaknesses: some environmental effects are not scientifically fully understood and so cannot be satisfactorily valued in economic terms. The uncertainty in the occurrence of environmental effects, their extent and the estimates of the values of these effects, are not fully addressed by CBA. The risk is that by putting monetary values the results are seen as a final value, and applied by decision makers, whereas it should be seen as an aid to decision making. Their conclusions were that one method can not deal with all aspect of environmental evaluation nor economic costs and benefits. Instead several methods needed to be combined to try to achieve a holistic approach.

Wightman et al. (1999) combined LCA and CBA to evaluate and compare the relative environmental impact and socio-economic costs of several products, made from either rapeseed oil or mineral oil. They used as a case study the chainsaw bar oil, which is a total loss oil and has substantial potential for substitution of the conventional oil (mineral oil) with rapeseed oil. The valuation of environmental impacts was achieved by the application of monetary values to emissions quantified in the LCA, based on environmental and human health impact associated with those emissions. Their conclusion was that this is a promising combination, even thought they identified large data gaps in the input, and a lack of explanation for the assumptions made in the studies they referred to.

2.4 Conclusion

The Packaging and Packaging Waste Directive is the outcome of a lengthy consultation process and negotiations; setting targets even though these have not been justified scientifically as representing either environmental or economic optimum levels.

LCA was chosen by the project's sponsor as the methodology to be used. An extensive literature review concentrated on the methodology and its application within waste management and policy development. Methodological issues were identified from the literature survey such as boundary setting, allocation and weighting. Economic valuation is a concept that can be used as a weighting indicator as all impacts can eventually be converted into money. LCA is still being developed as a methodology to adapt to new applications but also improve transparency, reliability and understanding of results for stakeholders. Several other methodologies are available to researchers, each with their benefits and drawbacks.

In the new millennium, more and more policies are being implemented that deal with specific waste streams in the context of Producer Responsibilities. Currently, there is an understanding that policy making should be developed on an informed basis, and thus a growing number of studies are being commissioned to help the process.

As discussed in section 2.2.4, and 2.2.5 studies previously conducted concentrate on: landfill versus incineration for paper waste or newsprint, comparing paper and plastics waste, plastic packaging. Yet the study the closest to this research is the RDC and Pira (2003) one commissioned by the European Commission. It concentrated on establishing the technical feasibility, environmental impact and economic costs associated by the Packaging and Packaging Waste Directive. The major differences is that the RDC – Pira study looked at packaging once it has become waste, the current project is looking at the whole life cycle of cardboard packaging, including waste.

The present research will develop a methodology that specifically relate to packaging wastes, using the case of cardboard packaging to illustrate its application. The present research looked at the whole life cycle of cardboard packaging waste rather than just when it becomes waste. This approach used can be transferred to other producer responsibility obligations regulations such as for waste electrical and electronic equipment (WEEE), batteries or end of life vehicle. So far projects that have studied the environmental impact of, for example packaging waste, or WEEE, have only looked at the products once they become waste rather than the whole life cycle.

3 Paper and board

This section first gives an overview of the paper and board industry in the UK. It goes on to present the manufacturing process of pulp and paper, and finally outline the main environmental impact linked to the industry.

3.1 Paper and board industry in the UK

Looking at all the packaging on the market, paper and board represent 40% of the overall. Of these 40% cardboard represents the majority around 60% (CEPI, 2005).

Table 3.1: Proportion of packaging by materials in the UK waste stream 2003

Paper and board	37%
Glass	23%
Metals	8%
Plastics	18%
Wood	14%

Source: (Department for Environment 2005)

In 2001 3,855,000 tonnes of paper packaging was put on the market. Businesses obligated under the Packaging Waste Regulations reported 3,134,034 tonnes of paper packaging. In that year, 2,030,944 tonnes of paper packaging were recycled, which represents 53% (ERM 2002).

The paper industry in numbers:

- 6.2 million tonnes of paper were produced in the UK in 2001,
- 1.2 million tonnes were exported and
- 7.6 million tonnes were imported.

The paper consumption in 2001 for the UK was 12.6 millions tonnes, an increase of 2% on the previous year. 84% of imports come from the European Union and Norway, and small quantities from North America.

The UK paper industry uses high levels of waste paper in the paper production process. In 2001: UK mills consumed 4.6 million tonnes of waste paper. In 2001 waste paper represented 67% of all fibre used in the production process, one of the highest in the world. This is also far greater than other industries, e.g., ferrous metals (45%), glass (20%), plastics (9%).

Although recycling is both economically and ecologically sound, recovered paper cannot be used in all paper grades nor can it be used indefinitely. There are technical constraints to the unlimited use of recycled stock. Re-pulping and processing decreases fibre length and, therefore, some virgin pulp is always required to maintain the strength and quality of the finished product. A recovered fibre can be recycled only about five to six times due to the progressive deterioration of its length. As a result, around 85% of waste paper comprises reusable fibre, 15% is waste. This means that 0.65 million tonnes of unusable material from waste paper arises in the UK paper mills annually.

The more times fibres are recycled; the more cleaning and fibre separation is required, which increases the amount of water used. The water is contaminated more quickly and needs to be changed more frequently, which leads to an increased use of a diverse range of chemicals. (ERM 2002)

The paper industry for the UK in 2001 is made of 61 companies operating 87 paper mills. The amount of waste paper utilised by the mills varies accordingly to the product being manufactured. Newsprint and corrugated case production uses the highest percentage of waste paper. Restricted reprocessing capacity is a significant barrier to increasing paper recycling in the UK. Indeed, paper mills in the UK are running at full capacity, yet high levels of paper imports are still required to satisfy paper consumption requirements. The Paper Federation believe that an increase in paper recycling in the UK can only occur with investment in new mills. This has to be considered when setting recycling targets, indeed how much new capacity is required and how much are new mills likely to increase environmental burdens, if they were only running at reduced capacity (ERM 2002).

3.2 Cardboard manufacturing process

This chapter presents the paper/board manufacturing process using virgin pulp, and the process involved when using recovered fibres. It also presents an overview of the environmental impacts associated with specific stages of the process. Figure 3.1 is a simplified diagram of the stages within pulp and paper production.

3.2.1 Virgin Pulp production

This section focuses on the production of pulp using virgin materials. The primary raw material used in paper manufacturing is cellulose fibre, which comes from wood. The wood used for papermaking comes from the parts of the tree that other commercial industries don't want - such as saw mill waste, forest thinning and furnishing industry. This was discussed in the literature by McDougall et al (2001), Ekvall (1999), Ekvall and Finnveden (1998) and Stromberg et al, (1997) (Stromberg, Haglind et al. 1997).

3.2.1.1 Wood

Wood consists of approximately 50 per cent cellulose (fibres), 30 per cent lignin (a resinous adhesive which holds the fibres together), and 20 per cent aromatic hydrocarbons and hemicellulose carbohydrates (Biermann 1996). In order to obtain cellulose in usable form for paper manufacture the fibres and lignin must be separated and impurities removed.

The higher the cellulose content of the pulp, and the longer the fibres, the better quality the paper. Hardwoods generally contain a higher proportion of cellulose but of shorter fibre length than softwoods, which are more resinous (Confederation of paper industries 2003).

When the wood is delivered to a mill it goes first into a de-barker and is then chipped.

3.2.1.2 De-Barking

First the wood needs to be refined before it can be transformed into pulp. First, bark is stripped from the logs by knife, drum, abrasion, or hydraulic barker. The stripped bark can then be used for fuel or as soil enrichment (Stanley 2001).

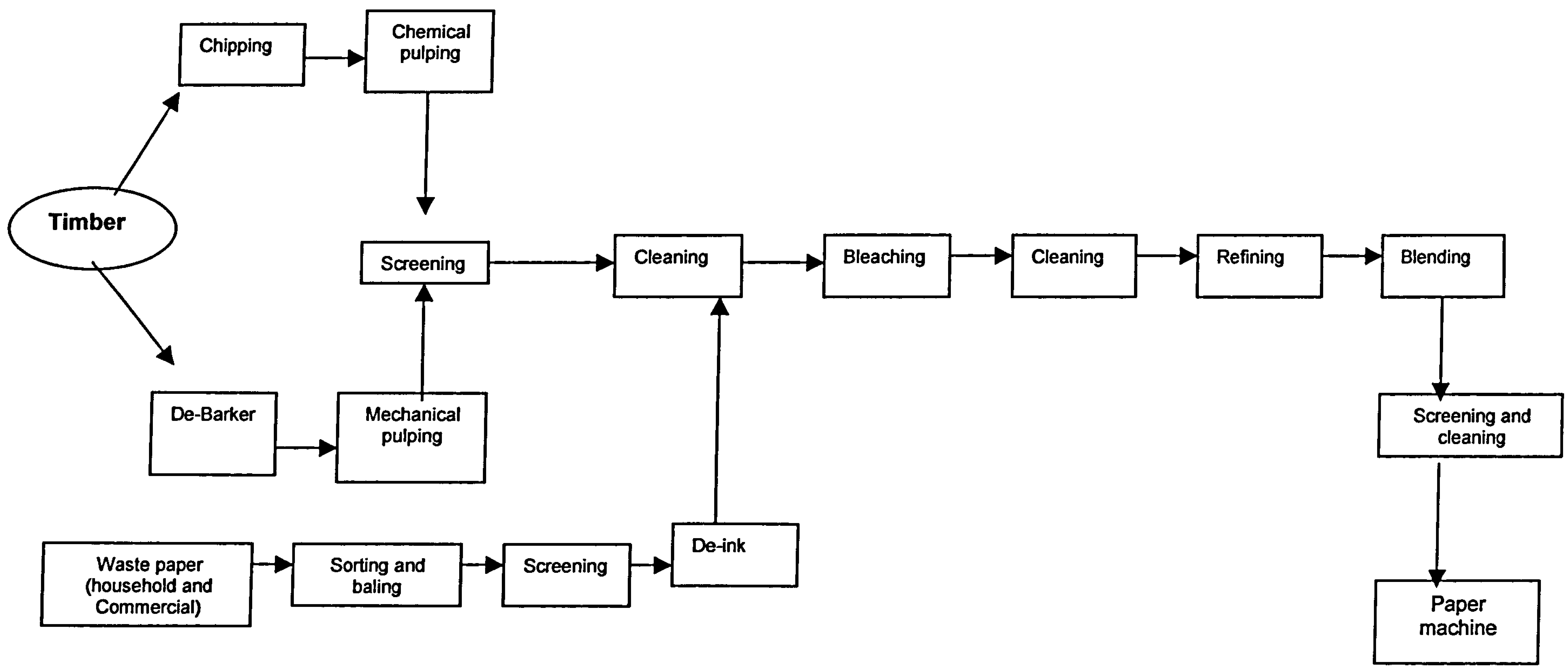

Figure 3.1: Pulp and paper making production, source adapted from CEPI (2005)

3.2.1.3 Chipping Machine

Stripped logs are chipped into small pieces by knives mounted in massive steel wheels. The chips pass through vibrating screens, whereby undersized chips, dust etc and oversized chips are rejected. Accepted chips are then stored ready for the pulping.

Three different pulping processes exist: Chemical pulping, mechanical pulping and thermo-mechanical pulping (TMP).

3.2.1.4 Chemical Pulping Process

Chemical pulping uses a combination of chemicals, heat and pressure. The wood chips are cooked with chemicals to dissolve the lignin that bonds the cellulose fibres together. There are several chemicals combinations that can be used to cooked the wood chips: caustic soda and sulphur, sulphuric acid or hydrogen sulphite with either calcium, magnesium or sodium (Biermann 1996). The choice of chemical will depend on whether the wood is soft or hard and the grade of paper to be manufactured (Smook 1996).

The chemical pulping process is energy self-sufficient as nearly all by-products can be used to fire the pulp mill power plant. Mills operate a closed loop system whereby 98% of the chemicals used in the process are recovered and re-used (Stanley 2001). In modern mills, recovery boiler operations and the controlled burning of bark and other residues make the chemical pulp mill a net energy producer which can often supply power to the grid, or steam to local domestic heating plants (Stanley 2001) . The chemical pulping process produces lower fibre yield than mechanical pulping, typically 50-60%. Chemical pulp, however, produces a strong liquid effluent that needs to be treated (Das and Houtman 2004).

3.2.1.5 Mechanical pulping

Mechanical pulping is a very cost effective process as the entire log apart from the bark is used, but it is an energy intensive process (Confederation of paper industries 2003). It simply uses mechanical energy to physically separate the fibres by forcing debarked logs, and hot water between enormous rotating steel discs with teeth that literally tear the wood apart. Alternatively, logs can be pressed against grindstones which is why this process is also known as groundwood pulp. The quality of the pulp is low as the fibres produced are broken by the grinding and is still surrounded by lignin (Das and Houtman 2004). This means the paper products produced are weak and turn yellow quicker.

Whilst the pulp yield per ton of wood is much higher than chemical pulping (around 0.95 tons) the amount of energy required is approximately double that required by the chemical process. Water consumption, however, is roughly one third of the amount required by chemical pulp mills (Confederation of paper industries 2003).

3.2.1.6 Thermo-mechanical pulping (TMP)/Chemo- Thermo- mechanical Pulping (CTMP)

Thermo-Mechanical-Pulp (TMP) and Chemi-Thermo-Mechanical-Pulp (CTMP) are a combination of the mechanical and chemical processes. De-barked logs are chipped in both, and then heated to extreme temperatures to soften them before passing through refiners for mechanical reduction to fibres. The difference is that chemicals are sprayed onto the chips in the CTMP process to assist with the softening of the chips during refining (Biermann 1996).

The main disadvantage of this form of pulping is the high energy demand. Both processes result in a stronger pulp, which can be applied to higher quality end uses (Das and Houtman 2004).

3.2.2 Recycled paper

Before waste paper/board can be recycled, it needs to be collected, sorted, graded and baled prior to its transportation to the mill. Recovered paper is generally sorted by recovered paper merchants or waste management companies (Confederation of paper industries 2003).

Waste paper is collected from the household waste stream as well as the commercial waste stream. Waste paper merchants collect used paper which is then sorted by hand into different grades. Paper not suitable for recycling is removed. The waste paper merchant will then bale the waste paper ready to be taken to the paper mill.

3.2.2.1 Screening

When the bales of waste paper arrive at the paper mill they are loaded onto a conveyor and passed into a circular tank containing water. This has a very powerful agitator at the bottom, which breaks up the bales into small pieces. The pulp mass created begins to look like thick porridge. This machine is known as a Hydrapulper and is fitted with special devices for removing unwanted contraries such as wire, plastic, paper clips and staples (McKinney, Howard et al. 1995)

3.2.2.2 De-inking

If white paper is being produced then the waste paper has to go through a de-inking stage. However for most cardboard and other packaging paper grade it is not required (Confederation of paper industries 2004). There are two main processes for de-inking waste paper - these are known as washing and flotation.

3.2.2.3 Washing

The waste paper is placed into a pulper with large quantities of water and broken down into a slurry. Most of the water containing the dispersed ink is drained through slots or screens that allow the dispersed ink particles through, without taking the pulp. Adhesive particles, known as 'stickies' are removed by fine screening (McKinney, Howard et al. 1995).

3.2.2.4 Flotation and cleaning

Again the waste is made into a slurry, the slurry is diluted, treated with chemicals (calcium soap) and cleaned by bubbling air through the liquor. This concentrates the ink particles in the froth, which can be skimmed off(McKinney, Howard et al. 1995). Because the ink is removed from the flotation machine in a concentrated form, the flotation system does not require a large water treatment plant.

Finally the pulp is centrifuged and washed, and the remaining particles of ink are dispersed.

3.2.2.5 Bleaching

For higher grade of writing and graph paper bleaching will be required. The type of bleaching agent used will depend on the pulp (mechanical or chemical) and the end use of the paper. Thus mechanical pulp which has a higher content of lignin (thus tend to become yellow quicker) will be bleached using hydrogen peroxide (mainly), hypochlorites or sodium bisulphite (Smook 1996). It should be noted that more than half of mechanical pulp produced is used for end applications that do not require bleaching (Stanley 2001). Chemical pulp is bleached using either - chlorine gas combined with chlorine dioxide or hyperchlorite, - oxygen delignification with peroxide brightening or oxygen delignification with chlorine dioxides (Virtanen and Nilson 1993).

The main problem of bleaching is the treatment of effluent which are heavily contaminated with ink, glues, coating and so on (Virtanen and Nilson 1993).

The presence of kaolin and chalk filler, ink and contaminants such as strings, staples, glue, lacquer, grit and the loss of broken fibres result in over a tonne of collected paper being required to produce a tonne of recycled fibre. Fifteen to twenty percent will be lost depending on the origin and quality of the input fibres and the pulping process (McKinney, Howard et al. 1995).

Recycling shortens fibre length, making paper weaker. As a results, fibres can only be recycled four or five times at the most. To strengthen recycled paper, virgin pulp is added in different amounts depending on the end paper required.

3.2.3 Papermaking Process

Once the pulp has been produced either from virgin or recycled paper by either a mechanical or chemical process it needs to be made into paper. At this stage the pulp is diluted with water and the output is referred to as 'stock'.

3.2.3.1 Refining

Before refining, the fibres are stiff, inflexible and form few bonds. The stock is pumped through a conical machine, which consists of a series of revolving discs. The violent abrasive and bruising action has the effect of cutting, opening up and de-clustering the fibres and making the ends divide. This is called fibrillation (Biermann 1996). In this state, the fibres are pliable and have greater surface area, which significantly improves the fibre bonding. The properties of the paper are directly related to the refining process. Refining used to be called beating (Confederation of paper industries 2004).

3.2.3.2 Blend Chest

The stock passes into a blending tank where chemicals and dyes can be added to obtain the required characteristics of the finished paper (Smook 1996). After passing through a second cleaning system the stock is now ready for the paper machine.

3.2.3.3 Screening and Cleaning

The stock contains undesirable fibrous and non-fibrous materials, which should be removed before the stock is made into paper or board. A second cleaning stage involves removing small particles of dirt and grit using rotating screens and centrifugal cleaners (Smook 1996), (Biermann 1996).

3.2.4 Papermaking Machine

The Paper Machine is a very large piece of machinery. The machine itself consists of 7 distinct sections. The flow box, wire, press section, drier section, size press, calendar and reeling up (Confederation of paper industries 2004).

3.2.4.1 Wet section

The first section of the machine is called the 'Wet End'. This is where the diluted pulp first comes into contact with the paper machine. The pulp is poured by the 'flow box', onto the 'wire' with the fibres distributed evenly over the whole width of the paper machine (Biermann 1996)..

As the paper stock flows from the flow box onto the wire, the water drains away through the mesh leaving tiny fibres as a mat on top of the mesh. As the stock travels half way down the wire, a high percentage of water has drained away. From this point the removal of water has to be assisted by suction from underneath the wire (Virtanen and Nilson 1993).

A dandy roll is situated near the end of the wire section. It is covered with a woven wire and is in contact with the upper surface of the forming web. The dandy roll usually carries a design to form a watermark. When the thin mat of fibres reaches the end of the wire, although it is still very moist and weak, it has become a sheet of paper (Confederation of paper industries 2004).

3.2.4.2 Drying Section

When the paper enters the dry end section of the paper machine it has a water content of approximately 50% (Smook 1996).

In this section of the papermaking machine the paper will be dried, pressed, sized, dried more and conditioned and carefully packed for despatch.

3.2.5 Conversion and Printing

Once the paper is made, a great deal of it is converted into a product. Converters specialise in transforming reels and sheets of paper and board into a vast array of finished products for distribution such as boxes, cartons and stationery. Converters sell their products to the public or to other manufacturers (Smook 1996).

Earlier on mechanical and chemical pulping has been discusses. However to manufacture cardboard, mechanical pulping is the process most widely used and thus included in the LCA conducted as part of this research.

3.3 Pollution related to paper manufacturing

Pulp and paper mills environmental impacts relate mainly to their effluent and air emissions, while paper mill effluents are relatively benign, the pulping stage is the main source of the industry's pollutants ((IIED) 1996).

3.3.1 Effluent

Production of both virgin and recycled paper gives rise to pollutants which are discharged to water, called effluents. When assessing these pollutants produced in paper making, three key parameters, among others, are monitored: total suspended solids (TSS); biological oxygen demand (BOD); chemical oxygen demand (COD) (Biermann 1996). This demand for oxygen depletes that available to fauna and flora, thus damaging wildlife near to, and downstream from, effluent discharges, resulting in eutrophication. These impacts happen at a local level ((IIED) 1996).

Much more toxic pollutants found in effluents are chlorinated organic compounds (AOX) resulting mainly from the bleaching processes, dioxins and furans. They are responsible for human toxicity and ecotoxicity (Stanley 2001).

There are hundreds of chlorinated organic compounds that exist and can be produce during the bleaching process. The nature and level of toxicity of these compounds depends on the type and amount of chemicals used during the process. These chlorinated organic compounds are persistent and bio-accumulative in the environment. They arise mainly from the mechanical pulping process and bleaching.

Dioxins are extremely toxic, persistent and carcinogenic. Furans are chemically similar but an order of magnitude less toxic and less persistent than dioxins. Dioxins and furans tend to accumulate more in the pulp itself than in the effluent. Dioxins and furans have several impacts on fish and mammals; the impact to humans is harder to establish. They are, however, suspected of causing miscarriages, birth defects, liver damage, skin complaints and behavioural and neurological problems.

Levels of heavy metals in effluent are of concern within the paper recycling process. Metals such as copper, chromium, lead, zinc, nickel and cadmium are commonly used in printing inks and are discharged not only to wastewater but also to waste sludges and some remain in the final paper product. Heavy metals contribute to human toxicity, and ecotoxicity.

However, the levels of these materials in recovered paper, and therefore in recycling mill wastes, have dropped dramatically in recent years as a result of similarly dramatic reductions in the levels of these materials in inks and pigments (Das and Houtman 2004).

3.3.2 Air emissions

Air emissions from pulp mills are primarily made up of particulates, hydrogen sulphide (impact locally by a foul smell), oxides of sulphur (SO_2) and oxides of nitrogen (NOx)(responsible for acid rain). Micro-pollutants include chloroform, dioxins and furans, other organo-chlorines and other volatile organics compounds (VOC)(mainly linked to photo-oxidant formation and human toxicity) ((Stanley 2001), (Das and Houtman 2004) and (Biermann 1996). As with liquid effluent discharges the levels of emissions are highly dependent upon the type of process technology employed and individual mill practice. Another important factor is the fuel type and quality. Technologies used today eliminate most harmful gas and particulate emissions.

The main concerns about dioxin and furans are linked to sludge disposed of via landfill or incineration. Dioxins are also known to be present in mill flue gases.

The contribution of the paper industry to global warming has been an ongoing debate for several years, with some suggesting that the absorption of carbon dioxide by plantation forestry more than offsets the emissions of greenhouse gases caused during the production (energy intensive), transportation and disposal of pulp and paper products. A study by the International Institute for Environment and Development (IIED) (1996) dismisses this argument, concluding that the paper cycle results in the net addition of some 450 million $C0_2$ equivalent units per year.

3.3.3 Solid Waste

Solid waste from paper manufacture ranges from 10- 250kg/t (dry equivalent) (Biermann 1996) Disposal is usually to landfill. Other experimental disposal techniques include using the waste as a soil improver but as with all disposal options, there is some concern about possible dioxin and heavy metal contamination (Smook 1996).

Relatively large amounts of solid waste and sludge result from the production of recycled paper although, obviously, overall recycling reduces waste volume. This waste comes in the form of bale wrappers and wire, sorting rejects and, more significantly, pulping and de-inking sludges, comprising water, cellulose fibre, fillers and ink. The sludge is contaminated with heavy metals and is of concern with respect to direct landfill, incinerator ash disposal and composting (McKinney, Howard et al. 1995).

3.3.4 Energy requirements

The pulp and paper industry is a major energy consumer. Whilst chemical pulping actually requires more energy than mechanical, the processes are largely energy self-sufficient through the burning of waste products to generate steam. Mechanical pulping and recycled paper cycle however requires imported electrical energy derived from fossil fuels (Stanley 2001). Electricity from fossil fuel is mainly linked to global warming.

3.3.5 Water

Abstraction of water for the industry is significant both in terms of quantities used and the possible damage caused to the adjacent environment. Problems associated with water use include increased sedimentation, increased water temperature, loss of habitat diversity, possible concentration of toxic material and lowering of water tables (Stanley 2001). It should be noted that nowadays mills work on a close loop water system and thus re-use most of the water within the process (IIED) (1995).

4 METHODOLOGY AND MODELS

This chapter outlines the methodology used, the purpose and scope of the study and its boundaries. It goes on to present the selected scenarios to be modelled, the software used for the modelling and data sources. It concludes with the characterisation factors chosen and the two valuation methods applied to the inventory results.

4.1 Goal and scope definition

4.1.1 Purpose

This study is intended to help the political decision-makers to understand and evaluate the environmental impacts associated with cardboard packaging wastes being recycled, incinerated or landfilled. Specifically the purpose is to assess the impacts associated with a range of recycling target levels for cardboard packaging waste within the context of the UK. LCA is widely used in political circles to justify decisions taken. However, it has to be highlighted that LCA does not provide an answer to a problem, but rather will help policy-makers to make an informed decision. LCA enables most environmental impacts to be quantified and maps out which impacts contribute to which problems, e.g. CO_2 emissions contribution to global warming.

4.1.2 Scope

The study took a broad national perspective, taking an overview of the issues involved. The geographical scope of the study was limited to the UK. The input data has been modelled using the computer package PEMS, developed by Pira, which enables the user to enter relevant data sets and produces an inventory. It also allows the user to evaluate the inventory results, using a range of assessment methods.

4.1.3 Functional unit

The functional unit for the systems is one tonne of cardboard packaging waste. Cardboard packaging was chosen as it represents 56% of paper/board packaging waste (Medis, 2005).

4.1.4 Boundaries

LCA needs to have boundaries in order to define what is being accounted for and assessed within a model, and what is being left out. The present research concentrated on the cardboard packaging life cycle from raw materials to waste management, including recycling activities.

The starting point of the studied system has been based upon literature review findings. McDougall et al (2001), Ekvall (1999), Ekvall and Finnveden (1998) and Stromberg et al, (1997) Stromberg et al (1997) all discussed the issue that today wood used by the pulp industry is the by-product of other industries, such as the furnishing industry, and forest management activities like tree thinning. It was thus decided not to take into account forestry impacts within the system under study. The boundaries for the system start when wood chips are delivered to mills.

The system takes into account raw material production, cardboard manufacturing, transport and waste management activities (including landfill, incineration and recycling). Credit for offsetting virgin material requirements is included in the system. Electricity usage and transport steps are also considered throughout the system.

The system boundaries end at the point of final disposal (i.e. landfill or incineration) for the fraction of cardboard packaging waste that is not being recycled.

4.2 The scenarios

4.2.1 The base scenario

The research concentrates on cardboard packaging waste, the targets chosen to develop the models are not to support any existing targets or propositions for new targets. They have been chosen to represent the current situation in the UK, and in other Member States in 2001, as well as the potential impacts from future changes in the Directive 94/65/EC targets. Figure 4.1 is a diagram of the base scenario.

The system studied concentrate on cardboard packaging. The system is developed so that environmental impacts associated with raw materials, cardboard manufacturing, waste management activities, energy input and transport activities can be assessed and compared between the scenarios.

The researcher developed the scenarios. The base scenario was decided at an early stage as it represents the UK situation. Scenario 3 (70% recycling) and 4 (80% recycling) were included due to the level of recycling taking place in some Member States. As the Directive and its targets were being reviewed, it was obvious that the revised target level should also be modelled. Once the modelling started it seemed obvious that extreme scenarios (100% landfill or incineration and a lower recycling level (35%) recycling) should also be included.

The base scenario (scenario 1) is representative of the UK situation in 2001 in reaching the Producer Responsibility Obligations (Packaging Waste) Regulations targets. The base scenario is thus looking at 53% recycling, with the 47% waste left being split between incineration (4.23%) and landfill (42.77%).

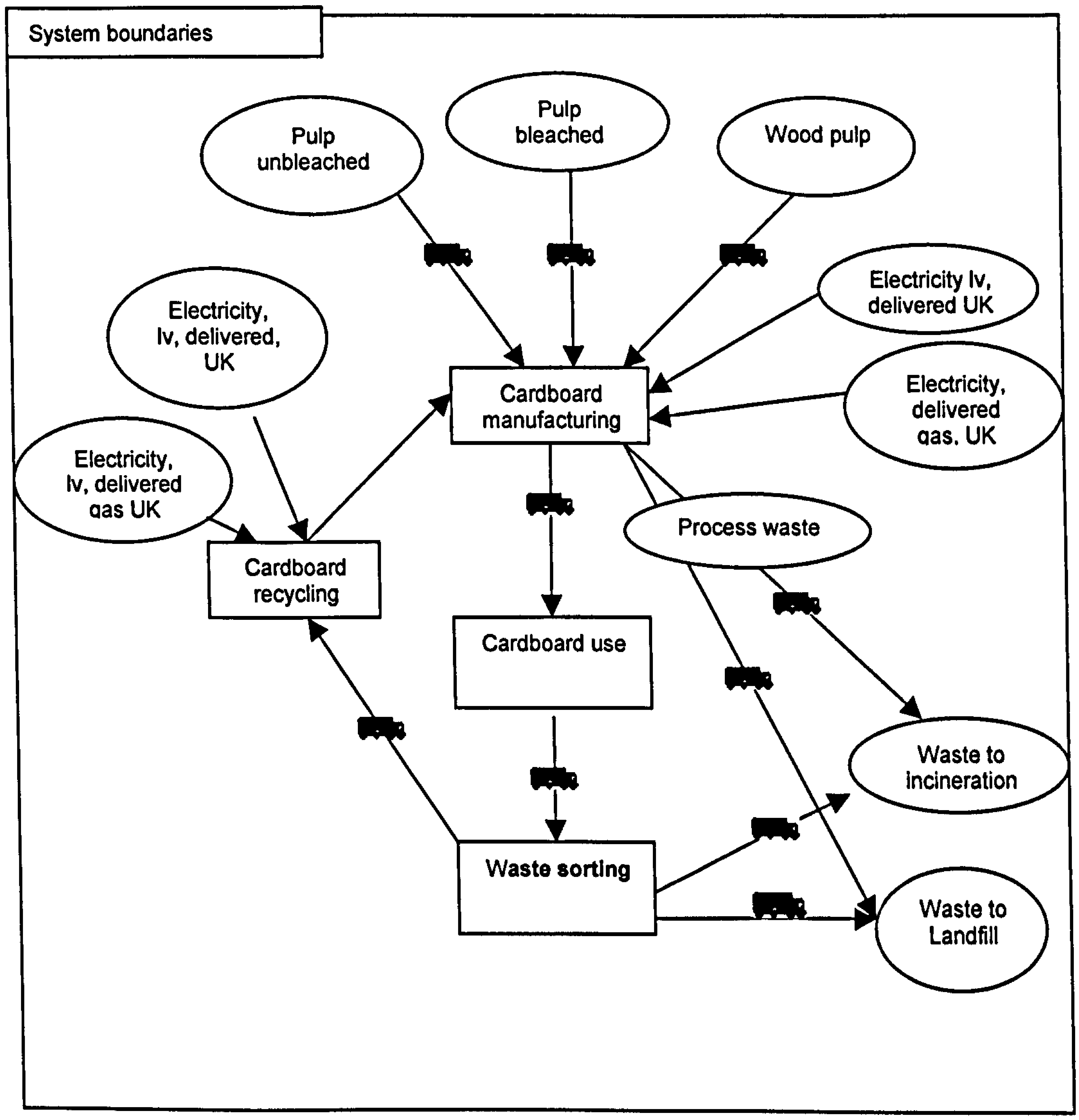

Figure 4.1: Simplified diagram of the LCA modelling of the scenario

4.2.2 The model

The following outline what is being included in the scenario and what assumptions had to be made.

The model includes:

- Input of wood pulp: it was identified from the literature review that approximately 94% of the wood used by today's European cardboard industry comes from managed forests in Europe, the other 6% comes from North America (PPIC, 2000).
- Input of recycled fibres: the model is assuming the use of 67% of cardboard waste as raw material (for the manufacturing of cardboard). Indeed, the literature review highlighted that in 2001, overall waste paper/board represented 67% of all fibres used in production processes in UK mills (ERM, 2002).
- Cardboard manufacturing data are from the PEMS database and BUWAL 250 (SAFEL, 1998). The data includes burdens associated with chemicals (and their production), waste, water and energy required in the process.
- The waste management module allocates the waste to landfill, incineration and recycling. The overall waste management situation in the UK in 2001 is that if cardboard packaging waste is not recycled then 93% are landfilled and 7% are incinerated (RDC et al., 2003). The incineration component includes the recovery of energy produced by burning the waste and credits the system with it.
- The recycling module includes the environmental burdens associated with chemicals, water and energy required by the process and waste emissions. The boundaries of this module begin with the delivery of waste paper to reprocessing facilities and ends with recycled fibre as raw material.
- Transport is calculated as burdens/ km tonnes. Transport is taking place between several modules and is represented by the lorries in figure 4.1. Each individual transport stage (or lorry in figure 4.1) is associated with specific distance, type of transport (rail, road or sea) and size (i.e. truck of 33t or 17t). The data relate to the type of vehicle and road conditions (i.e. urban, rural or motorway) and is extracted from PEMS database. The transport distance to landfill and incinerator on average for the UK is 25 km, which assumes that they are situated locally. These average distances to waste disposal and recycling faculties are extrapolated from the RDC and Pira study (RDC et al., 2003);. Regarding distances associated with the waste module, only the data

relating to high-density populated area (i.e.> 200 inhabitant/Km^2) are considered, as this represents 84% of the UK population. The transport distances of raw materials input (wood, and pulps) assume that they are from Europe, as 84% of UK imports are from Europe. The input of raw materials from the UK is only 6% of the overall raw materials inputs. The transport requirements between waste sorting and reprocessing facilities are based on averages from Valpack (2003). The cardboard manufacturing and use are also derived from the RDC and Pira study (RDC et al., 2003) and (SAFEL, 1998). All transport returns are assumed empty. This assumption will be tested in the sensitivity analysis.

- Energy input, the data are relevant to the UK as low voltage energy delivered, which is representative of the energy delivered from the national grid and gas supply. Delivered energy means that the energy loss during transmission and distribution are included, if undelivered then the data don't account for transmission and distribution of that energy. These losses are attributable to the heating of power line by the electrical current flowing through them. In the UK and heat loss. In the UK the distribution loss is around 6.5% - 7% (Ofgem, 2005).
- The data for energy and process efficiency are based on 1000kg of product being manufactured. PEMS

The energy requirements, transport and efficiency processes are automatically adjusted by PEMS in relation to the functional unit and any changes in flows around the system when compiling the results. Thus when changing the level of recycling taking place, the flows of materials going to landfill, incineration and recycling are amended. The flows into cardboard manufacturing are amended to reflect higher recycling input and lower virgin materials inputs. The system then recalculates energy and transport depending on the quantities in the flows.

4.2.3 Further scenarios

Further scenarios are modelled to evaluate and compare how environmental impacts might change when recycling levels are amended. The scenarios are as follow:

- Scenario 2 models 60% recycling with the 40% left being divided between landfill (37.2%) and incineration (2.8%). These target levels represent the minimum target for paper packaging set by the European Directive 94/65/EC revision and to be reached for 2008 by Member States (see section 2.1.4),
- Scenario 3 models 70% recycling with 30% landfill, which are levels already reached by some Member States,

- Scenario 4 models 80% recycling and 20% landfill (Germany reached 85% in 1997),
- Scenario 5 models 35% recycling and 65% landfill,
- Scenario 6 models the impacts of 100% landfill,
- Scenario 7 models the impacts of 100% incineration.

Scenarios 5, 6 and 7 are modelled to understand what environmental burdens exist at much lower recycling levels, and at 100% incineration and recycling, but also to enable comparison with the level of recycling planned by politicians.

The scenarios modelled in the study assume that cardboard packaging waste is being recycled and reused within the system. It is also assumed that all scenarios could be achieved with the current reprocessing capacity in the UK. This is a theoretical situation as the UK overall paper and board industry has reached its maximum recycling facilities, when the recycling level for paper/board packaging waste is at 53% in 2001. Thus to reach much higher targets there is a need to take into account either the development of new facilities and markets or greater transport distances if cardboard packaging waste was to be exported. An extra analysis is conducted and presented in section 4.3 to model how the scenarios' impacts might be affected when considering different transport distances and cleaner energy.

Due to data availability limitations the modelling of new reprocessing plants are not included in the modelling. It can be assumed that with increased reprocessing facility, initially the overall environmental impact of one tonne of recycled cardboard packaging waste would increase, as facilities might not run at 100% capacity. Yet this is debatable as new technologies are likely to results into improve environmental performances of the process.

Modelling was not undertaken to assess the impacts associated with levels of recycling greater than 80% and it should not be assumed that further increases in recycling would continue to be beneficial. Collection of the remaining 20% of material would involve significantly greater travel distances, the recovery of recyclable materials is likely to become less efficient, and it is also doubtful that a market for such a high level of recycled fibre will actually exist within the UK in the short to medium term.

4.3 Software

4.3.1 Review of LCA software

Several LCA software packages are available on the market, they can be distinguished through different elements: price, the developers, computer requirement and interface for running the software, database available and application. The main market players today are SimaPro developed by Pré-Consultant, TEAM/DEAM developed by the Ecobilan Group, the Boustead model developed by Boustead, and PEMS developed by Pira International (Packaging Industry Research Association). Each package has its advantages and drawbacks, and choosing the appropriate tool can be difficult.

Menke et al. (1996) and Jonbrink et al. (2000) offer a comprehensive overview of LCA software. Both looked at a large number of tools, 14 and 24 respectively. The (Jonbrink et al., 2000) survey showed that different software tools are intended for different types of users: environmental or design engineers, or all types of users; and designed for different types of LCA: function-based LCA, screening LCA, accounting LCA, or all types of LCA.

Demonstration version for SimaPro, PEMS, TEAM and LCAit were obtained or downloaded and assessed by the researcher. The choice of software was dictated by:

- Databases relevance and availability
- Its interoperability with other database packages (i.e. embedded coding to import other database),
- Ease to input user define data, and management of data
- Characterisation and valuation methodologies delivered with the packages.
- Reporting and results manipulation was also an important factors, i.e. can the graphs or inventory data be exported to other packages (excel or word).
- Finally the user friendliness of the products.

After reviewing the literature and accessing demonstration version of selected software packages, the choice for the present study was made to use PEMS4, for the following reasons. PEMS complies with the ISO standard for LCA study, and is supplied with a large selection of characterisation models as well as valuation methodologies. Pira is a leading consultancy in packaging issues at all levels: development, improvement, research and environmental assessment. PEMS4 was developed primarily for packaging applications, and thus provides a relevant database to the case study. PIRA is a UK based company, which simplified training and support. PEMS had the highest user-friendly interface. Finally the price was attractive, as a large discount is applicable to academic use. However, this does not mean that any other package would not have been as good. Alternative tools might have been selected due to their specific focus on packaging or paper and pulp, such as KCL-Eco and LCAiT, however these two software packages are developed in Finland and Sweden and data provided with the package are only relevant to these countries.

4.3.2 Using PEMS

This section outlines how the models were developed and analysed using PEMS software. PEMS is based on Microsoft Access software.

PEMS works in three dimensions:

- A graphical interface in which the LCA system to be analysed can be created and edited
- A database manager, which allows the user to view, edit and administer databases
- A Reporting and graphic section to analyse system inventory, create report and graphs.

The software is relatively easy to use.

Building a model

When opening PEMS the user is in the graphic interface. The first step is to fill in project information detailing the name of the system, data used functional unit and any relevant comments. From here a model can be build by adding: processes – materials (inputs) – energy – transport and waste block. These are simply represented by squares or rounds with a symbol and a name, a truck represents transport. When adding a block the data to be attached to that block can either be from database or entered by the user. The transport block requires a type of transport to be chosen and the distances entered. The distances are calculated as burden/tonne km . All blocks are linked by flows, i.e. how much quantity is flowing from one block into another, but are otherwise independent. Once this is completed the functional unit for the model needs to be set, this will be the reference flow for compiling the results. For the present research the functional unit was 1000kg of cardboard packaging.

Saving the data

Once the system is compiled the results are saved. If the data used within the model are user specific then it needs to be saved to a user database. This is simply done by using the database manager tools. This can then be used in other modelling by selecting the user database.

Analysing the model

An analysis of the model can then be compiled. The results can be be presented for each block in the model, they can be analysed all together or a few can be selected. From here life inventory report or summary report can be produced. Characterisations and valuation can be conducted using set methodologies (the user chose which one to apply) or the user can define its on using the database manager.

The results are presented in graph format, double clicking the graph give access to the actual data. The graphs and the data can be exported to excel or word by using the copying function.

For each characterisation analysis it is possible to see the contribution of the most significant burdens/flow to a category, i.e. for global warming it will show the respective contribution to the impact of carbon dioxide and methane, and will group all other contributing emissions under 'other'.

For the inventory summary and impact assessment the graph can de scaled this enables to see all values on one graph. To calculate the actual values for a scaled graph it is necessary to multiply the value by the figure in bracket.

Comparing several models

There is the possibility to analyse several model simultaneously. To do this all models to be considered have to compiled and saved. In the reporting interface any saved models can be selected (here the overall results for each model will be presented, the breakdown by block is not possible). The same procedure as above can then be applied.

4.4 Data quality

Inputs (wood chips and waste paper) and outputs (emissions) vary considerably depending on the pulping techniques being used, paper production processes, distances between mills, waste collections, recycling facilities and technology, type of incinerators etc. Thus the model uses average figures. Although there are limitations to the applicability of average figures, it is consistent with the aim of applying LCA at a generic, policy-oriented level. The use of average data will identify the types and scales of impacts likely to arise from cardboard packaging waste recovery/recycling target options. Data in this thesis was derived from the PEMS4 database and from a additional sources discussed in the following section.

4.4.1 Environmental data sources

Electricity and other energy:

- PEMS database is sourced from the report "Environmental benefits of offset energy" (ETSU, 1996) the data are for the UK average voltage delivered.

Waste management (landfill and Incineration):

- PEMS database
- "Integrated Solid Waste Management: A Life cycle Inventory", (McDougall et al., 2001).

Materials:

- PEMS database,
- European Database for Corrugated Board Life Cycle Studies, (FEFCO et al., 2000).
- BUWAL 250: Life Cycle Inventories for Packaging, (SAFEL, 1998).

Transport

- PEMS database (vehicle emissions)
- Derived from RDC – Pira "Evaluation of costs and benefits for the achievements of reuse and recycling targets for the different packaging materials in the frame of the packaging and packaging waste directive 94/62/EC" (2003)
- Valpak, private conversation

4.4.2 Limitation in data

Average data are used for the modelling, this is relevant because of the political context of the research, site-specific data would not be relevant to the whole of the UK. However by using average some accuracy in the data will be lost.

By aggregating several data sources together for the modelling, the uncertainties within the results are increased. This is due that different database will use different methodologies to collect and compile their data.

4.5 Environmental impacts methodology

After compiling the model it needs to be analysed. Within an LCA a large array of environmental impacts can be assessed and evaluated.

4.5.1 Environmental impact categories

For this specific research the environmental impact categories chosen reflect the literature finding such as in Bloemhof-Ruwaard et al. (1996); Johnson (1993), Hanssen (1998) and Heijungs (1992) (also see section 2.2.6). It is also a fairly standard set of mainstream impacts studied through LCA. Impact assessment has been carried out for the following categories using the CML methodology (however global warming within the CML methodology is reliant on the Intergovernmental Panel on Climate Change calculation). They are all readily available in PEMS 4. the definition given below for each impacts is that within PEMS manual (Pira, 2000).

Global warming

The gases involved in the greenhouse effect all have the property of absorbing energy and emitting thermal infra-red radiation, thus an increase in the atmospheric concentration of these gases will change the absorption of infra red radiation in the atmosphere. This produces a rise in the average global temperature. The Global Warming Parameter was developed by the Intergovernmental Panel on Climate Change (IPCC) to express the potential contribution of different gases to the greenhouse effect, taking into account their different properties in lifetime and absorption level (Houghton, 1997). Expressed as carbon dioxide (CO_2) equivalent, the major contributors are carbon dioxide and methane (CH_4). Global warming is seen as a major threat to climate changes.

Acidification

Acidification results from acid deposition, which leads to a decrease in pH, of mineral content of soil and increased concentrations of potentially toxic elements in soil. These effects are caused by acid rain and the major pollutants associated with this are sulphur dioxide and nitrogen oxides. The potential to cause acidification is calculated on the basis of the number of H+ ions which can be produced per mole of substance, using SO_2 as the reference substance (Pira, 2000).

Eutrophication (also known as Nutrification)

Eutrophication is caused by the addition of nutrient to soil or water systems and leads to an increase in biomass. Any nutrient can have a eutrophication effect, but nitrogen and phosphorous are the most important. Substances with a potential for causing eutrophication are aggregated using nurtification/ eutrophication potential (NP'S) which are a measure of the capacity to form biomass compared to phosphate (Pira, 2000).

Human toxicity

This impact category is based on toxic effects such as disease of the respiratory tract, kidney failure or liver damage. The potential contributors are emissions of organic and inorganic compounds (including heavy metals) into the air, water or soil. The potential contribution is based on the evaluation of the pollutants absorbed by a human body, using limits such as Acceptable Daily Intake (ADI). The effects are given not in relation to a reference compound but in kilogram body mass (Pira, 2000).

Ecotoxicity

This category focuses on aquatic ecotoxicity, and is based on the toxic effects on organisms in surface waters. Ecotoxicity factors are based on the MTC values (Maximum Tolerable Concentration) of the EPA (Environmental Protection Agency, USA) which have been defined for emissions into water. The impact is given not in relation to a reference substance, but in m^3 water (Pira, 2000)..

Photo-oxidant formation

Also known as summer smog, this indicator concentrates on the formation of photo oxidants in the lower atmosphere, which are produced by the reaction of volatile organic compounds (VOC) with nitrogen oxides in the presence of ultra-violet radiation. The characterisation factors correlate with the intensity with which the pollutants contribute to the formation of ozone. The reference substance is ethene (C_2H_4) (Pira, 2000).

4.5.2 Valuation methodologies

Characterisation of the results is only a partial aggregation of the results. It classifies the inventory results enabling decision-makers to have an understanding of the potential environmental effects of the emissions quantified during the inventory. Yet the ability to base decision-making on characterised results alone can be limited by the need to determine which impacts carry most importance (weight) in the particular decision.

The present study aim is to establish the optimum level of recycling targets for cardboard packaging. This implies that an aggregated value reflecting overall impact is required.

The valuation/weights step rates the importance of different impact categories against each other. This is achieved by converting indicator results for selected impact categories to a common scale by using numerical factors based on values. This can potentially include a final aggregation to a single indicator. The impact assessment data may then been converted to monetary values through the application of economic valuation to each environmental impact category.

For the valuation step two approaches are available and have been discussed in the literature review in section 2.2.6.4. For this study two methods have been selected on the grounds that they assess the results using different approaches.

CML uses the midpoint approach and aggregate the results to give a single indicator value. The results are still presented as environmental effects such as global warming, eutrophication. This approach is deemed more 'accurate' than the endpoint techniques as it is based on scientific understanding of emissions effects to the environment (Hertwich and Hammit, 2001). However the results stay relatively abstract with regard to the units they are reported in, which can make it difficult for stakeholders to reach a decision.

- Eco-Indicator 99, perhaps the most known and used method, uses the endpoint methodology and attempts to predict damages to human health, ecosystem quality and energy. It then aggregates predicted damages in terms of a single indicator (Goedkoop and Spiriensma, 1999). While the midpoint methodology will stop at quantifying global warming potential or eutrophication potential, the endpoint methodology will translate these into potential endpoint damage to health or the ecosystem (Heijung et al, 2003).

The weighting schemes applied by both methods are available in appendices 4 and 5. Both methodologies are available in PEMS and have been used as set in the program.

4.6 Conclusion

This chapter concentrated on presenting the breadth and depth of the study. It described the scenarios developed for modelling with the underpinning assumptions and limitations. After a description of the software and the data sources, assessment techniques applied to the results of the modelling have been explained as well as a description of the chosen impact categories. The next chapter presents the results of the study.

5 RESULTS

This chapter is divided into three parts. The results of analyses of the scenarios described in chapter 4 are presented in sections 5.1 and 5.2. Section 5.3 presents the results of two extra scenario where first cleaner energy use is modelled and another one which considers greater transport distances. Section 5.5 concentrates on the valuation of the results using two types of methodology: CML and Eco-Indicator 99. Throughout this chapter the recycling levels are referred to by just the percentages. This is also true for the legend within the graph i.e. 35% means the model considered 35% recycling. Section 5.6 presents the results of two sensitivity analysis, the first looks at the transport assumption, the second one focuses on final waste disposal.

5.1 Life cycle inventory results

The life cycle inventory summarises all inputs and outputs within the system boundaries. The following results are of the life cycle inventory in terms of environmental burdens compared between all the scenarios.

The models are explained in details in section 4.2.1. The models include the whole life cycle of cardboard packaging, from raw materials to manufacture of cardboard, including energy requirement, recycling and final waste disposal. Transport between the stages is also included, see figure 4.1. The functional unit (the system is calculated by reference to that unit) is 1000kg of cardboard packaging. Every time the recycling level is changed at the waste sorting level, the model is recalculated to reflect the changes in the burdens per 1000kg of cardboard packaging through its life cycle. PEMS recalculates automatically all flows and burdens through the system as recycling levels are amended.

The following results present burdens per 1000kg of cardboard packaging (in kg, unless stated otherwise); and are scaled (the actual values have been multiplied by the value in brackets) so that the different impacts can be represented in one figure. If the results were not scaled many of the burdens in figure 5.1 could not be seen, as for example the heavy metals are released in only very small quantities (0.49 kg at the highest but are perhaps some of the most harmful burdens) and other emissions are released in much larger quantities such as gases (400 kg at the highest)

Figure: 5.1 presents a general summary of the compiled inventory using the default summary inventory function of PEMS, for which category definitions are available in appendix 2. It can be concluded that while every burden is reduced with higher recycling, some reductions are more substantial than others.

In some cases (e.g. VOC-halogenated) it is apparent that the choice of waste management strategy is critical (i.e. differences between the maximum and minimum impact account for almost all the system impact); in others (e.g. VOC-aromatics) it is evident that impacts are much more moderately affected by choice of waste management strategy.

The scenarios that have been assessed suggest that the various impacts all respond in a linear manner to altered levels of recycling. There are no 'cusps' in burdens that would indicate that the optimum level lies somewhere within the examined range of scenarios.

In all but five of the twenty-seven burden categories considered the 100% incineration scenario appears to be a better option than 100% landfill; the 100% incineration is also the only option to be credited with negative burdens in any category. From Figure: 5.1 it also appears that the burdens associated with 100% landfill are similar to those associated with the intermediate levels of recycling.

The largest environmental burden attributable to the scenarios in terms of kg is water (unspecified) usage, which is reduced with higher recycling levels. However water usage only falls by around 7% from 35% to 80% recycling. This can be explained by the continuous effort from industry to reduce water usage and maximise closed loop recycling and re-use of water within a plant.

The second largest burden comes from carbon dioxide emissions (Gases A, in Figure: 5.1). Here all CO_2 emissions are added together; this burden decreases as more recycling takes place. It has to be highlighted that incineration has the largest contribution to CO_2 emissions, 59% more than at 80% recycling. The reduction in CO_2 emissions between 35% and 80% recycling levels amounts to 14.7%.

The third largest burden is attributable to fossil and biotic reserves (energy input). In Figure: 5.1 it can be seen that incineration is the scenario with the lowest contribution, 61% less than at 80% recycling. This is explained by the credit to the system for energy generated by burning waste. Looking at all other scenarios, the energy input decreases by almost 10% at 80% compared to 35% recycling.

However, it has to be highlighted that these burdens are not directly comparable as they are in different units. Furthermore, the biggest values are not necessarily the most significant (e.g. 1kg of heavy metals released to water has much more environmental impact than 1kg of water usage). Section 5.2 intends to interpret how the environment is being affected by these burdens.

The largest burden reduction achieved by increasing recycling level is in the family of VOC emissions. Thus VOC alkanes are more than halved (54.4%), and VOC halogenated burdens fall by 49% when recycling levels increase from 35% to 80%.

Some other noticeable reductions in burdens when comparing 35% to 80% are: a 52% decrease in solid waste generation, 29% in toxic substance releases and 23% reduction in heavy metals (airborne).

The inventory summary can not be used to establish the significance of the burdens in term of environmental impacts, but summarises all emissions which are attributable to them. It takes into account water usage fossil/biotic resources and mineral reserves as depletion of finite natural resources.

The life cycle inventory results are interesting, but there is a need to conduct a life cycle assessment to understand which emissions contribute to which environmental impacts, and how higher recycling levels increase or minimise the occurrence of impacts.

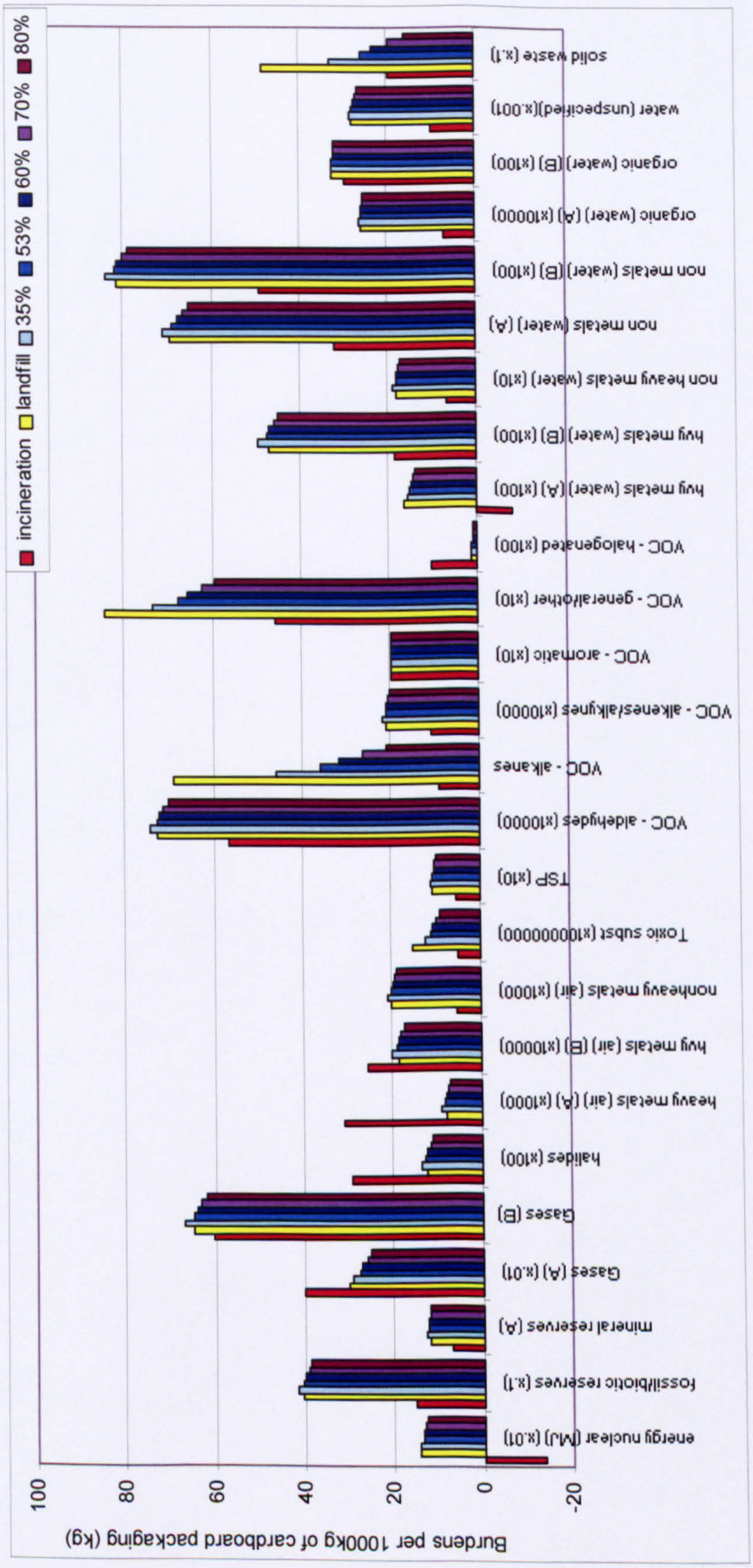

Figure: 5.1 Inventory summary

5.2 Life cycle impact assessment

5.2.1 Introduction

This section presents the life cycle impact assessment (LCIA). The LCIA enables relative environmental impacts of cardboard packaging waste recycling targets to be assessed. When interpreting the LCIA, it should be noted that the LCIA results establish potential impact. It is important to highlight that the impacts are relative to each other (i.e. between different targets). Similarly, where large differences do occur between the impacts of one scenario and another, this does not necessarily imply that the difference is significant in real terms, if the size of the greater impact remains very small in real terms.

The principle and methodology of LCIA has been presented and discussed earlier on in section 2.2.6, but the following is a short reminder of the different stages:

Characterisation: quantifies the relative contribution of individual environmental burdens to specific impact categories. It converts impacts into chosen units: e.g. amounts of CO_2 and methane into their global warming potential.

Normalisation: can be used to relate the results of the classification/characterisation steps to total emissions in a certain area over a given period. Normalisation increases the comparability of data from different impact categories and provides a better basis for weighting options by relating the magnitude of impacts in different categories to reference values; an example of a reference value could be the total contribution to an impact category by a nation.

Valuation/weighting: aims at converting and possibly aggregating indicator results across impact categories resulting in a single value, sometimes with monetary measures. Weighting has always been a controversial issue, mainly due to its reliance on social, political and ethical values, whereas the preceding steps are based on traditional natural sciences.

5.2.2 Analysis of standard scenario results

5.2.2.1 Overview of results

Figure 5.2 presents an overview of the categories that are presented in more detail later on. For each category the impact value for each scenario is shown as a proportion of the present day situation (i.e. 53% recycling). This form of presentation enables those impact categories that are significantly affected by the choice of waste management strategy to be readily identified. In the case of global warming, it may be seen that the benefits of increasing recycling levels up to 80% are significant (-20%), yet for eutrophication they are more moderate (-5%). It is also evident that incineration has a largest negative impact on human toxicity, whilst the 100% landfill scenario is associated with high levels of global warming. For all other categories, incineration scores well, particularly against lower levels of recycling.

The impact categories chosen to be presented are the ones that have the greatest contributions and known to be relevant to the paper industry, and also the ones that are on the political agenda (such as global warming with reference to the Kyoto Protocol). For this reason the results focus on the following:

- Global warming
- Human toxicity
- Acidification
- Ecotoxicity
- Eutrophication
- Photo-oxidant formation
- Energy resources

A summary of the impacts categories and how they impact on the environment can be found in section 4.5.

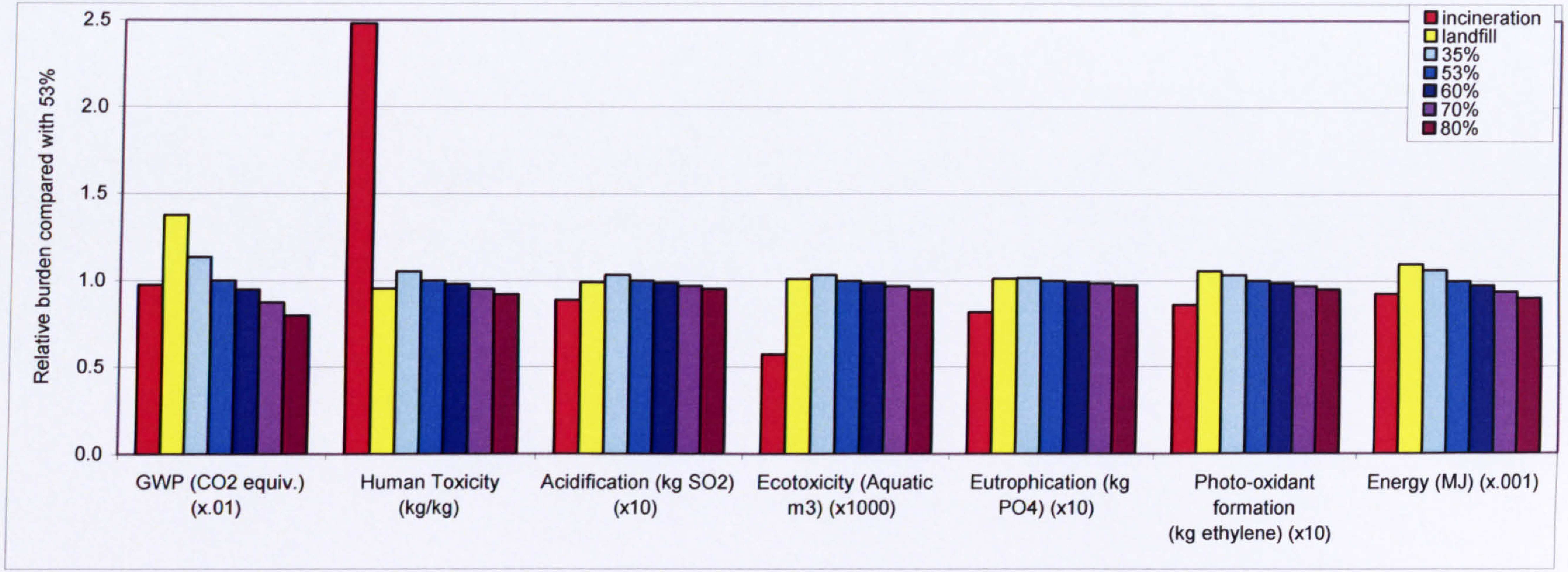

Figure 5.2: Relatvie burden compared with 53% recycling

5.2.2.2 Global warming

Global warming is calculated as CO_2 equivalent, and the main attributable environmental emissions are CO_2, methane, non-methane VOC, NOx and SO_2. Figure 5.3 shows that the system's total contribution to global warming reduces with higher recycling levels. Looking at the two main contributors, namely CO_2 and methane, proportionally greater reductions are achieved for methane (54%) than for CO_2 (13.5%) when recycling level goes from 35% to 80%. Looking at 100% landfill the impact is always greater than recycling, yet for incineration it is not as clear-cut. Indeed CO_2 emissions are 46% higher than at 80% recycling, but methane emissions are 57% lower than that at 80% recycling. This is consistent wit the fact that landfills are responsible for 26% of the overall UK methane emissions (Baggott et al, 2004).

At 35% recycling, CO_2 represents 63% of the total global warming potential and methane 38%. At 80% recycling the overall impact is reduced by almost 30%, but the contribution of CO_2 emissions represent 78% and methane only 25%. Thus the choice of waste management has a greater impact on methane emissions than on CO_2.

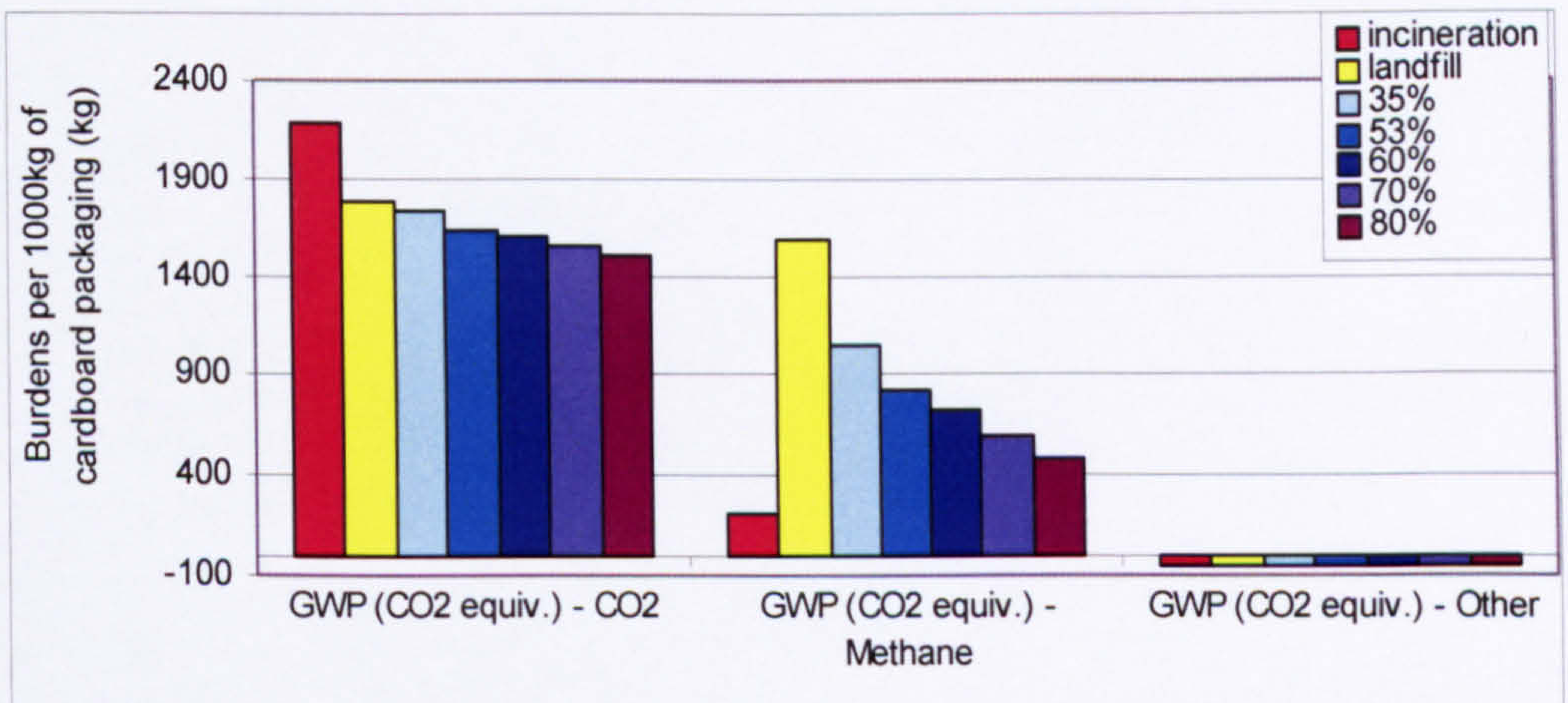

Figure 5.3: Global warming impact (kg CO_2 equivalent)

5.2.2.3 Human Toxicity

Human toxicity is mainly concerned with emissions that are attributable to carcinogenic, radiation and respiratory (inorganic and organic) impacts.

Figure 5.4 illustrates the contributions of the relevant burdens for this impact, and shows that incineration is the worst scenario. Hence, arsenic (As) emissions are reduced by 95% at 80% recycling compared to 100% incineration. When considering the recycling scenarios, As emissions are lowered by 40%, and Pb by 34%, when recycling rises from 35% to 80%, their contribution to the overall impact is relatively small. The dominant contributor emissions are SO_2 and NOx, respectively contributing about 40% and 30% to the impact.

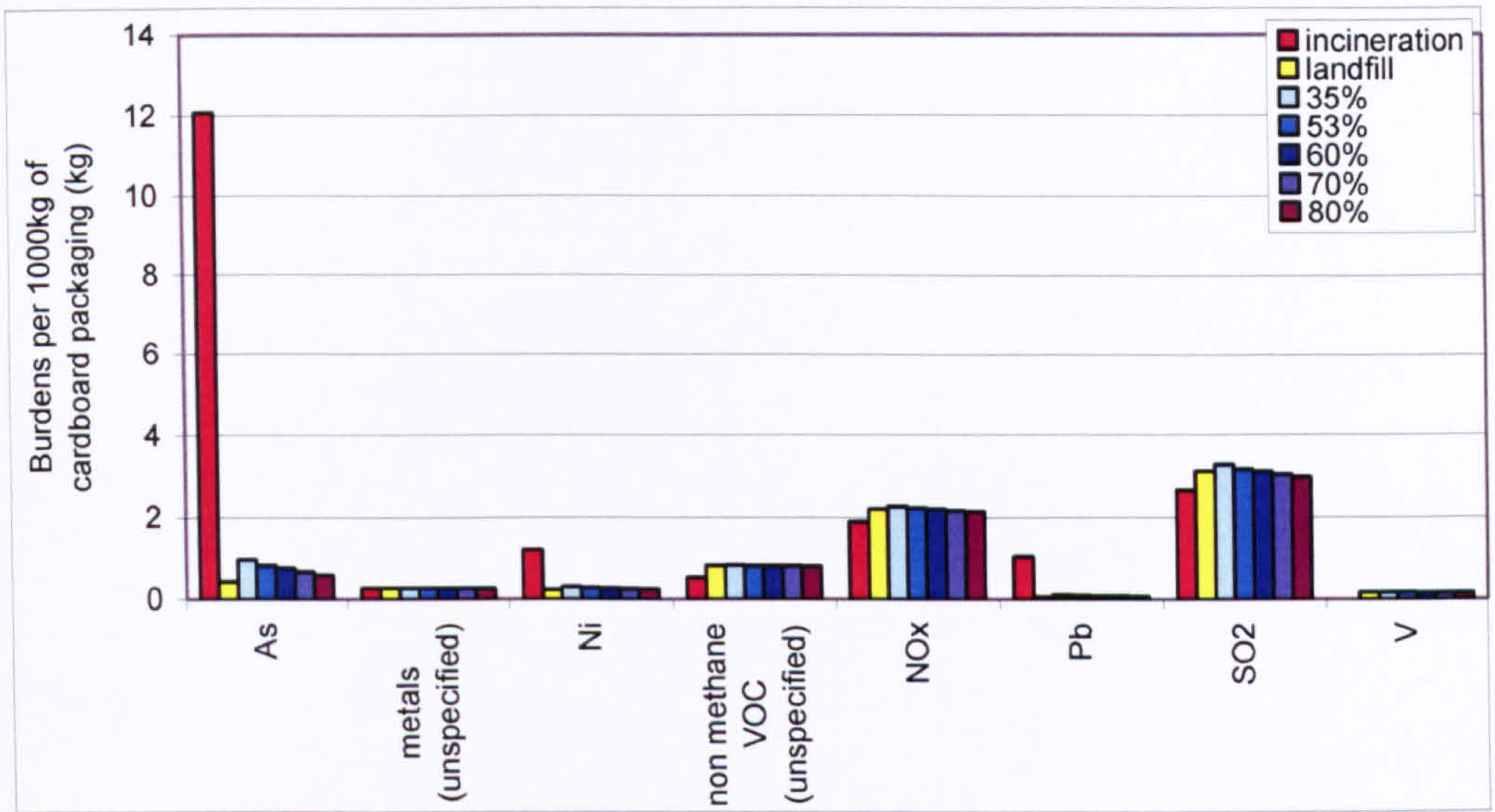

Figure 5.4: Human toxicity impact

5.2.2.4 Acidification

Figure 5.5 presents the main burdens contributing to the acidification impact; SO_2 and NOx dominate the contribution with respectively around 53% and 41% responsibility for the potential occurrence of the impact. These ratios remain almost constant within each of the recycling scenarios. Incineration has the lowest contribution, mainly due to the lower levels of NOx and SO_2 emissions both 11% less than at 80% recycling. However, HCl emissions are 64% higher in the incineration scenarios compared to 80% recycling. Emissions of HCl are reduced by 19%, and by 18% for HF at 80% recycling compared to 35%. The largest augmentation is in acid, with 35% increase at 80% recycling compared to 35%, even though the total level of emission is very small.

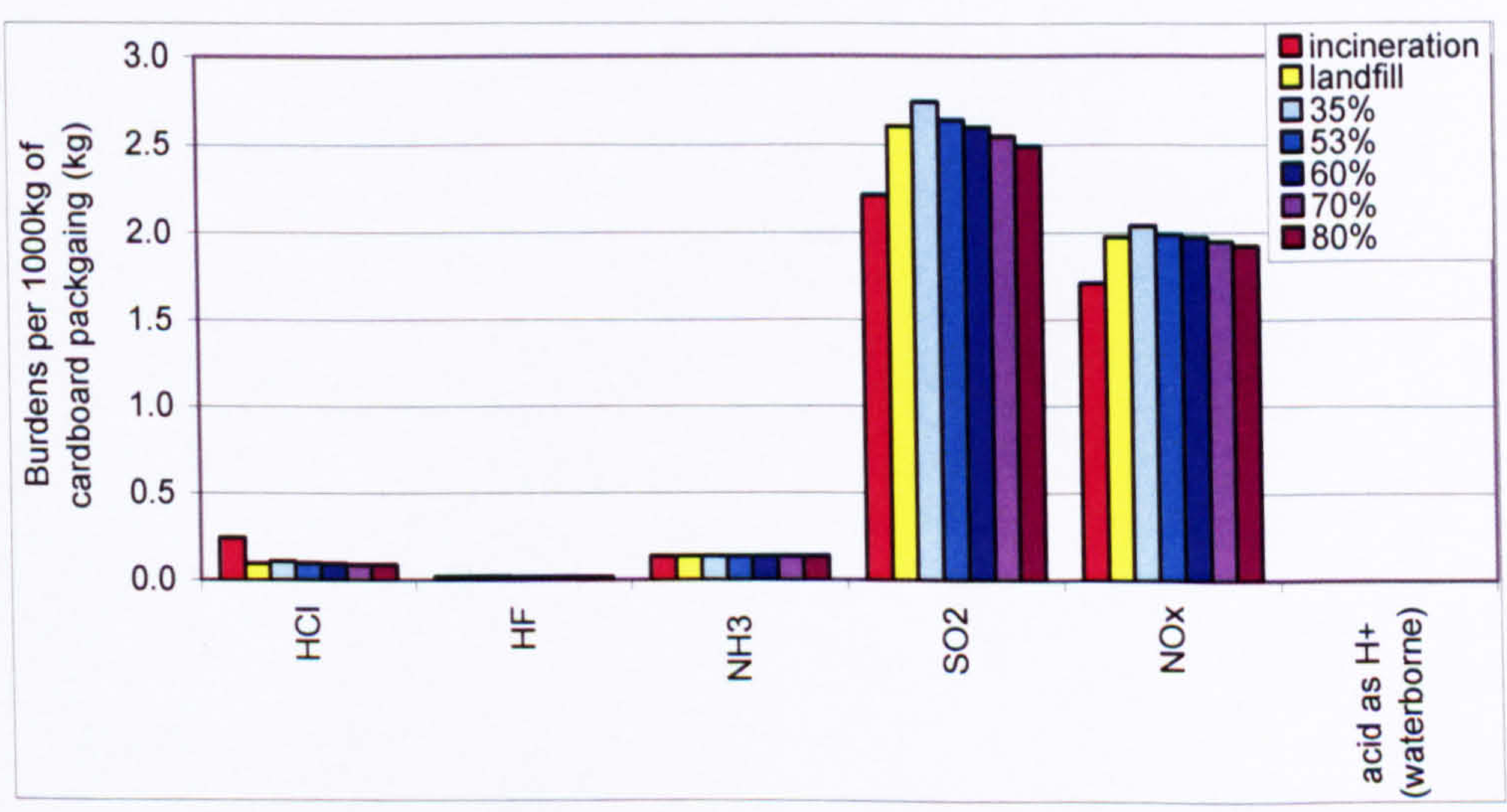

Figure 5.5: Acidification impact (kg SO_2 equivalent)

5.2.2.5 Ecotoxicity

In the context of the paper industry, ecotoxicity is mainly concerned with water emissions, but also transport. Figure 5.6 outlines the contributors, few of which are moderately affected by the scenarios. This seems to indicate that burdens relevant to ecotoxicity are occurring mainly at the manufacturing stages (see section 5.2.3). Incineration has the lowest impact with most emissions being greatly reduced, in the order of 50-60% (apart from metals and nickel (Ni)). Looking at the recycling scenarios, all emissions (except for metals) benefit from a reduction in the order of 10%, Cadmium (Cd) being the largest at 13% when comparing 35% to 80% recycling. Ni sees the smallest reduction with only 3.5%.

The metals are constant which seems inconsistent. When looking at the results for each scenario individually, it appears that raw materials input (mainly wood pulp) and recycling are the principal contributors to metal emissions. Thus at 80% recycling metals emissions are actually increase in the order of 10%, but at the same time the one from raw material is also reduce in the same order of magnitude.

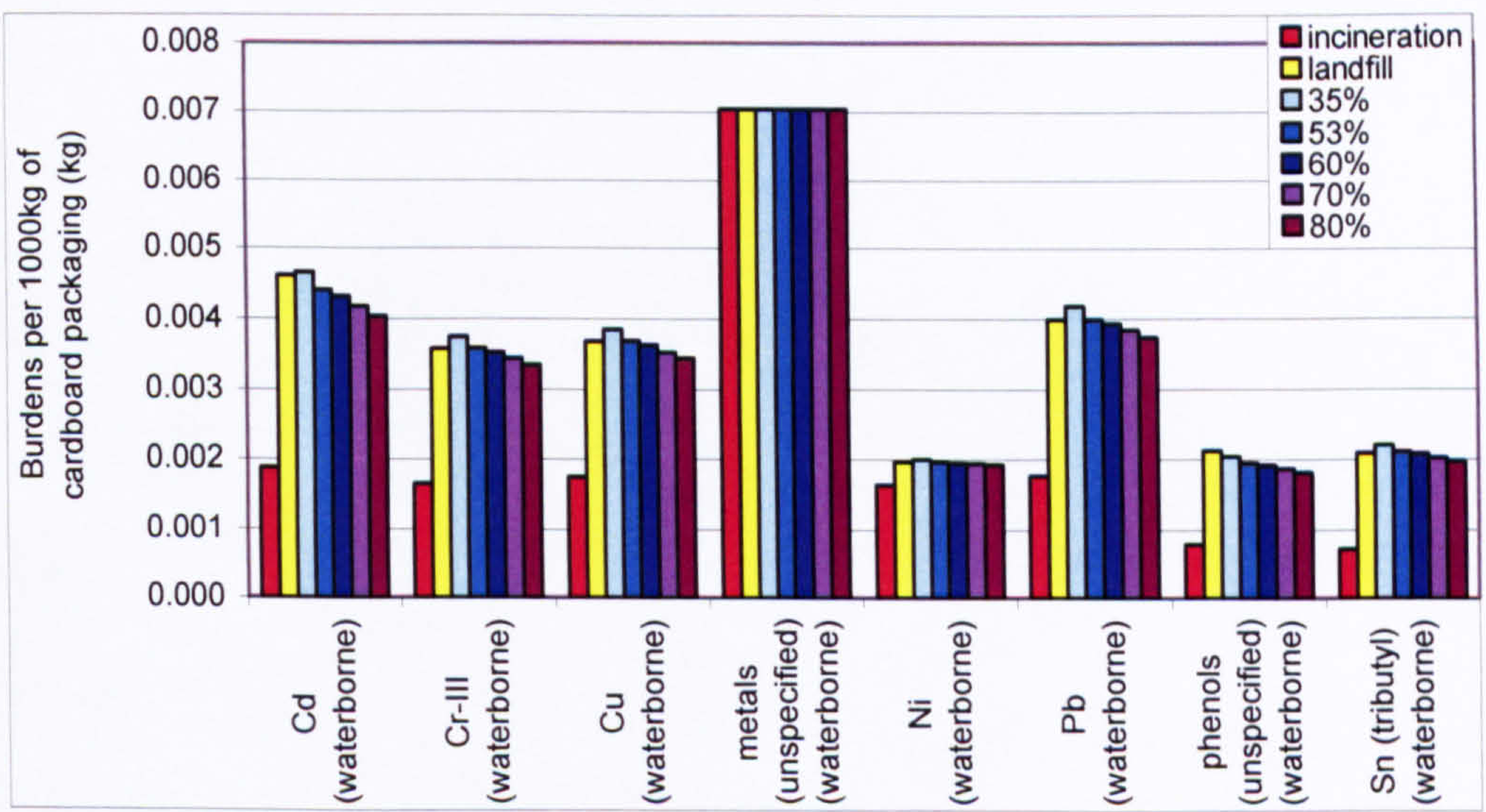

Figure 5.6: Ecotoxicity impact (waterborne)

5.2.2.6 Eutrophication

The main burdens contributing to Eutrophication (figure 5.7) are phosphates, which are responsible for half of the impact occurrence, NOx at 25% and nitrogen (organic bonded, waterborne) at 13%. Incineration has the lowest contribution; nitrogen (organic bonded, waterborne) emissions are lowered by 91%, 70% for N (waterborne) compared to 80% recycling. Two emissions, N (waterborne) and COD, noticeably reduce with higher recycling, by 54% and 20% respectively. However it has to be emphasized that these emissions have a very low contribution to the overall impact.

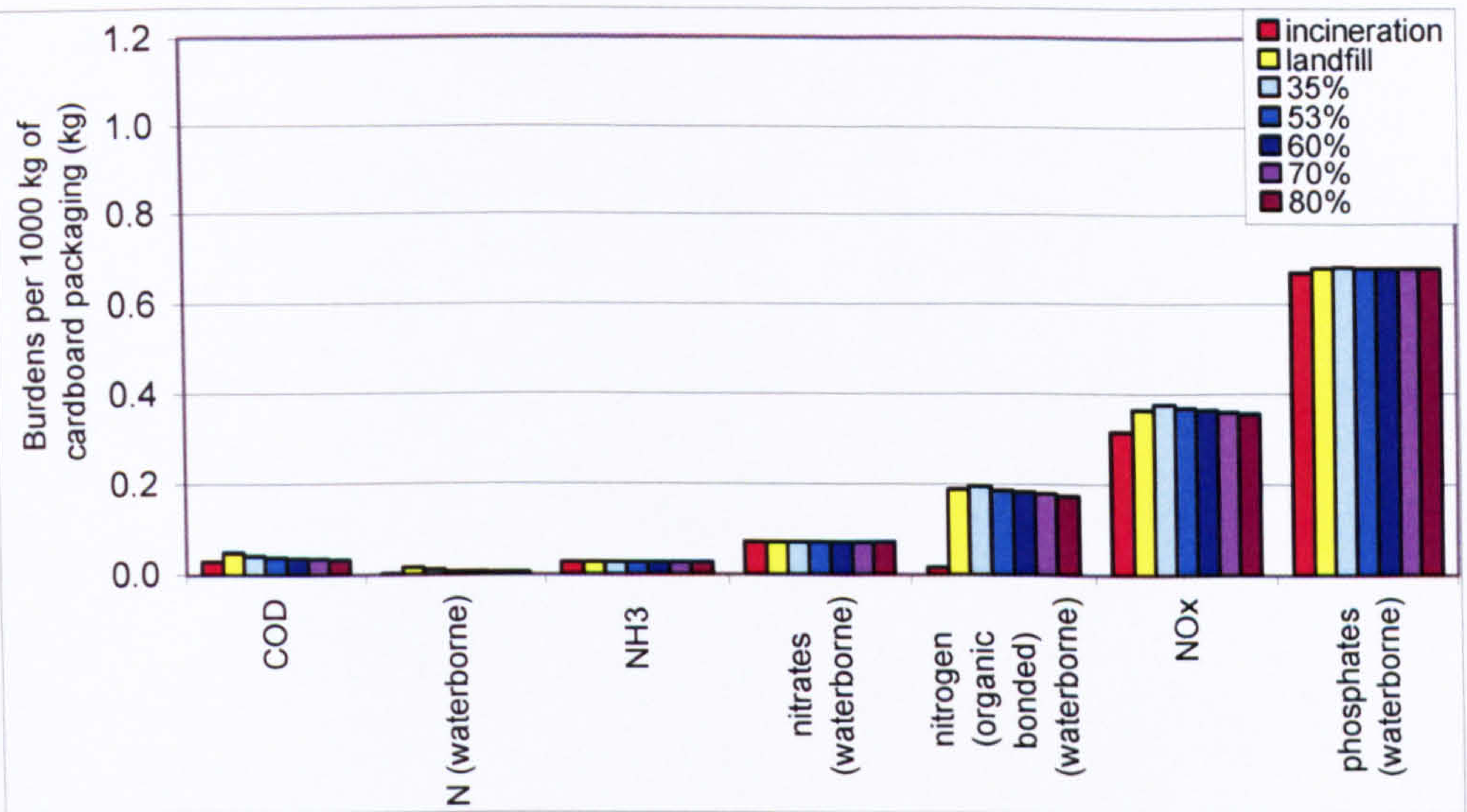

Figure 5.7: Eutrophication (kg PO_4)

5.2.2.7 Photo-oxidant formation

Photo-oxidant formation is dominated by the contribution of NOx at around 58%, and aromatics on average 35% of the overall impact. From Figure 5.8 it can be noted that aromatics and non-methane VOC are constant whatever the scenario. This indicates that the emissions are linked to the manufacturing stage rather than recycling or waste disposal. Xylene is also hardly affected. For all emissions, except HC, 100% incineration represents the best scenario. HC and methane emissions are greatly reduced at higher recycling level, HC benefits from a reduction of almost 60% at 80% recycling compared to 35%, and methane by 54%.

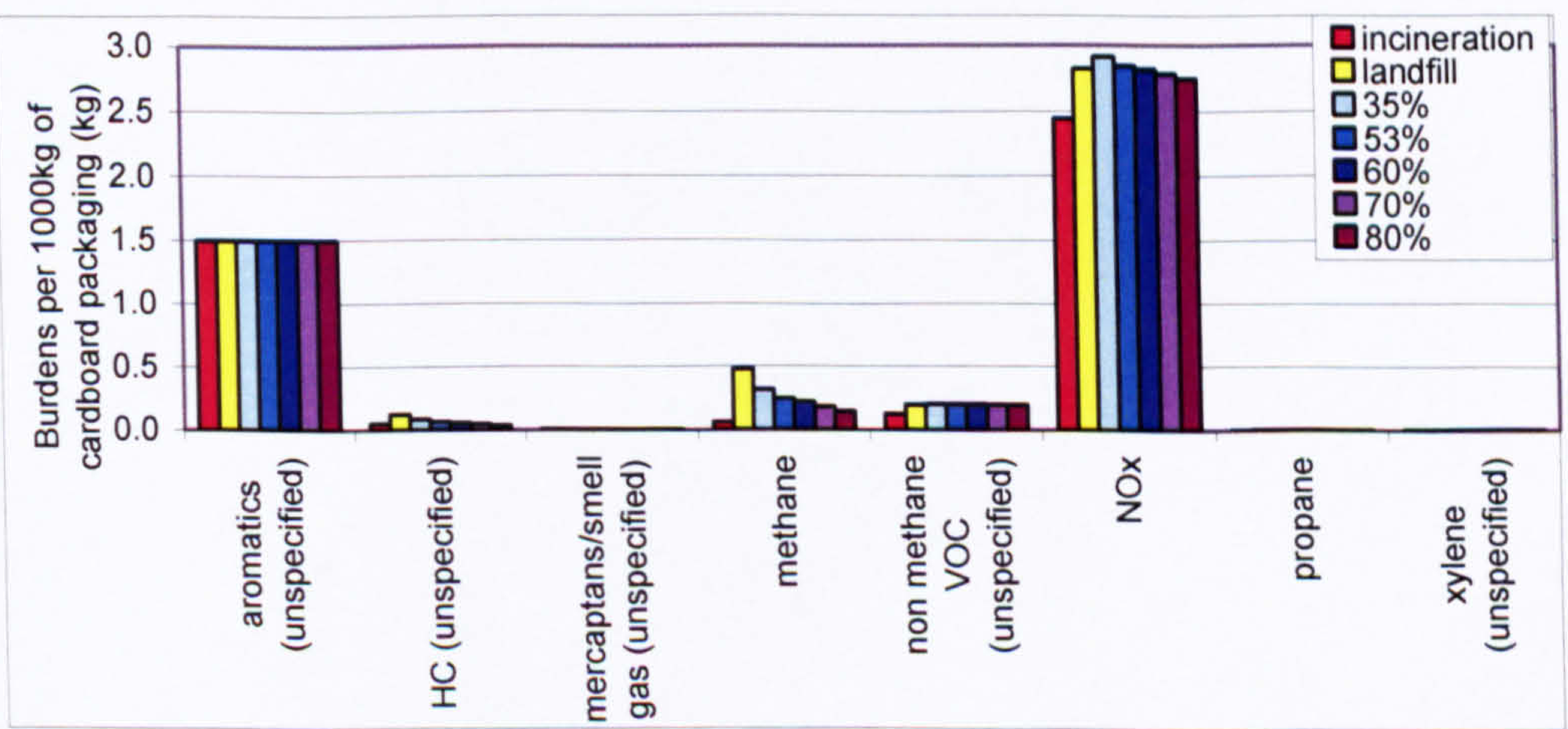

Figure 5.8: Photo-oxidant formation

5.2.2.8 Energy resources

Figure 5.9 illustrates the different uses of energy resources within the scenarios. The dominant sources are from gas reserves at around 70% followed by 20% for oil reserves, and hard coal at 8%. All sources are represented in the same proportions regardless of the scenarios. An increased level of recycling reduces the use of energy, by 5% for gas at 35% vs. 80% recycling, and 6% for oil. 100% Incineration represents 95% less usage of oil reserves, this is explained by the credit to the system for the energy generated by incinerating the waste.

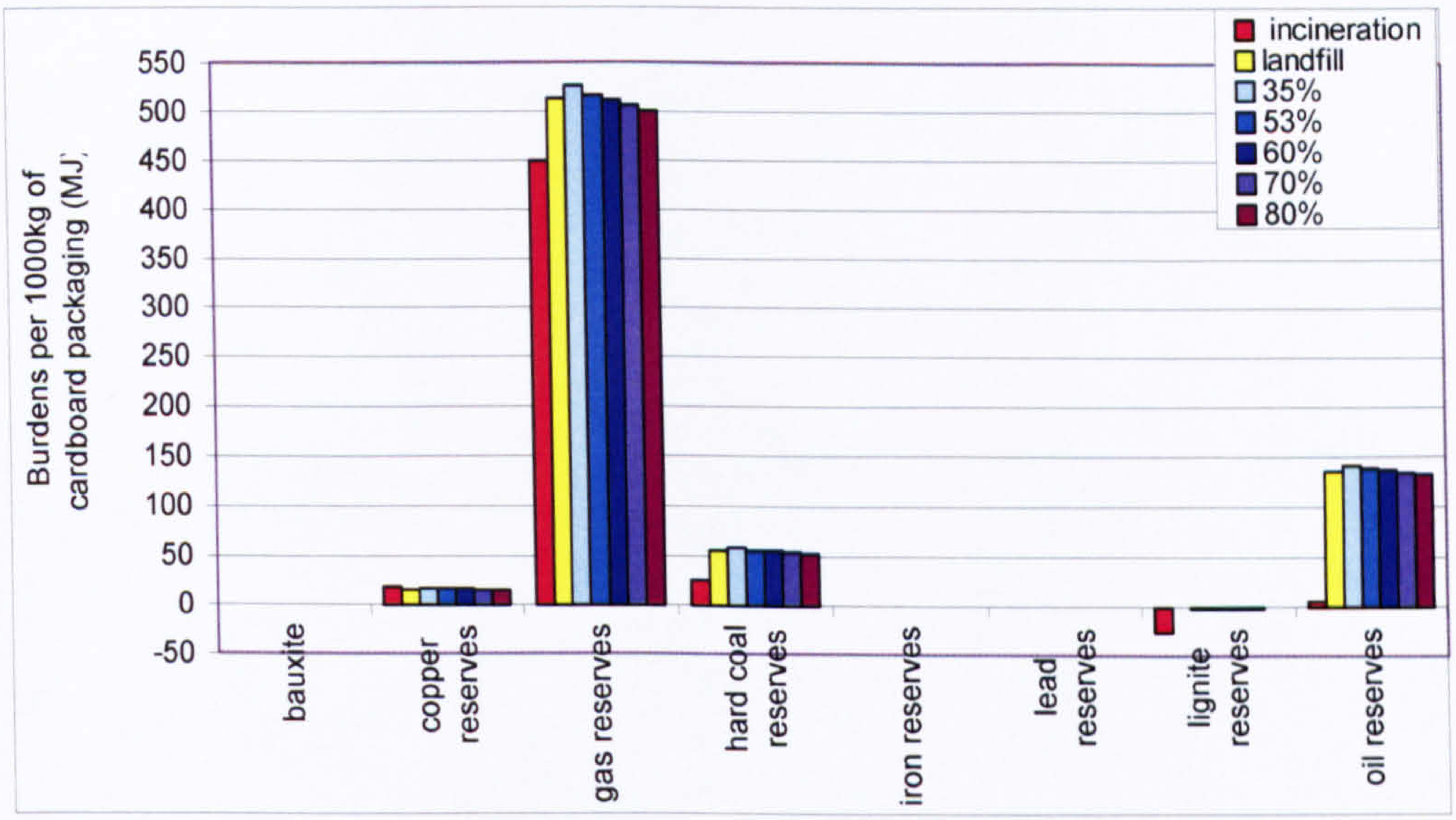

Figure 5.9: Energy resources (MJ)

5.2.3 System analysis

From this overview analysis, it is obvious that a higher recycling level is best in terms of lowering potential environmental impact. Yet the analysis so far does not give an insight into what part of the system is responsible for the majority of the burdens. Thus the results have also been considered in terms of the system's component 'blocks'. The system analysis is achieved by using the full analyses for each individual scenario, when compiled the results are produced for each individual modules included in the system. Because each transport requirement is entered individually as a module (the lorries in figure 4.1) and attached by flows to the relevant modules (see section 4.2.2); it is thus easy to allocate the correct transport data to each block. The same is valid for energy. The block are organised by adding together the results for all relevant modules for a block.

The systems are separated into three blocks:

- The input, cardboard manufacturing process and transport,
- Waste management for final disposal including transport, landfill and incineration process,
- Recycling that comprises transport from waste sorting to recycling facilities and recycling processes.

This exercise helps to understand where in the system the highest burdens occur and whether altering the level of recycling moves the burdens along the life cycle of packaging or not. Looking at Figure 5.10, it is obvious that there is not one answer but rather a mix of outcomes.

It has to be acknowledged that the incineration block is the only one benefiting from credits. This is true for four out of the seven studied impacts. This is due to the fact that the boundaries of the system account for the generation of energy during the incineration process (see section 4.2.2). In the 100% incineration scenario, the waste management block is responsible for 64% of the potential occurrence of human toxicity and 38% of the global warming burdens.

In the 100% landfill scenario, the waste management block dominates the contribution to global warming impact, hence being responsible for 56% of its occurrence; this was already observed in section 5.2.2.2. For all other categories, its contribution is marginal.

For three of the seven categories studied, noticeable changes take place with higher recycling. The recycling block becomes the highest contributor to these impacts once

80% recycling is reached, whereas at 35% the manufacturing block is the dominant one.

Looking first at global warming, it can be noted that at the 35% recycling level, the impact is dominated by the waste block (45%) and the manufacturing block (40%). However, once the 80% recycling level is reached, the recycling block becomes the dominant contributor (41%). Regarding eutrophication, at 80% recycling level the recycling block is overwhelmingly responsible for the potential impact at 65%, whereas at 35% recycling level it is the manufacturing block at 71%. Similarly for photo-oxidant formation, manufacturing is the dominant contributor (68%) at 35% recycling, but once 80% is achieved the recycling block is responsible for 60% of the overall impact.

The manufacturing block dominates the categories of acidification, ecotoxicity and energy, even at the higher recycling levels.

This brief analysis reinforces the findings that higher recycling translates into lower overall environmental impact, although with higher recycling levels there is a tendency for manufacturing-related impacts to decrease and for the recycling block to begin to dominate. As the manufacturing of cardboard material is the primary contributor to the environmental impacts, it might be relevant to concentrate on further improving the paper manufacturing technologies as well as increasing recycling levels. However at presents investments in new technology or in research and development for improving current technology are hindered by current legislation that favours one avenue rather than others (Radetzki, 1999).

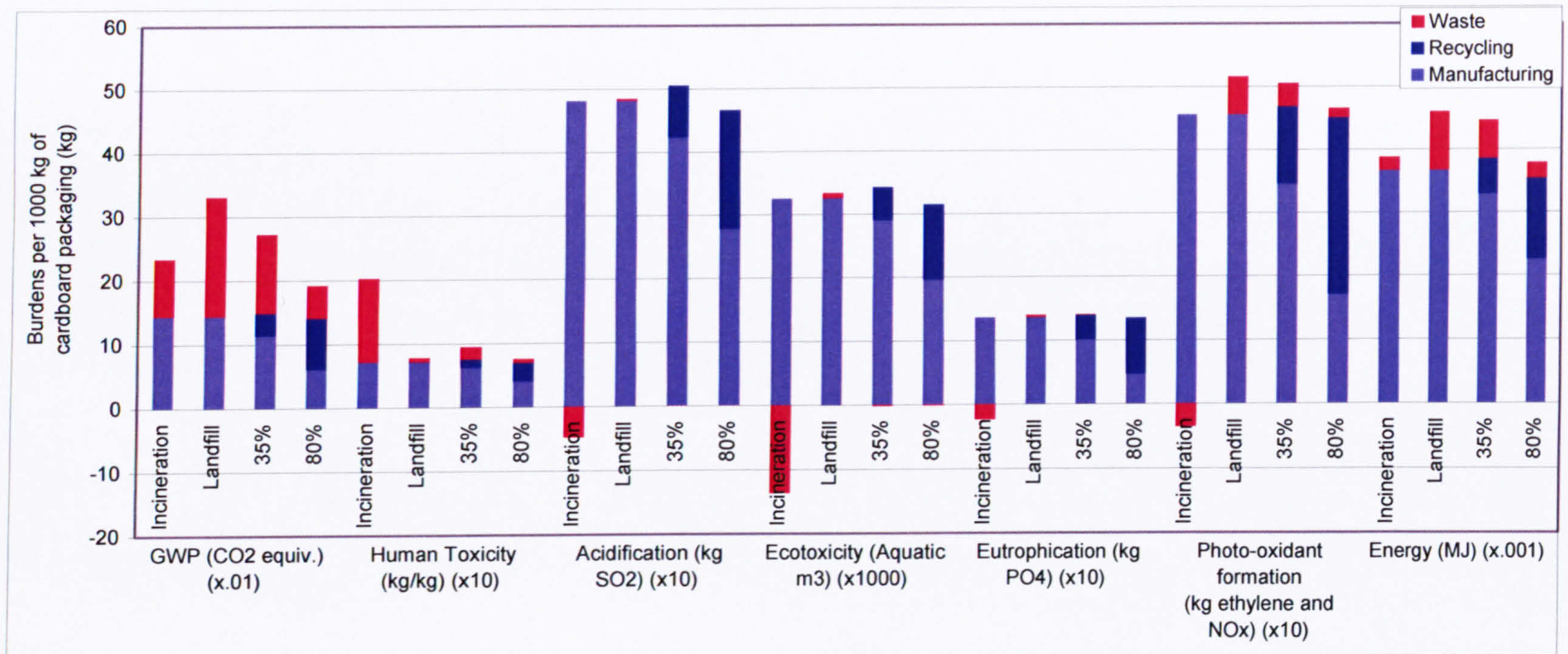

Figure 5.10: Blocks contribution to environmental impacts (in real terms)

5.3 Testing energy and transport

In this section the same scenarios as previously are used. First the energy requirements of the model are amended to include more renewable energy. This is to evaluate the impact newer recycling technology might have on the overall environmental impact of cardboard packaging. The second part look at transport distances of cardboard packaging waste and assumed that part of the separated cardboard waste is exported to Europe (see section 4.1).

5.3.1.1 Energy

The main parts of the system that require large amounts of energy are the manufacturing of cardboard, and the recycling of cardboard packaging waste. As seen in the previous figure (5.10) energy represents a large impact, and higher recycling only has a limited reduction effect, 20% less at 80% recycling compared to 35%. As this study concentrates on evaluating the impact of recycling targets, this part focuses on how more renewable energy usage would affect the total environmental impact of the system. Thus the results presented here are those of scenarios where 30% of energy requirement for the recycling process comes from clean energy (hydro energy).

5.3.1.1.1 Inventory results

Figure 5.11 illustrates a summary report of the inventory for the clean energy scenarios (which are the ones denoted (E)) against the standard scenarios. The use of clean energy for 30% of energy requirements within the recycling process results in reducing the total burdens to the environment for all emissions. However, a few are unchanged, including VOC-halogenated, heavy metals (A) airborne and toxic substances.

The largest benefits in emissions from using clean energy are in the reduction of gases (A) (CO_2 emissions), several types of VOC emissions and organic (B) waterborne, which see their levels reducing further with higher recycling levels. Thus, CO_2 releases fall by 5% at 35% (E) compared to 35%, and by 14% at 80% recycling. VOC (aromatics) benefits from 13% at 35% vs. 35% (E) and 35% at 80% vs. 80% (E). Organic (water) B releases fall by 11% from 35% to 35% (E) and by 29% at 80% to 80% (E).

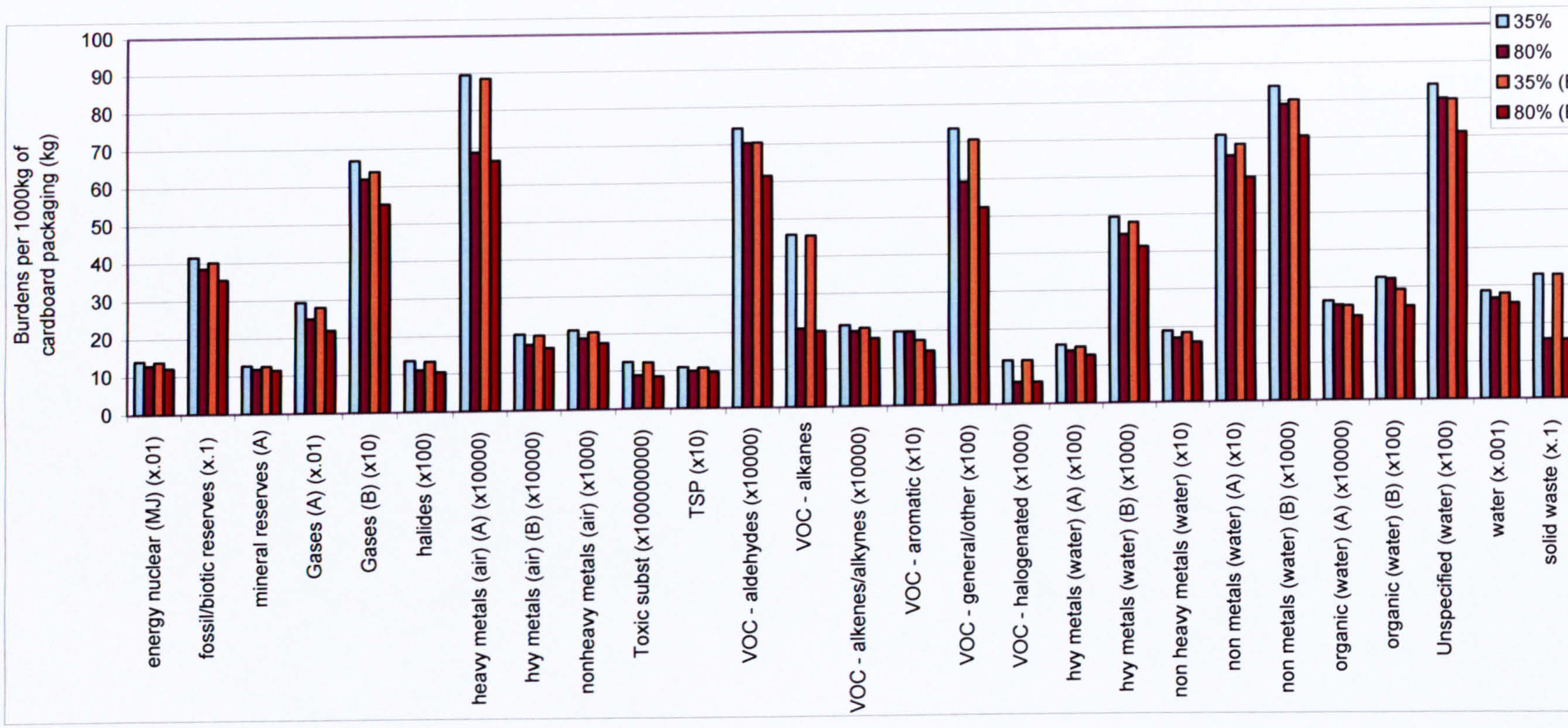

Figure 5.11: Inventory summary

5.3.1.1.2 Global warming

Figure 5.12 illustrates that CO_2 emissions benefit from the greatest overall reduction compared to standard scenarios, with the partial use of clean energy. Indeed the CO_2 emissions drop by 8% when comparing 53% to 53% (E) and by 14.8% at 80% to 80% (E). Methane emissions are almost unaffected this is due to the fact that final waste disposal to landfill is the largest contributor to methane emissions (see section 5.3.1.2.). The overall impact is reduced by 20% at 53% vs. 80% and by 25% from 53% (E) to 80% (E).

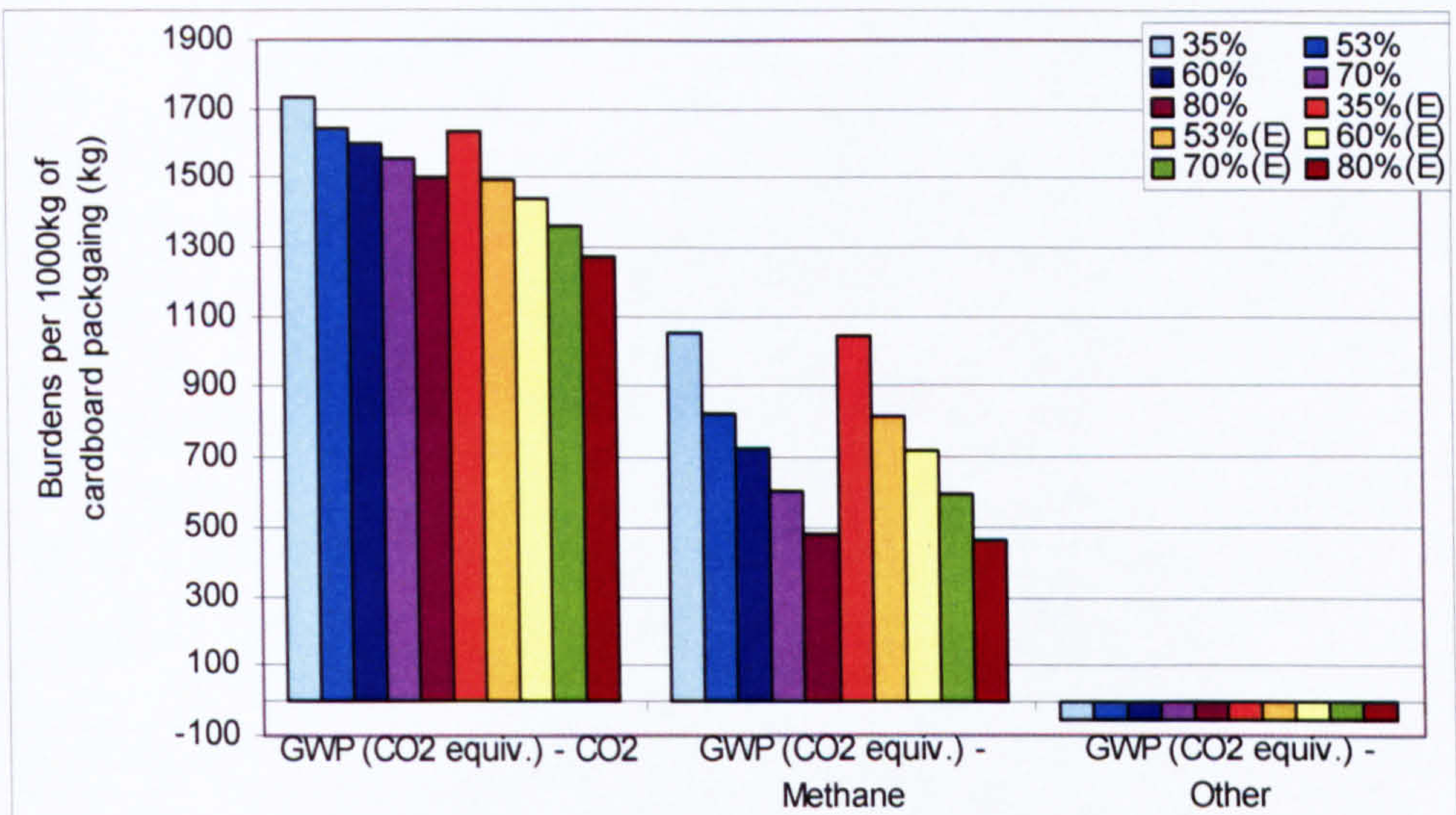

Figure 5.12: Global warming (kg CO_2 equivalent)

5.3.1.1.3 Human toxicity

Figure 5.13 shows that all emissions contributing to human toxicity are reduced by the use of clean energy, apart from As and Pb which remain constant. The contributions of two burdens to the impact are greatly reduced, aromatics benefit from a 17.5% reduction from 53% to 53% (E), reaching 26% at 80% vs. 80% (E), and metals emissions are lowered by 14.5% when comparing the 53% scenarios and by 22% at 80% levels.

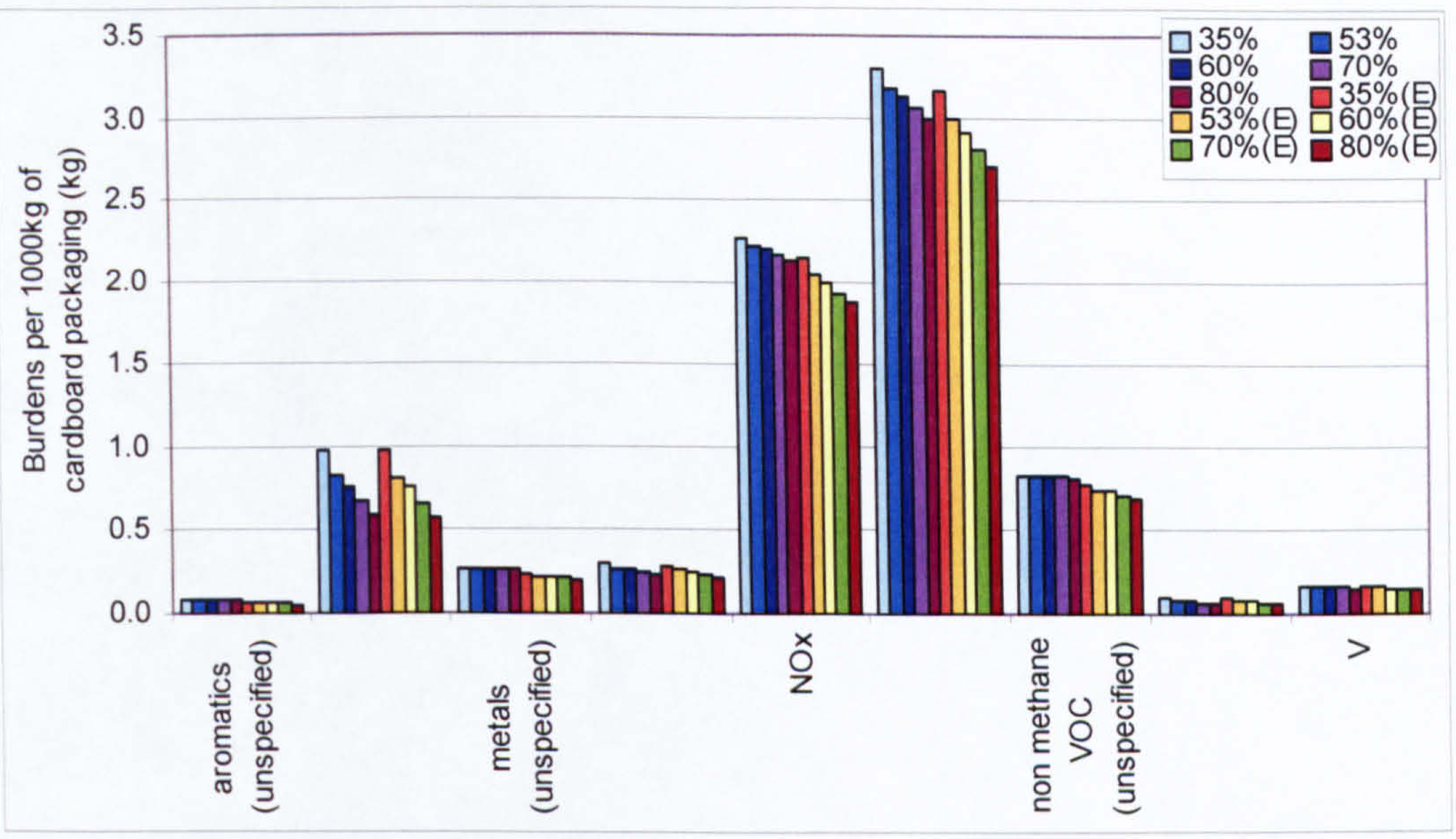

Figure 5.13: Human toxicity

5.3.1.1.4 Acidification

As can be seen in Figure 5.14, NOx and SO_2 are the largest contributors to acidification and are the ones mostly benefiting from the use of clean energy. Thus, NOx emissions are lowered by 3.5% at 53% vs. 80%, by 8.2% at 53% (E) vs. 80% (E)%. SO_2 emissions are reduced by 5.6% and 9% for the same scenarios.

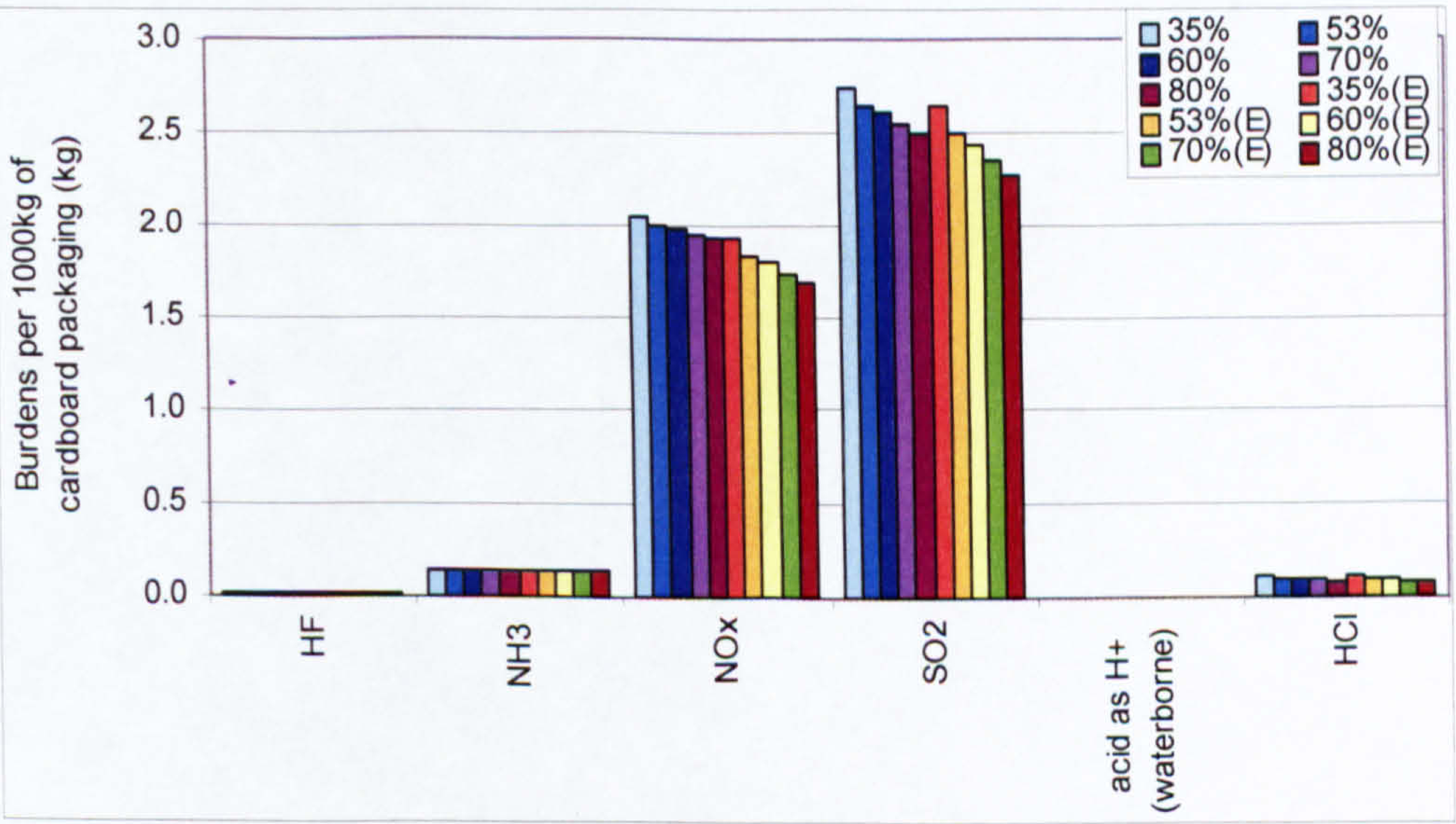

Figure 5.14: Acidification (kg SO_2 equivalent)

5.3.1.1.5 Ecotoxicity

The largest benefit that can be seen in Figure 5.15 is for mercury (Hg) emissions, which are reduced by 4% between 53% and 80% and reach 22.6% between 53% (E) to 80% (E). All other emissions are lowered on average by around 10%.

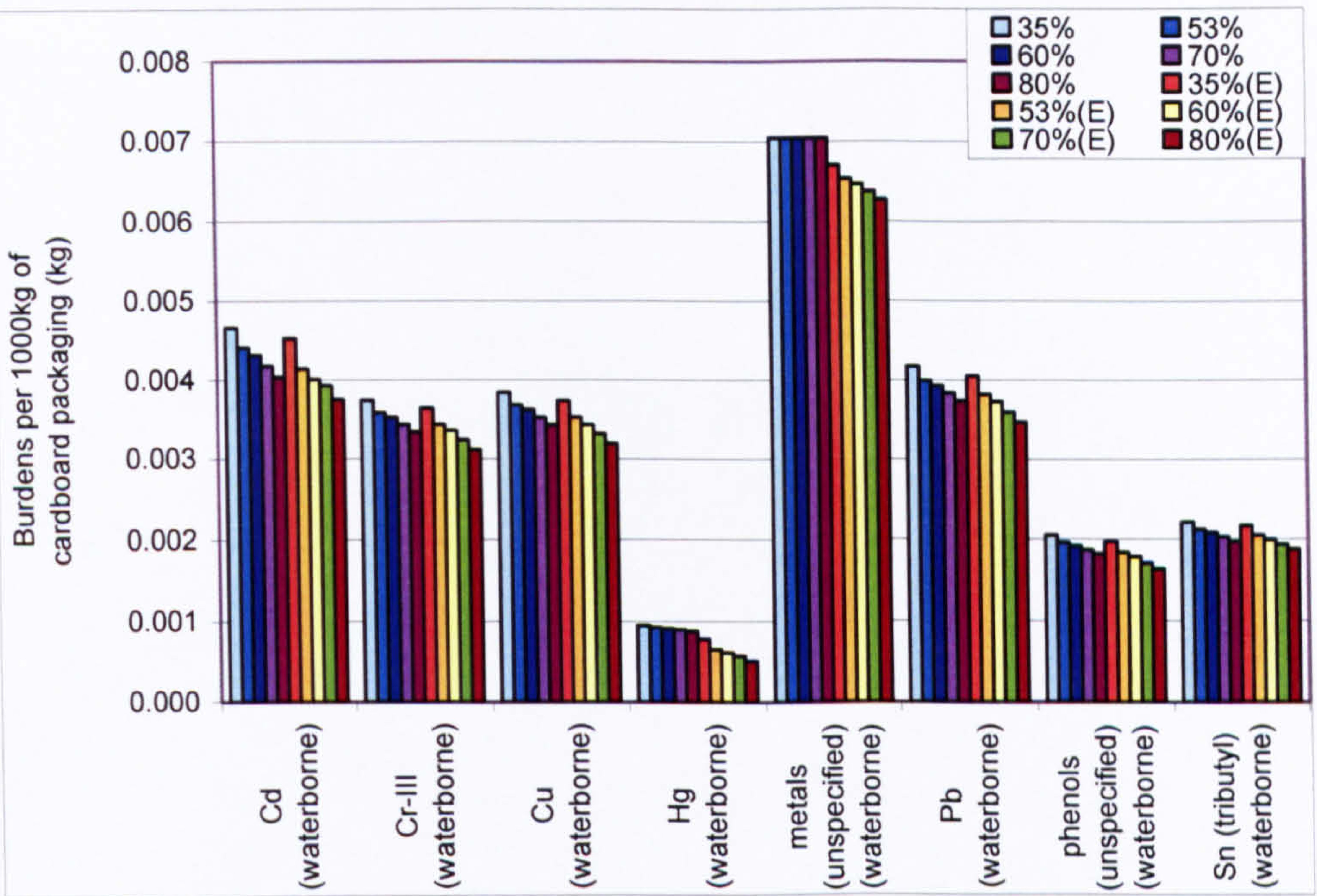

Figure 5.15: Ecotoxicity

5.3.1.1.6 Eutrophication

The emissions that are mostly reduced through the use of clean energy are phosphates and nitrogenous compounds, whereas both were constant in standard scenarios. Phosphates are reduced by 10.7% (Figure 5.16), and nitrogenous compounds by 7% from 53% (E) to 80% (E) (Figure 5.17). When looking specifically at 80% vs. 80% (E), Nitrogenous compounds are lowered by 19% and phosphates by 26%.

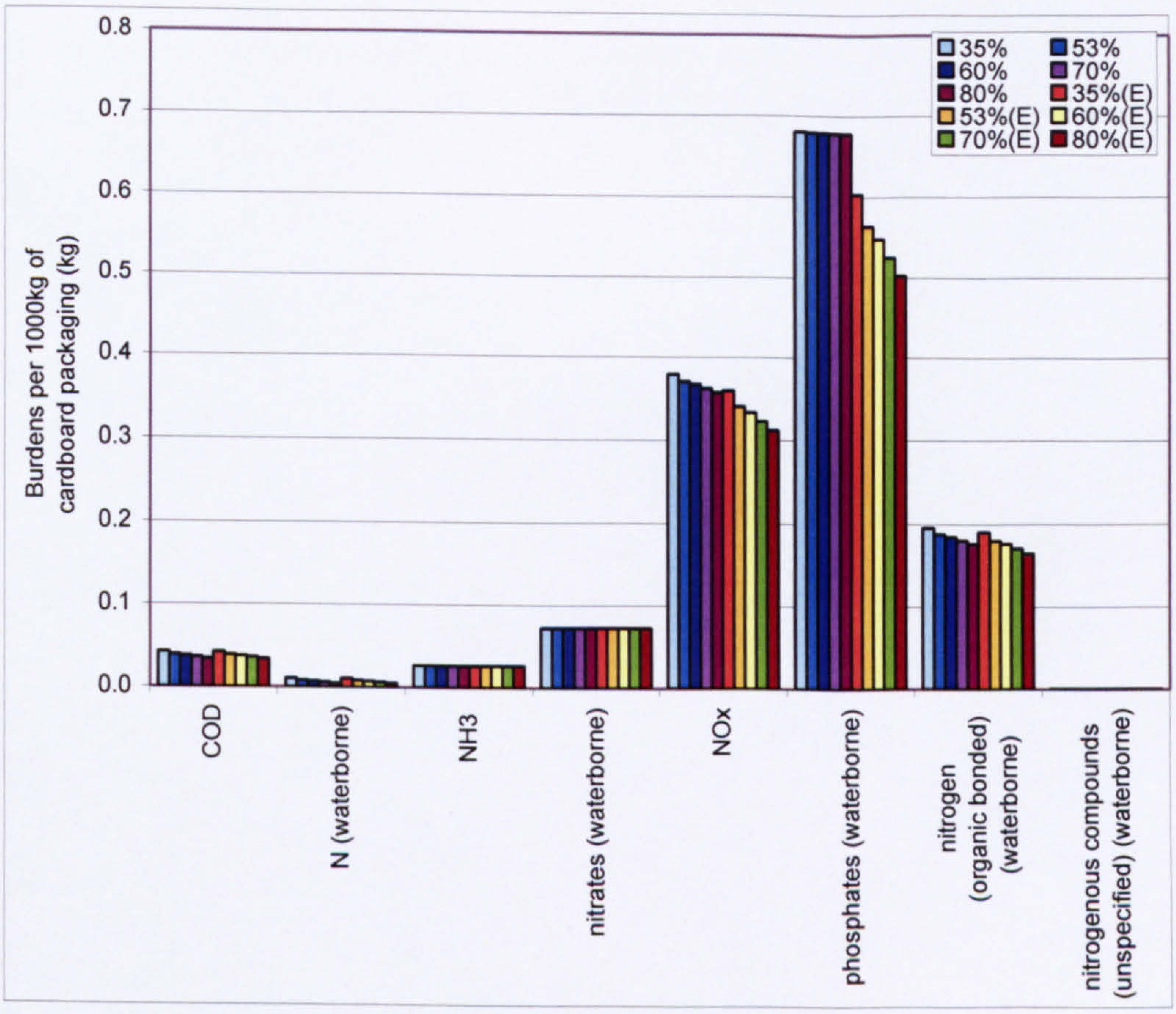

Figure 5.16: Eutrophication (kg PO_4)

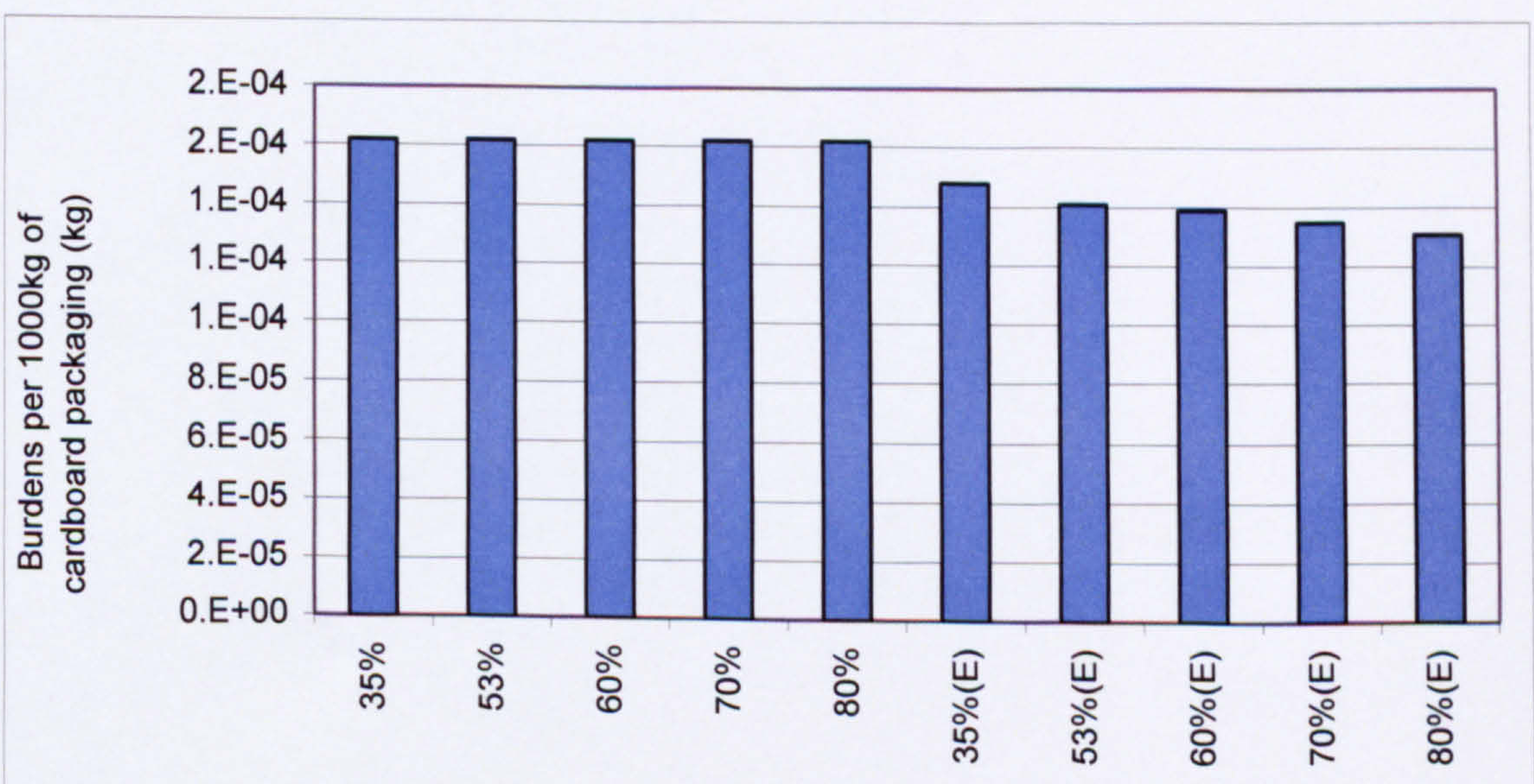

Figure 5.17: Nitrogenous compounds in Eutrophication impact

5.3.1.1.7 Photo-oxidant formation

Most of the emissions contributing to photo-oxidant formation are reduced by the use of clean energy as can be seen in Figure 5.18. However, whilst HC and methane benefit from great reductions in standard scenarios, these are not affected by using clean energy. The largest improvement is seen in aromatics emissions that plummet by 17.5% at 53% (E) vs. 53% and reach 26% at 80% (E) vs. 80%, whereas they were little affected in standard scenarios. All other emissions are reduced in the order of 8% at 80% (E) compared to 53% (E).

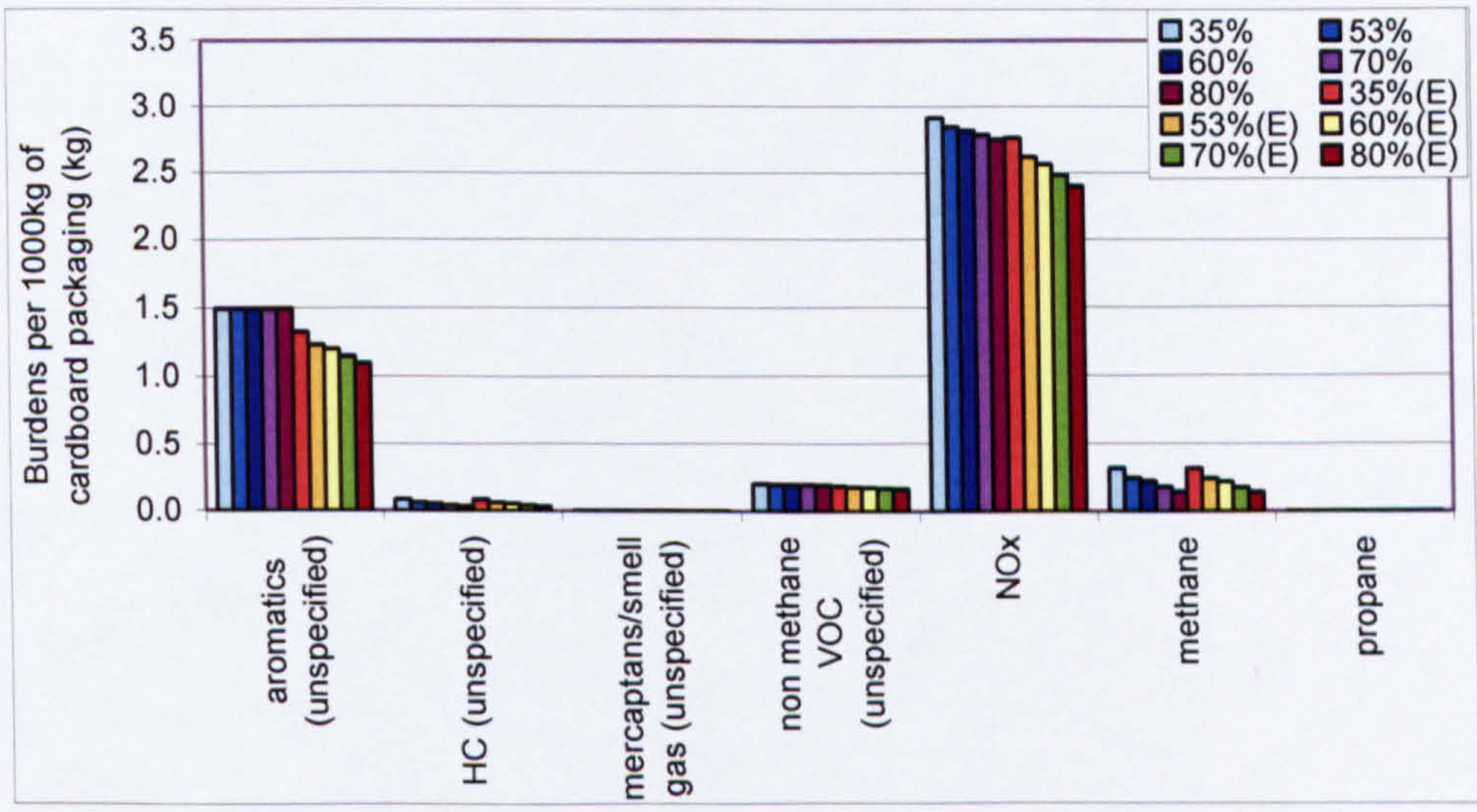

Figure 5.18: Photo-oxidant formation

5.3.1.1.8 Energy resources

The energy resource most minimised by using a cleaner one is gas, which represents the largest input in the standard scenarios. Figure 5.19 shows that gas input is reduced by 9% from 53% to 53%(E) and by 15% at 80% (E) vs. 80%. Most other resources see a reduction in the order of 2.5 to 6%.

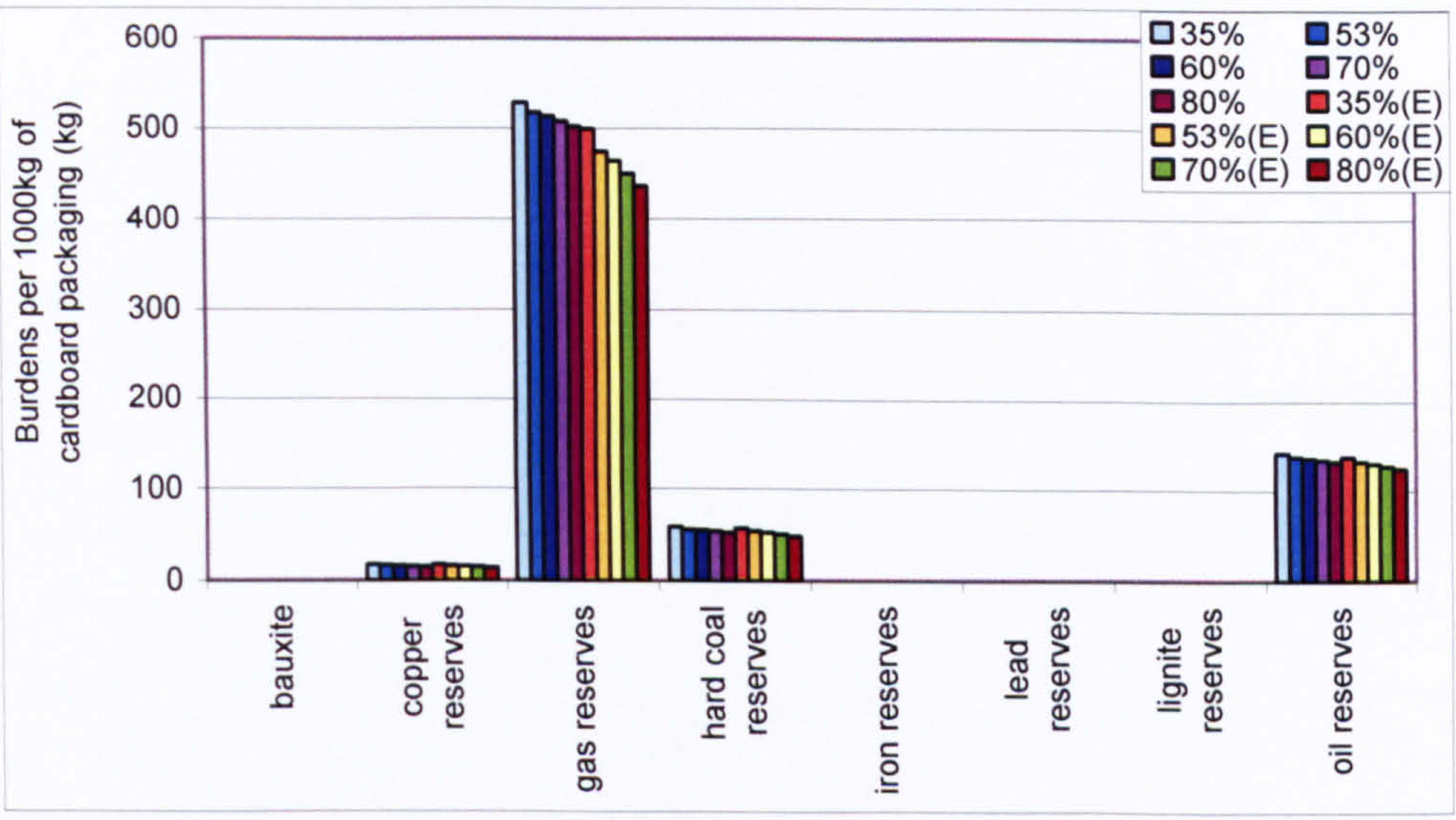

Figure 5.19: Energy resources (MJ)

5.3.1.2 Transport

Transport is not the largest contributor to the models' environmental impacts. However, it was deemed relevant to model different transport distances between the waste sorting and recycling, and how they might affect the models, as this is a very likely scenario to happen.

The main stage where transport distances might be changing is from waste collection to final disposal and recycling facilities. The results presented below correspond to the following assumption - it is known that the current UK recycling facilities for paper and board have reached their full capacity, and that there is more waste paper on the market than that required by the paper and board industry. Thus, a growing fraction of waste paper including cardboard packaging waste is being exported, mainly to Europe but also to Asia. Here we concentrate on Europe as the destination. It is assumed that for levels of recycling above 55% the extra cardboard packaging waste is being exported to Europe by boat, across the channel and then by trucks, mainly on highways and with only 5% of journeys being urban. Averages distances are being used; it was assumed that the waste departed from within central England some 250km away from Dover. The distances for Europe are 800 km assuming it is going to France or Germany.

The following results compare the environmental burdens of scenarios for 60, 70 and 80% recycling, for the standard assumptions against new transport distances for waste recycling. These are the scenarios initialled (T).

5.3.1.2.1 Inventory results

Figure 5.20 represents the life cycle inventory summary and highlights that by altering transport distances, most environmental burdens increase. The data presented here, again, are scaled so that comparisons can be made through the figures.

For eight of the burdens that showed decreases in response to increases in the level of recycling under the standard scenarios, the reverse is now evident. Mineral reserves and toxic substances are the ones with the most significant increases; usage of mineral reserves increases by 25% when the level of recycling increases from 60% (T) to 80% (T), while it reduces by 3% in the standard scenarios, and emissions of toxic substances increase by 17.7% against a 15% reduction in the standard scenarios.

On the other hand, many burdens still benefit from an overall decrease in emissions with increased recycling, though the scale of the reduction is lower than in the standard scenarios (see table 5.1).

A few burdens, including heavy metals (A) airborne, VOC aldehydes and halogenated, are unchanged by higher transport distances.

	60% - 80%	60 % (T) – 80% (T)
Gases (A)	- 7%	- 5.5 %
TSP	- 4%	- 1%
Non-metals (water) (A)	- 3.6%	- 1%

Table 5.1: Burdens emissions level differences

Figure 5.20 is an overview of emissions level, but there is a need to understand how these increases or decreases in emission levels might affect their contribution to potential environmental impacts.

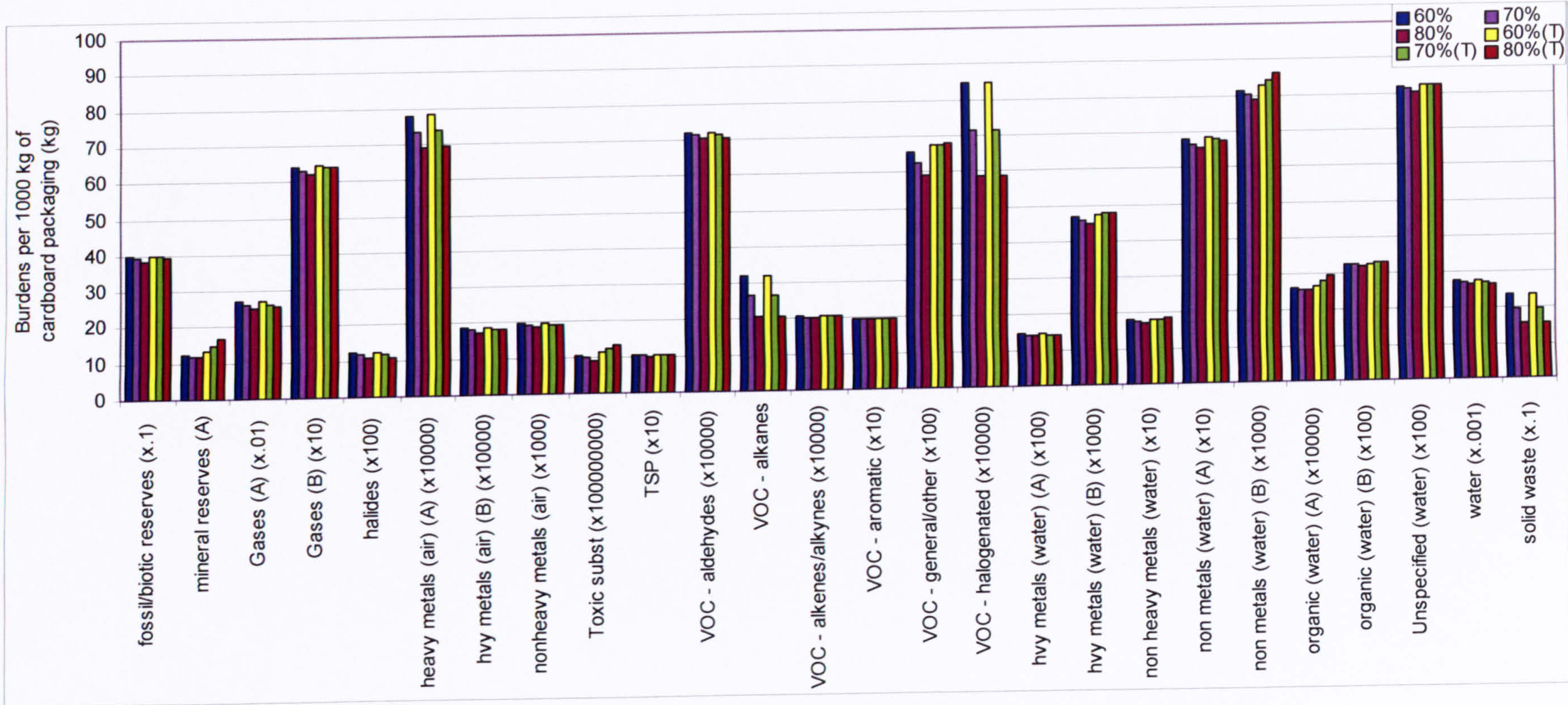

Figure 5.20: Summary inventory report

5.3.1.2.2 Global warming

Global warming impact falls in response to increases in the level of recycling, even when transport distances are increased. CO_2 emissions are reduced by 5 % and methane by 34.7% when recycling goes from 60% (T) to 80% (T) (see Figure 5.21). The reduction is greater for the standard scenarios, with an overall fall of 6.5% from 60% to 80% recycling. The extra transport has no impact on methane emissions.

Total emissions rise slightly when comparing standard scenarios to transport scenarios. Thus at 80% (T) the total value of CO_2 emissions is 1.4% higher than at 80%.

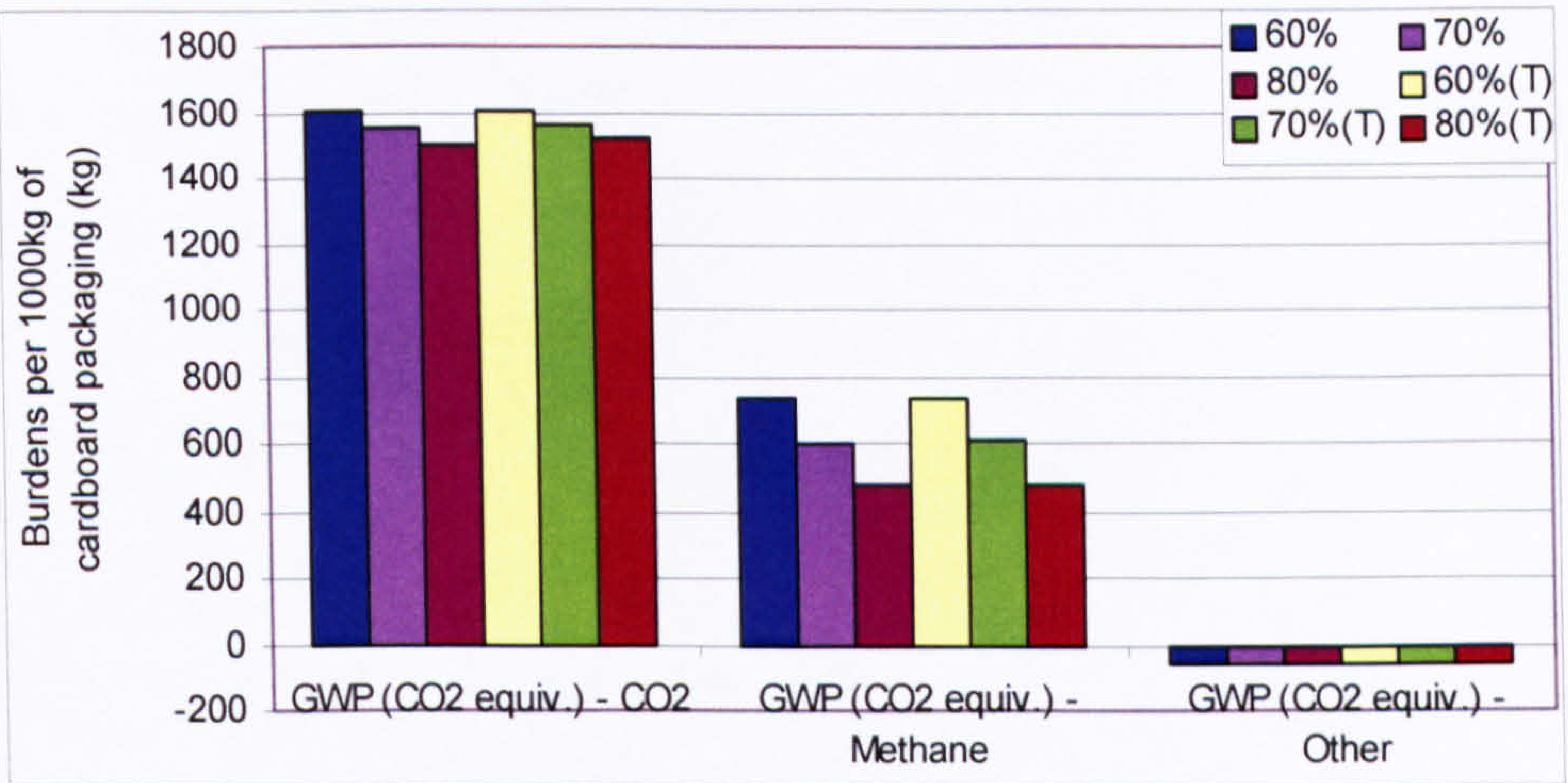

Figure 5.21: Global warming (kg CO_2 equivalent)

5.3.1.2.3 Human toxicity

Figure 5.22 presents human toxicity impact; the first comment is that non-methane VOC and NOx emissions rise as the recycling level increases in the transport scenarios whereas they reduced in response to increased recycling in the case of standard scenarios. Thus, non-methane VOC benefits from a 1% drop from 60% to 80%, but goes up by 12.5% from 60% (T) to 80% (T). For NOx, this amounts to a reduction of 2.6% in the standard scenario compared to an increase of 0.5% with extra transport. Metals and As are unaffected by higher transport distances.

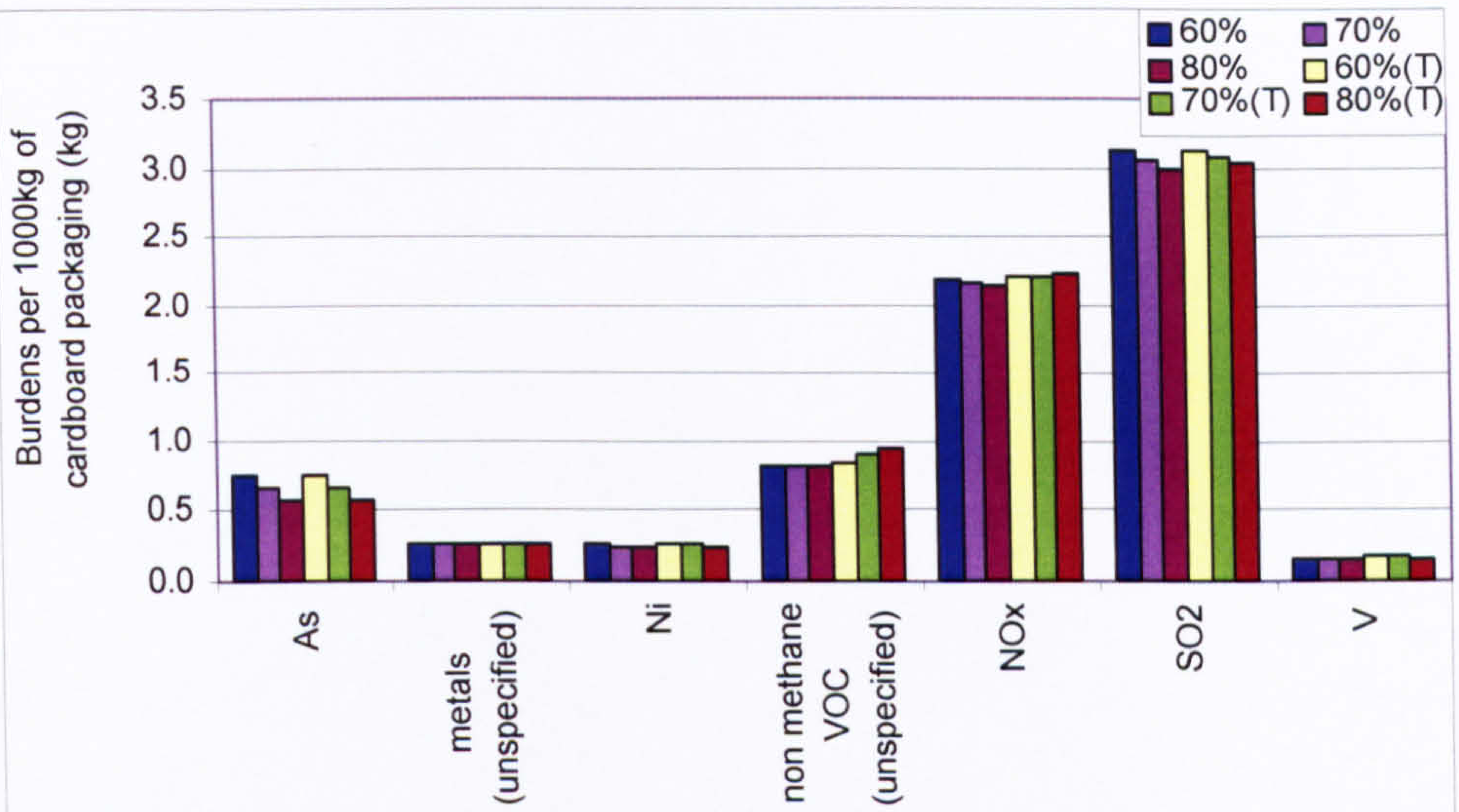

Figure 5.22: Human Toxicity

5.3.1.2.4 Acidification

Regarding the potential environmental impact of acidification, Figure 5.24 highlights that only NOx and acid emissions (see Figure 5.23) rise with more transport. The most dramatic increase is for acid emissions, which escalate by 67% from 60% (T) to 80% (T) recycling level, compared to an equivalent rise of just 11.7% in the standard scenarios. However, the contribution of acid emission to the overall acidification impact is very small. NOx sees an increase of 4% at 80% (T) compared to 80%, which amounts to an increase of 0.5% from 60% (T) to 80% (T), compared to a reduction of 2.6% for the standard scenarios at the same recycling levels.

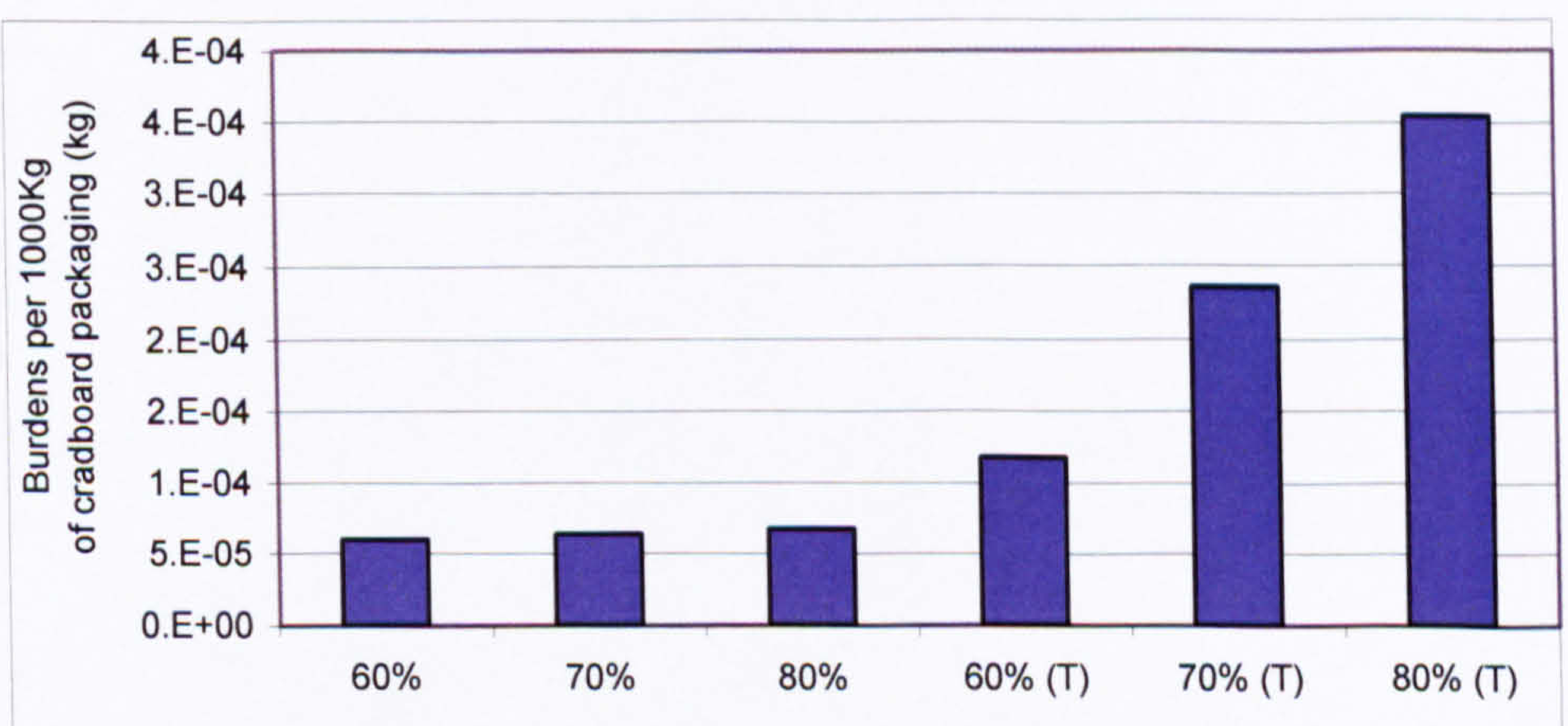

Figure 5.23: Acid emissions (H+) in acidification

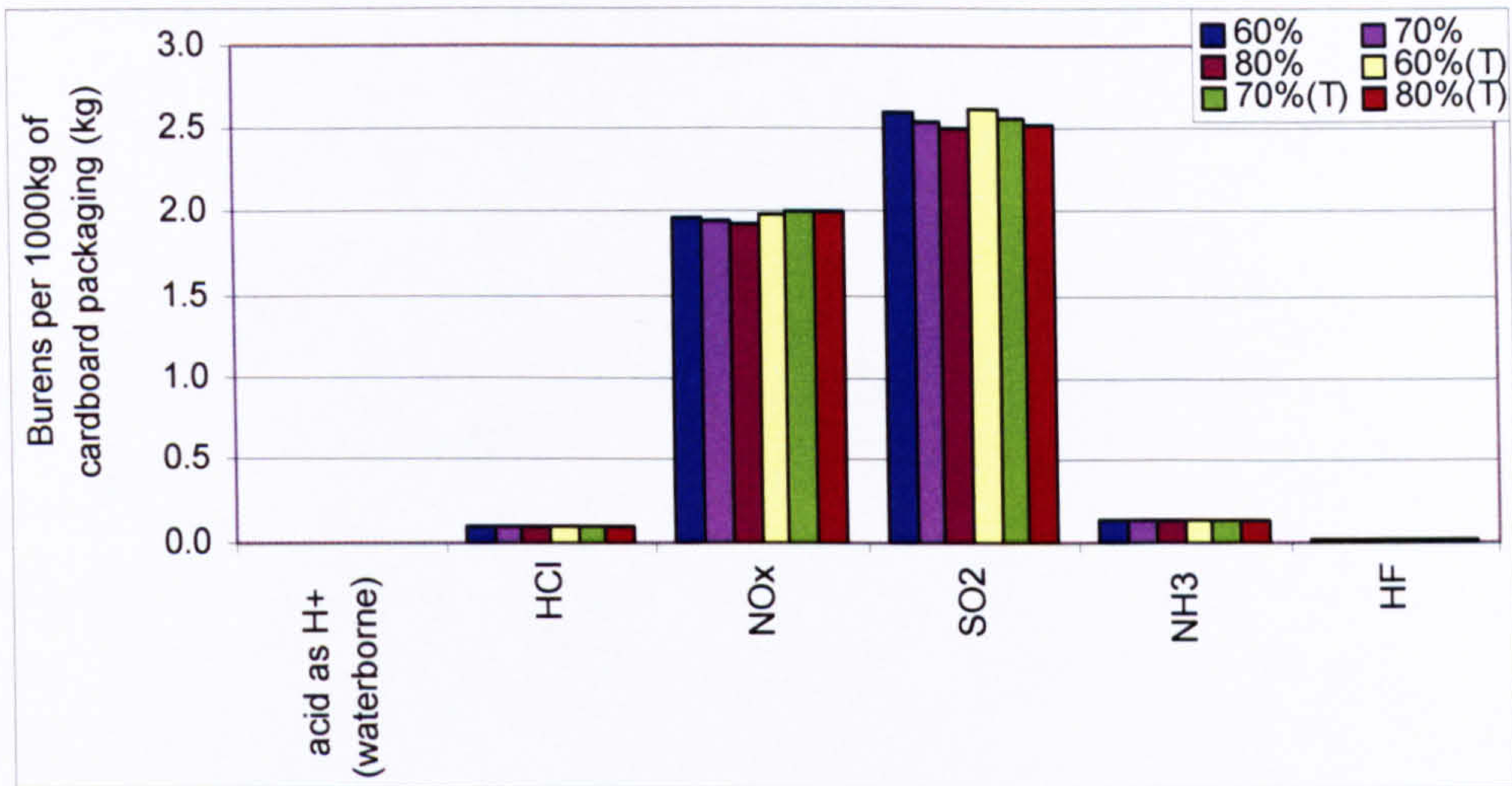

Figure 5.24: Acidification (kg SO_2 equivalent)

5.3.1.2.5 Ecotoxicity

Longer transport distances contribute to an increase in toxicity impact on the environment; phenols (waterborne) and Sn emissions are the only ones that increase with more transport and more recycling (Figure 5.25). Phenols increase by 3% at 60% vs. 60% (T) and by 13.4% at 80% vs. 80% (T); the overall increase is 5.8%, whereas in the standard scenarios the levels of phenols are reduced by 5.5% at 80% recycling compared to 60% recycling. Sn emissions are overall increased by 1% in the transport scenarios against a 5% reduction in the standard scenarios. All the other emissions benefit from an overall decrease of emissions with further recycling but to a lesser extent than in the standard scenarios.

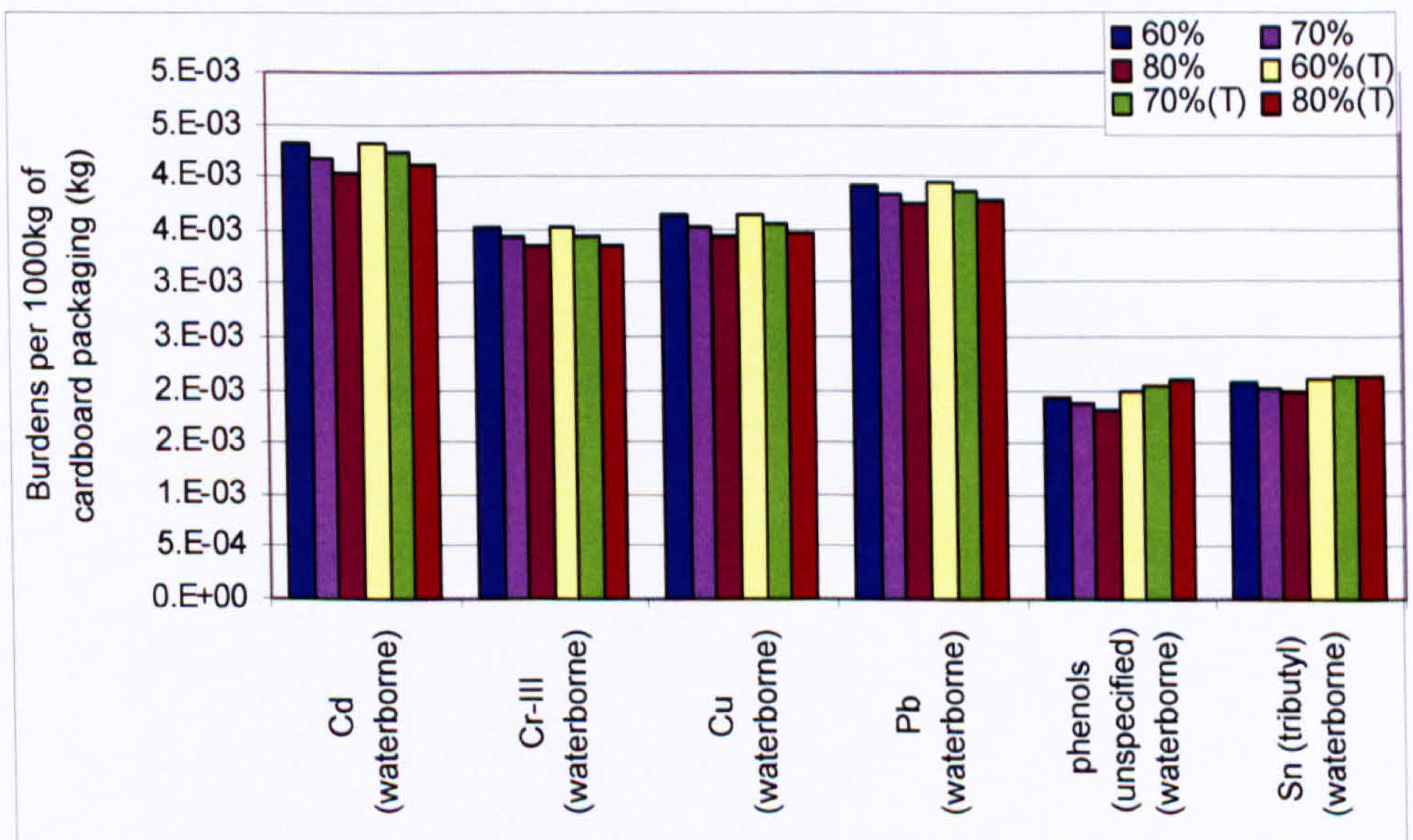

Figure 5.25: Ecotoxicity

5.3.1.2.6 Eutrophication

The burdens contributing to eutrophication impact behave similarly in standard and transport scenarios, as can be seen in Figure 5.26. Only NOx outputs are increased with more transport, by 0.5% between 60% (T) and 80% (T) against a reduction of 2.6% for standard scenarios.

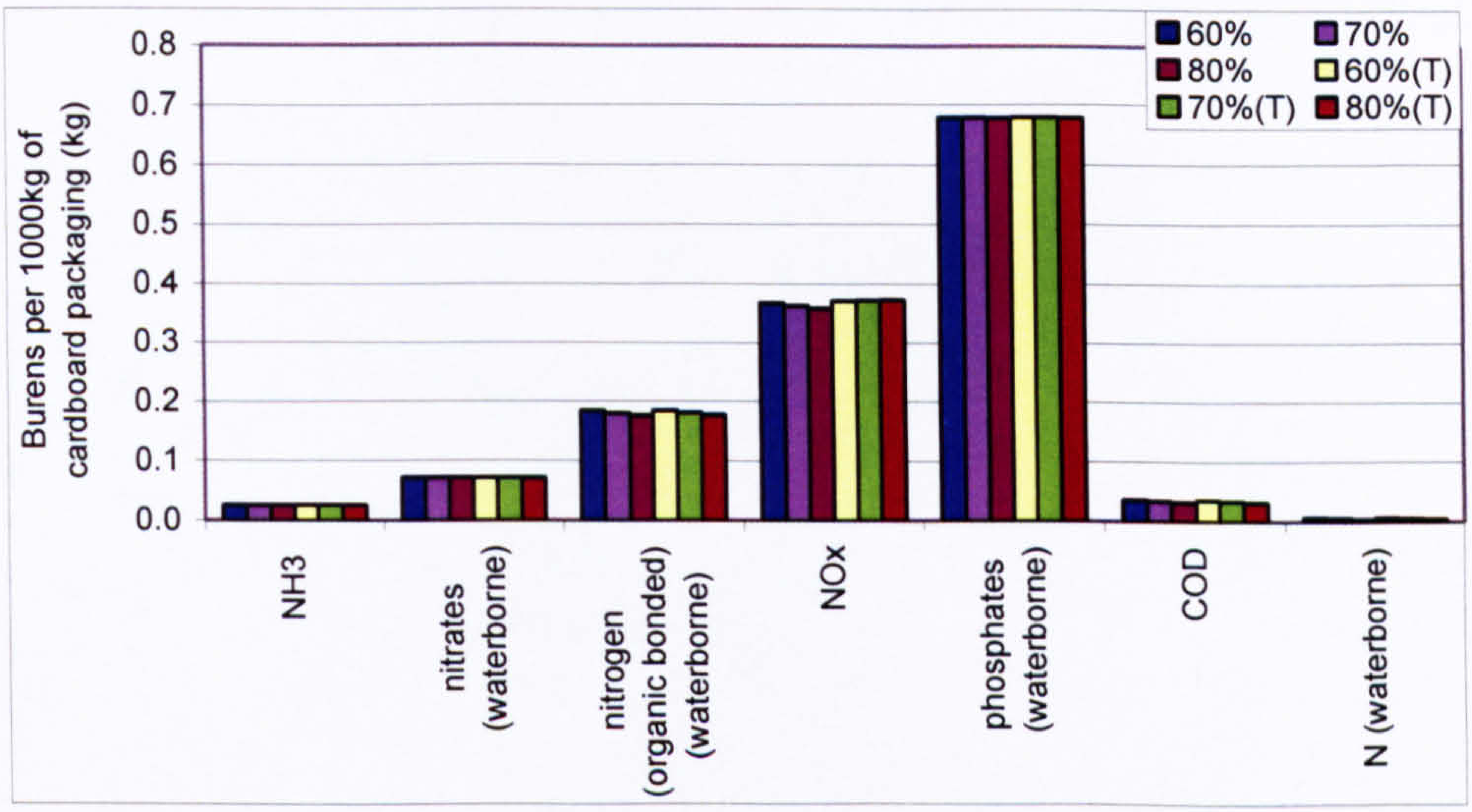

Figure 5.26:Eutrophication (kg PO_4)

5.3.1.2.7 Photo-oxidant formation

It can be seen in Figure 5.27 that extra transport impacts much more on non-methane VOC emissions (+12.5% from 60% (T) to 80% (T) compared to -1% in standard scenarios) than on any other burdens. The only other two emissions affected are NOx and propane but to minimal level.

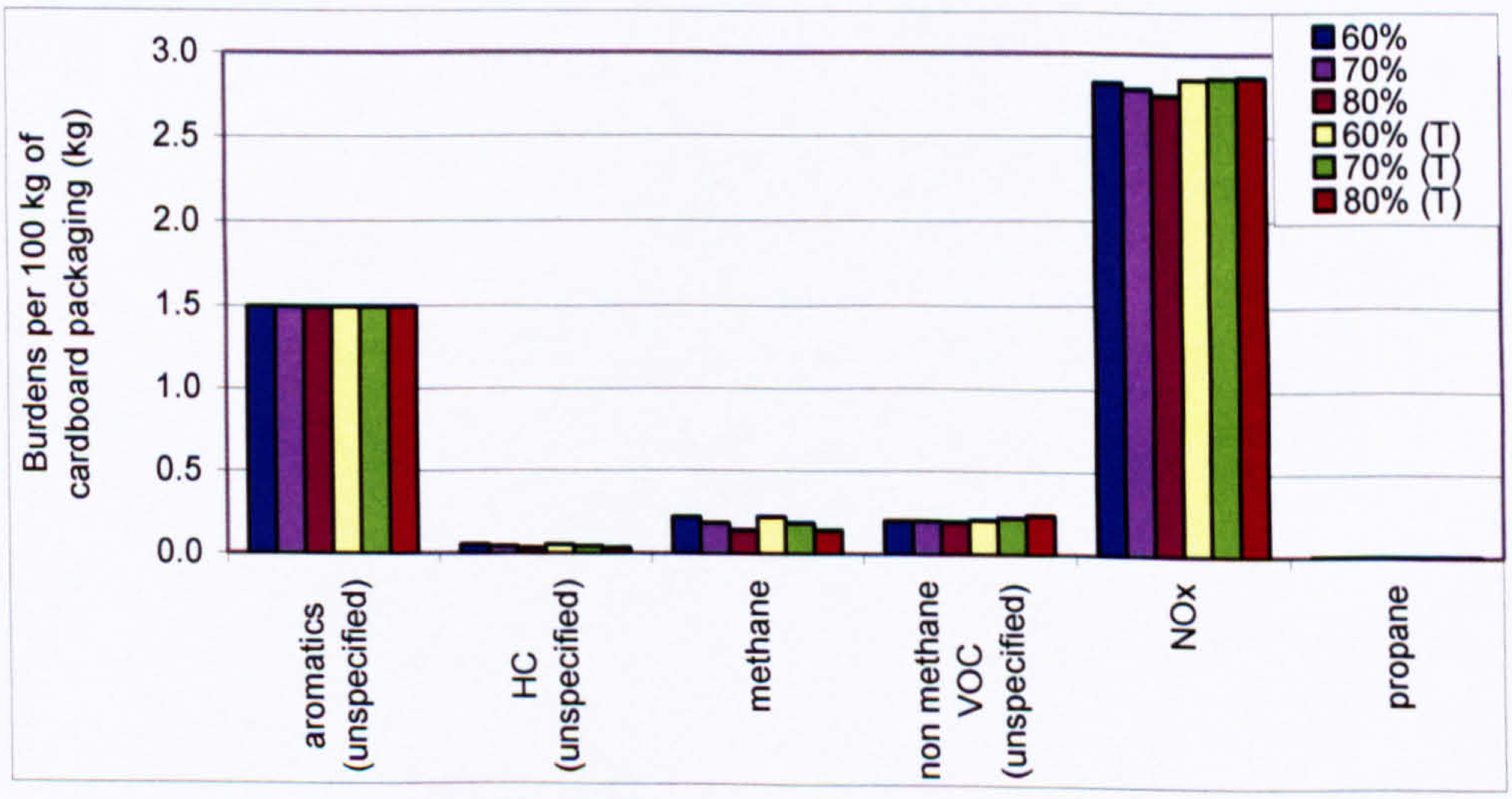

Figure 5.27: Photo-oxidant formation

5.3.1.2.8 Energy Resources

Figure 5.28 presents the environmental impact on resources. The overall energy requirements are increased in correlation with recycling levels by 1.2% at 60% compared to 60% (T) and by 6.5% at 80% vs. 80% (T). It can be noted that transport's main impact is on oil reserves, which increases by 20.5% from 60% (T) to 80% (T). This is a noticeable difference as in the previous models, there was a small drop of 3% for the same levels of recycling. The overall burdens from oil reserves escalate by 31.7% between 80% and 80% (T). Regarding the other types of resources it is noticed that they behave as in the standard scenarios.

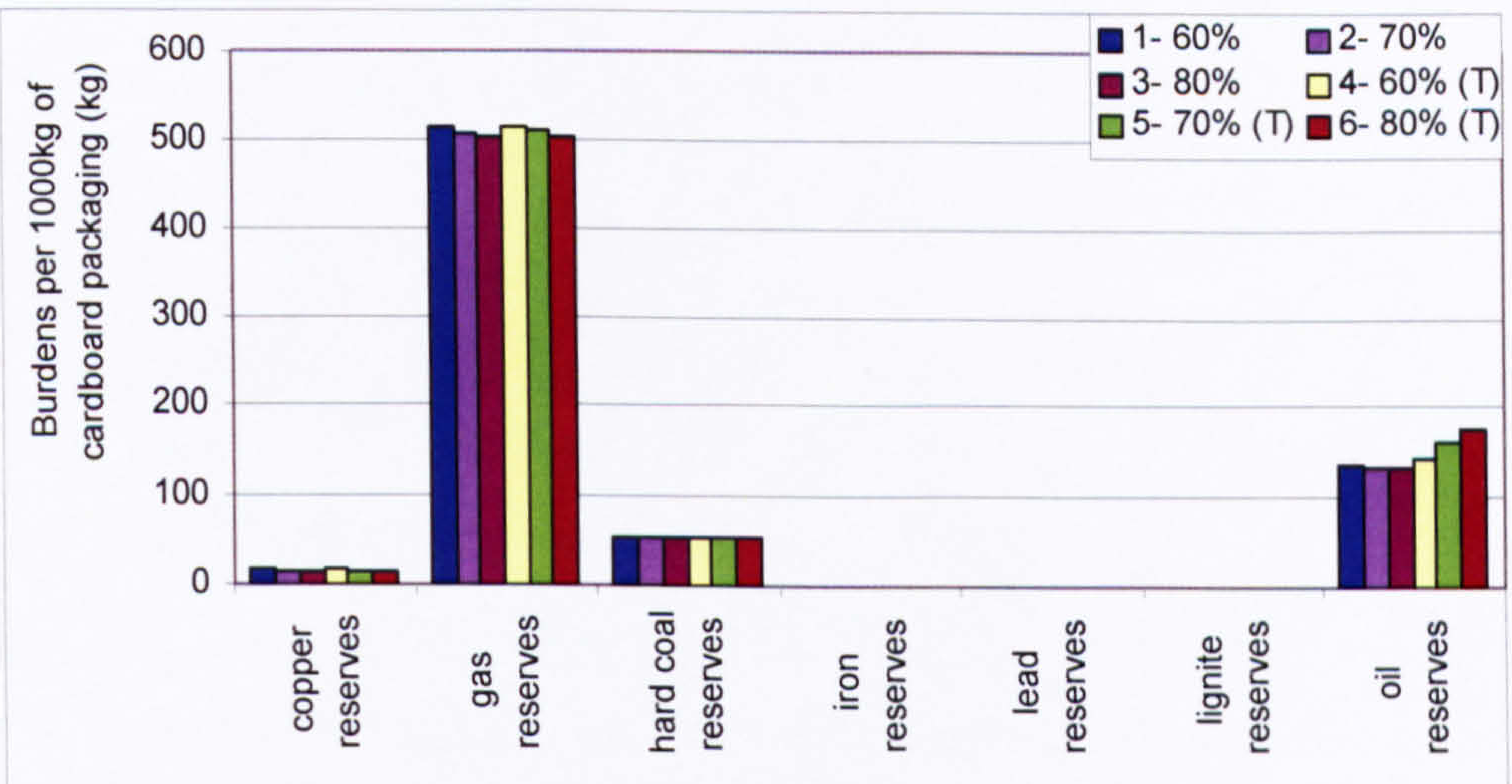

Figure 5.28: Energy resources (MJ)

5.4 Characterisation conclusion

The LCA analysis enables the environmental burdens of the various target options to be evaluated in terms of specific impact categories. The following seven impact categories were identified as being most relevant in the current UK political context: global warming, human toxicity, acidification, ecotoxicity, eutrophication, photo-oxidant formation and energy use. Figure 5.29 presents a summary comparison of the potential impacts associated with the range of recycling scenarios considered, normalised to the present situation (i.e. 53% recycling). Figure 5.30 is also a summary illustration but in real terms. This allows those categories that are dominant in terms of quantities to be identified.

By weight, the dominant impacts of the scenario are energy requirements and global warming, though this is not an indication of the severity of their potential damages.

Raising the level of paper packaging waste recycling, has the highest benefits for global warming, When recycling is increased from 53% to 80%, the global warming potential impact is reduced by 20%, energy by 10% and photo-oxidant formation by 8%.

Some specific burdens are dramatically improved by increasing the level of recycling. At 80% compared to 53% VOC alkanes are reduced by 42%, VOC halogenated by 38%, solid waste by 39% and toxic substances by 19%. All the remaining potential environmental burdens studied are more moderately affected by higher recycling.

From the characterisation results, incineration appears as a better option for several environmental categories. Thus it has considerably lower contribution to ecotoxicity and eutrophication, and more moderately for acidification and photo-oxidant formation than at 80% recycling. This is explained by the credit to the system of energy recovered during the incineration process. However, incineration is worse than high levels of recycling for global warming and overwhelmingly dominates human toxicity impacts, with levels more than double those exhibited by any other scenario. Incineration is worse for human toxicity mainly because of its arsenic and nickel emissions (see section 5.2.2.3).

Landfill is the worst option for global warming, and photo-oxidant formation. Indeed landfill contribute to 26% of UK's overall methane emissions (see section 5.2.2.2) which the most potent greenhouse gases, methane is also a key player in photo-oxidants. For all other considered impacts it is worse than high levels of recycling.

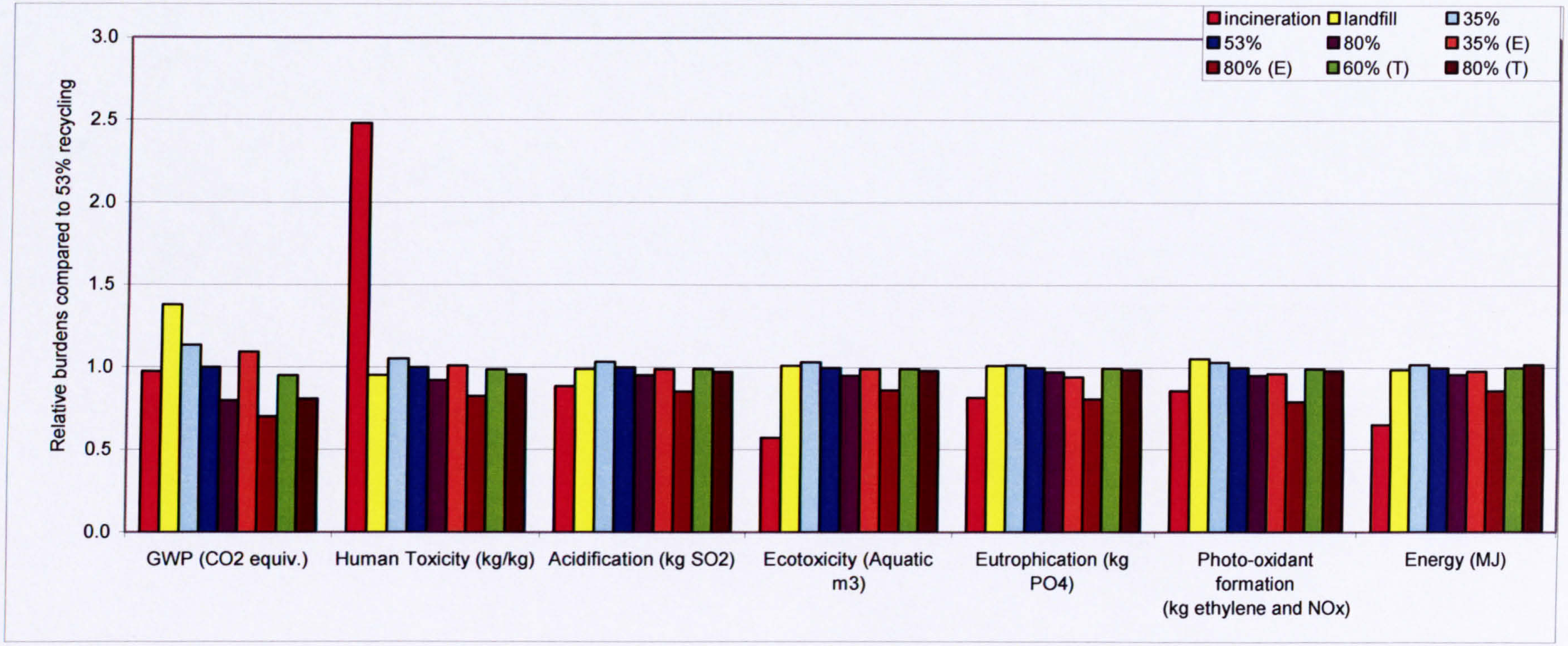

Figure 5.29: Summary of characterisation (relative burdens compared with 53% recycling)

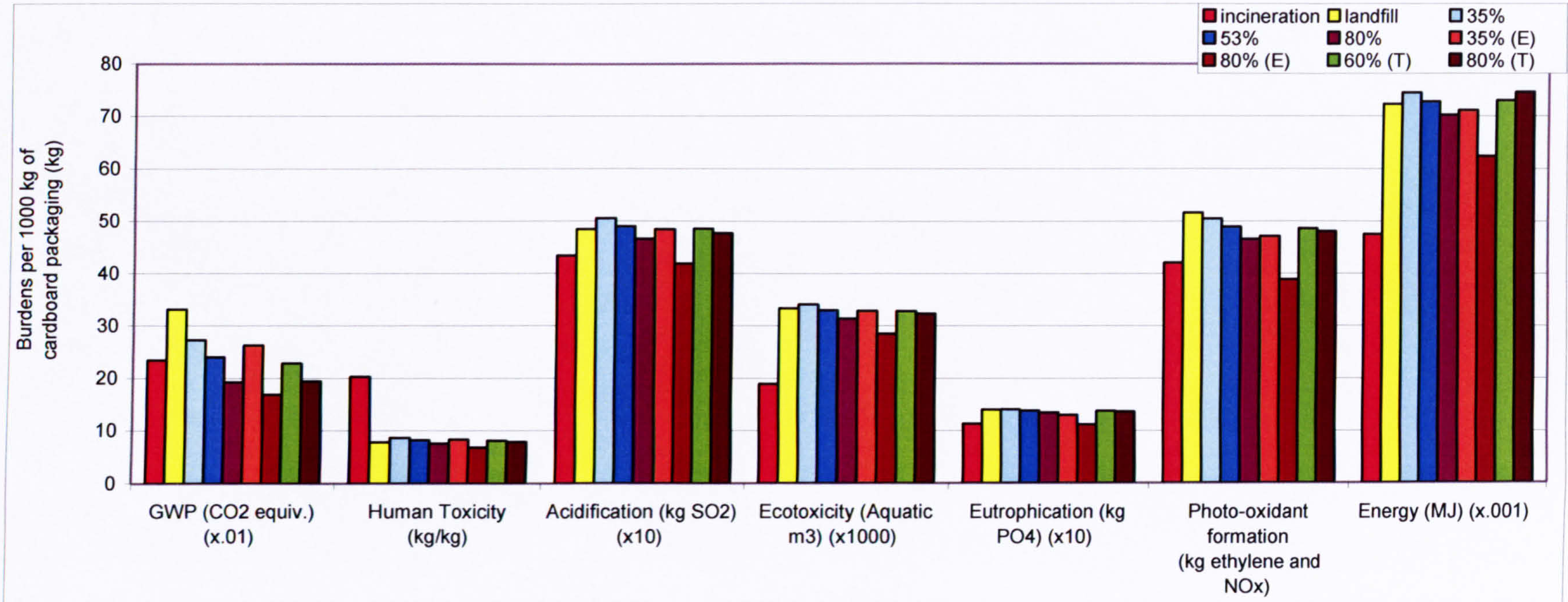

Figure 5.30: Summary characterisation

The modelling of cleaner energy usage scenarios showed that significant reductions in all impacts could be achieved, and that landfill and incineration are worse than recycling for most impacts. Comparing the present situation to a potential future scenario of 80% recycling coupled with cleaner energy use, would enable a 42% reduction in the global warming impacts associated with paper packaging waste to be achieved. This is the best possible outcome from all the scenarios considered.

Conversely, when perhaps more realistic assumptions regarding transport distances are considered, some of the benefits of increased recycling are moderated. For example, the benefits to human toxicity are reduced by almost half, at 53% to 80% the impact is reduced by 8%, whereas at 53% vs. 80% (T) this is only 4.4%. This is also valid for ecotoxicity.

All other emissions are similarly affected by about 2% - 3% compared to the standard scenarios. Although all emissions are increased by more transport, they mostly still benefit overall from higher recycling (i.e. the 80%(T) scenario is a better option than the 53% scenario). Energy is the exception, with a rise in energy use of 2.5% when 80%(T) is compared with the current situation (53%).

Two other relevant environmental impacts linked to recycling levels that are not presented in the characterisation but are of interest, are landfill space and transport. It was calculated that by increasing recycling levels from 53% to 80% requirements for landfill space could be reduced by 46%, but to the detriment of waste and recycled fibre transported by road that would rise by 16%. This would contribute to road congestion, noise and aesthetic impact, in addition to other emissions linked with transport such as those identified in the transport scenarios.

Recycling levels greater than 80% have not been considered, as it was felt that this would involve too much conjecture regarding the development of facilities and markets. However, it is likely that there would be limited benefit in going beyond this level. Although economic costs have not been explicitly considered within this study, it is likely that both the economic and environmental implications of attempting greater recovery are likely to outweigh benefits.

The results, as presented, represent a snapshot of potential environmental impacts. Some of which are high on the political agenda. They give a clear indication of which course of action would help in achieving set objectives, such as the UK commitment to reduce greenhouse gases contributing to global warming, in the context of the Kyoto Protocol. However, they may not be very comprehensive to stakeholders, and it is less clear which policy should be adopted in order to move towards the most sustainable solution in a broader context. In order to make a judgement on this type of policy objective it is necessary to employ valuation (weighting) methodologies, as described in the following section.

5.5 Valuation

The final stage of the LCA is valuation (also known as weighting), which applies weights to the identified impact categories and thus facilitates a comparison based on homogeneous units. For the present study, two different methodologies of valuation are presented, as they both have a very different approach to evaluating the environmental impact of a system.

Figure 5.31 presents the valuation results using the CML methodology, which is based on a scientific understanding of the effect of pollutants on the environment, also known as a midpoint approach. Figure 5.32 illustrates the valuation results using the Eco-Indicator 99 methodology, which presents the relation between the impact and the damage to human health and to the ecosystem, also known as an endpoint approach. Both methodologies have been described in section 2.2.6.4

This choice was governed by the literature review, where it was felt that too often the results of previous studies were only presented in economic terms and/or in terms of potential harm to humans. It seemed relevant to establish the results in an environmental approach first, giving the decision makers the opportunity to understand what types of emissions are resulting from the scenarios. CML methodology is also based on European figures which is relevant to this study. However, the results will also be presented using a damage function methodology so that the types of valuation can be compared and discussed in term of a decision-making approach. Eco-indicator 99 was selected for this approach. The choice was governed by the wide usage of eco indicator in general for LCA, and the fact that data are European. The Eco indicator 99 methodology is defined in section 2.2.6.4.2. EPS is another methodology using the damage function approach. EPS was perceived less applicable as it consider some more subjective impact such as aesthetic values, and the data are world average which is less relevant to the study.

Both methodologies are available in PEMS software. The weighting applied by each of the methods can be found in appendices 4 and 5. The approach used to develop the weights within each methodology is described in section 2.2.6.4. The researcher did not attempt to define its own set of weighting.

The CML weight were developed using the scientific knowledge of the impact emissions have, it also included the severity of potential impact.

Eco-indicator 99 weights were developed by using experts panel views (see section 2.2.6.4.2). It considers damages to human health, ecosystem quality and resources.

The first conclusion that can be drawn is that higher recycling translates into reduced environmental impact whichever methodology is used. However the worst scenario differs depending on the valuation method used.

5.5.1 Valuation result using CML

Figure 5.31 illustrates the results, in term of single indicators, of valuation using the CML approach. It is obvious that photo-oxidant formation is valued as the dominant impact category resulting from all scenarios, followed by global warming; both are noticeably reduced with higher recycling.

The CML approach rates 100% landfill as the worst scenario, this is mainly attributable to a much higher contribution to photo-oxidant formation and global warming.

The outcome of the valuation using CML is that at 80% recycling scenario the overall environmental impact is reduced by 11% compared to 53% recycling, by 30% when compared to 100% landfill, but only 6% compared to 100% incineration. It should be pointed out that incineration only becomes a worse scenario once 60% recycling level is reached.

This seems to indicate that the present waste management mix is not optimum whereby 93% of wastes that are not recycled are landfilled and only 7% incinerated. The overall environmental impact for all the recycling scenarios could be further reduced if this were split more equally, if not dominated by incineration.

Concentrating on the results drawn from the extra scenarios it is evident that the use of cleaner energy has large benefits. Comparing 35% to 35% (E), the total impact of the scenario is reduced by 7% and at 80% to 80%(E) this reaches 20%. The advantages of cleaner energy also increase with higher levels of recycling, thus when comparing 35% to 80% the overall environmental impact reduces by 15%, but reaches 26% for 35% (E) to 80% (E). The impacts that enjoy the largest reduction are photo-oxidant formation and global warming, both reduced by almost a third from the present situation (53%) to 80% (E).

Looking at the scenarios of transport, even with higher transport distances the overall environmental impact of the scenarios reduces with higher recycling, however, this is by a smaller magnitude. Indeed comparing 60% (T) to 80% (T) the environmental impact score is reduced by 6.5%, whereas for 60% to 80% the reduction is 8%.

For the recycling scenarios, the contributing impacts that benefit from the largest reductions are global warming and photo-oxidant formation. Looking at 100% landfill, global warming is largely higher than that for recycling. Whereas for 100% incineration, human toxicity contribution to the overall scoring is almost three times higher than for any other scenario. Indeed at 80% recycling it represent 4% of the total environmental impact compared with 11% for incineration.

One drawback with the CML valuation method is that decision-makers are left with environmental impact and no understanding of the potential damage inflicted through them. This is why it was decided to have a brief presentation of valuation results using a different methodology. Eco-Indicator 99 has been selected because by definition it presents the relation between the impacts and the damages to human health, the ecosystem and resources (see section 2.2.6.4.2), which may be more meaningful to stakeholders. In the context of eco-indicator 99 damages to human health means "the absence of premature death, sickness or irritations caused by emissions from industrial and agricultural processes to air, water and soil" (Goedkoop and Spiriensma, 1999).

5.5.2 Valuation results using Eco-Indicator 99

Figure 5.32 presents the same results from the inventory using an endpoint approach to interpret the environmental impacts. It is obvious from Figure 5.32 that the 100% incineration scenario has the worst score largely due to its potential carcinogenic damage. However new technologies (such are improve flue gases cleaning) and new regulation means that incineration emissions will be lowered and cleaner. The European Directive on Incineration set stringent pollution controls which will push for newer technologies already available to be put in place and invest further in research and development. The Directive came into force in the UK in 2002.

Here again a higher recycling level translates into a lower overall score by a third from 35% to 80%. Carcinogens benefit from the largest reduction, 42% at 80% compared to 35%, ecotoxicity 31% and climate change by 28%. However, ecotoxicity and climate change make only minor contributions to the overall score.

Looking at the clean energy and transport scenarios results, further recycling again results in lower scores. The overall scoring of clean energy scenarios is reduced by 30% from 35% (E) to 80% (E), which is marginally better than that of the standard scenario which is lowered by 28% when comparing 35% to 80% recycling scenarios.

Cleaner energy further enhances the benefits of recycling. Climate change scoring is reduced by 34% at 80% (E) vs. 35% (E) against 28% for the equivalent standard scenarios, and respiratory (organic) by 23% for clean energy scenario compared to 8% for standard scenarios.

Focusing on transport, again higher recycling still means lower scoring, but with reduced benefits. Between 60% to 80% the overall score is reduced by 15%, whereas for 60% (T) to 80% (T) it is only by 13%. The category for which there is a large increase in scoring is ozone depletion, which reaches 15% when comparing 60% (T) to 80% (T). For the same recycling levels in the standard scenarios this is a reduction of 2.7%. However, ozone depletion represents a very small (less than 0.5%) part of the overall score.

Two other impacts that also increase with higher transport distances are resource fossil and respiratory organic. The resource fossil's score rises by 2% when comparing 60% (T) to 80% (T), whereas for the same recycling levels in the standards scenario the score drops by 2.5%. The score for respiratory organic also rises by 0.4% for the transport scenarios against a fall of 3% for standard scenarios.

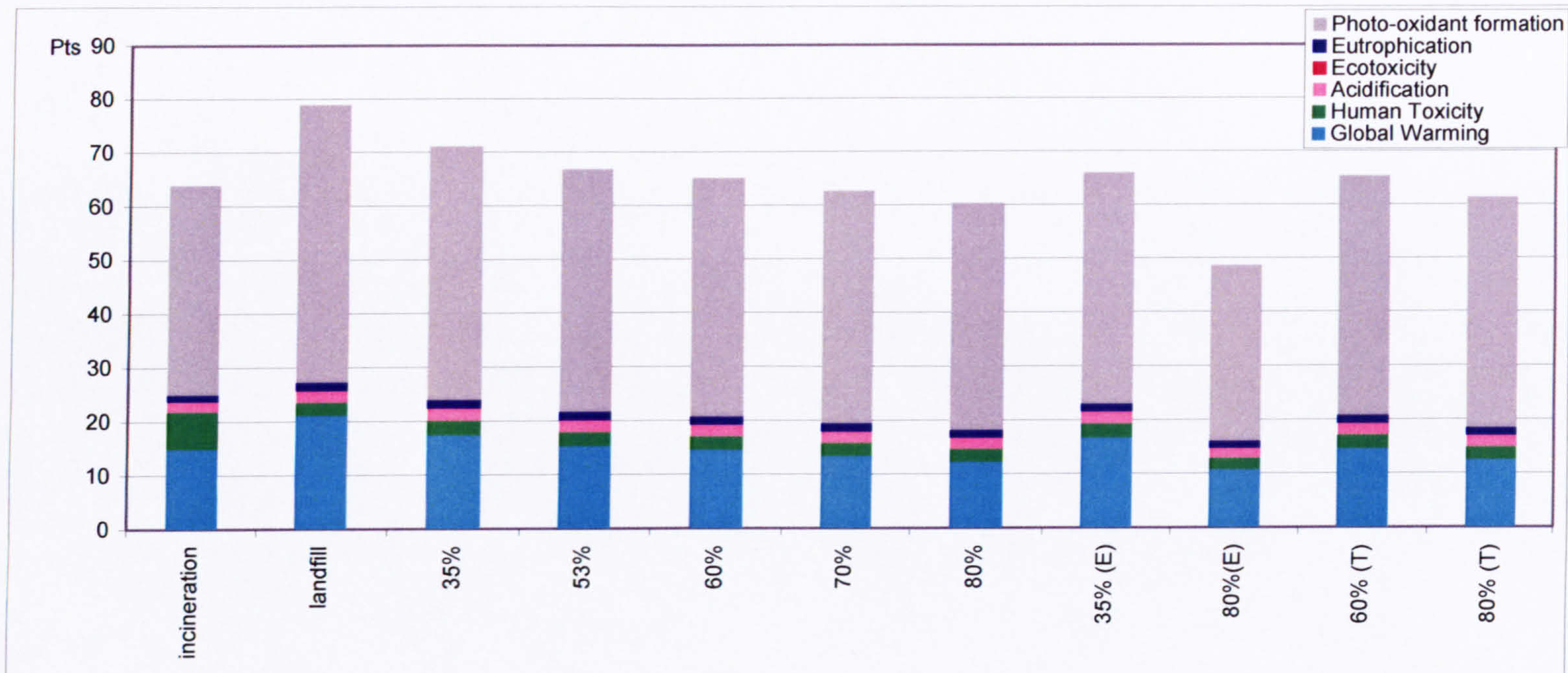

Figure 5.31: Valuation using CML

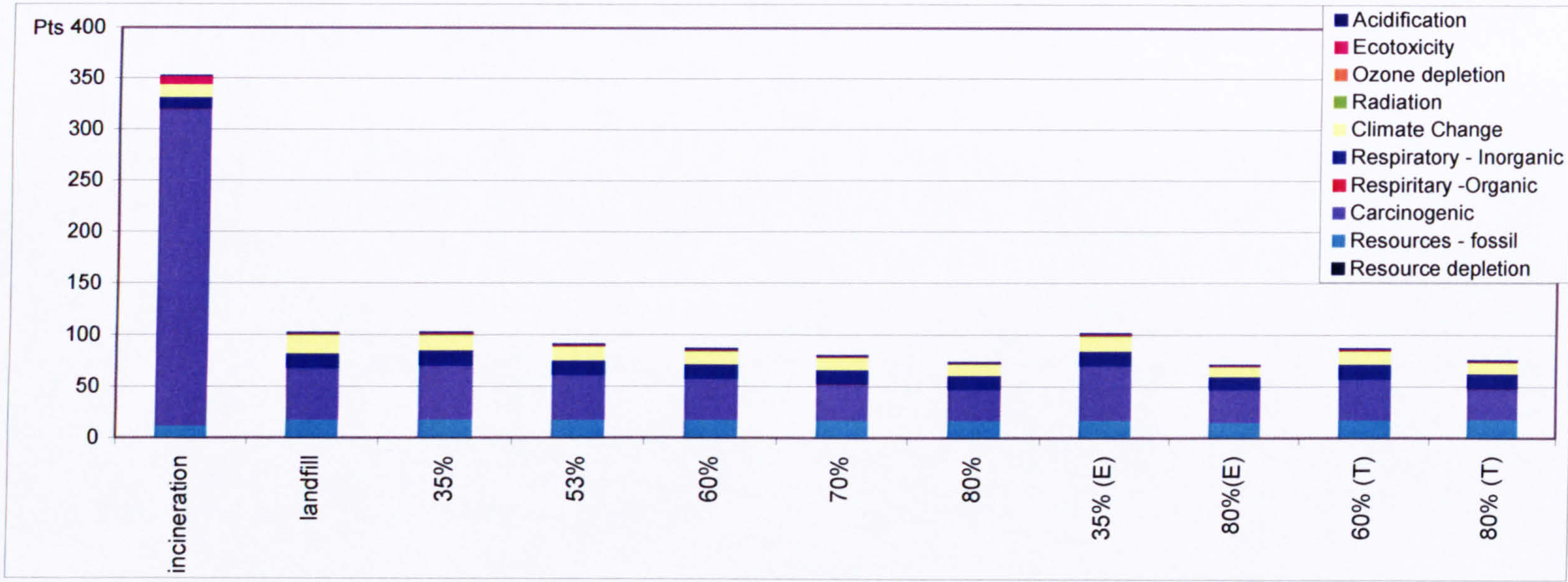

Figure 5.32: Valuation using Eco Indicator 99

5.6 Sensitivity analysis

The aim of the sensitivity analysis is to test the robustness of the assumptions underpinning the model.

Several assumptions are tested in this section. The first one is on transport and tests the sensitivity of the results if all returns are laden rather than empty. The second one is final waste disposal route and the last one concentrates on energy efficiency within cardboard manufacturing processes.

5.6.1 Transport

One of the assumptions made in the standard scenarios is that all transports returns are presumed empty. However, in the commercially focus world we live in, it is unlikely to happen. Transport represent a considerable financial cost and thus transport companies in general will maximise its use to reduce the financial cost per product tonne transported. Thus once a load has been delivered, be it at the cardboard manufacturing plant or at a recycling plant, the lorry is likely to collect more loads from other customer for his return trip.

In this situation the environmental burdens attributable to transport have to be re-allocated between the loads. Thus, if the round trip is done fully loaded then the environmental burdens should be apportioned to both loads. The allocation used in this case is 50% of the environmental burdens are allocated to the system under study. The other 50% environmental burdens are attributed to the return load and lay outside the boundaries of the studied system.

For the propose of the sensitivity analysis transport stages between - virgin materials and cardboard manufacturing, cardboard manufacturing and cardboard use, cardboard use and waste sorting, waste sorting and recycling, recycling and manufacturing – assume that all returns are made fully laden. The environmental impacts associated with return loads are assumed to be outside the boundary of this study.

5.6.1.1 Inventory results

Figure 5.33 represents the life cycle inventory summary for 100% incineration, 100% landfill and the scenario of 53% recycling and 80% recycling (standard scenarios) compared to the same scenarios when changing the transport assumptions (noted (L) in the graphs). It confirms that by assuming that all returns in transport are laden, most environmental burdens are reduced. The data presented here, again, are scaled so that comparisons can be made through the figures.

Some emissions are more sensitive to this assumption than others. Thus toxic substances, which represent very small quantities, see their emission levels reduced by a third. VOC general and organic waterborne emissions are reduced further by 15%, and non-heavy metal waterborne by 10% compared to standard scenario of 53% recycling.

A few emissions stayed almost unchanged with the new assumptions. Halides, VOC halogenated, heavy metals waterborne, and non-heavy metals airborne. This re-enforce the finding in section 5.1, where it was concluded that these specific emissions were most sensitive to waste management activities.

It should be noted that gases A (all CO_2) emissions are only reduced by around 3% compared to standard with the new transport assumption. This finding supports the one presented in section 5.2.3, where the waste block was found to be the main contributor to global warming.

Fossil fuel resources are reduced further by around 3% compared to the standard scenarios, which indicate that the overall impacts of fossil fuel are from manufacturing and reprocessing processes. This again supports the finding in section 5.2.3, where energy requirement fell with higher recycling, but also that a part of the energy requirements shifted between manufacturing and recycling.

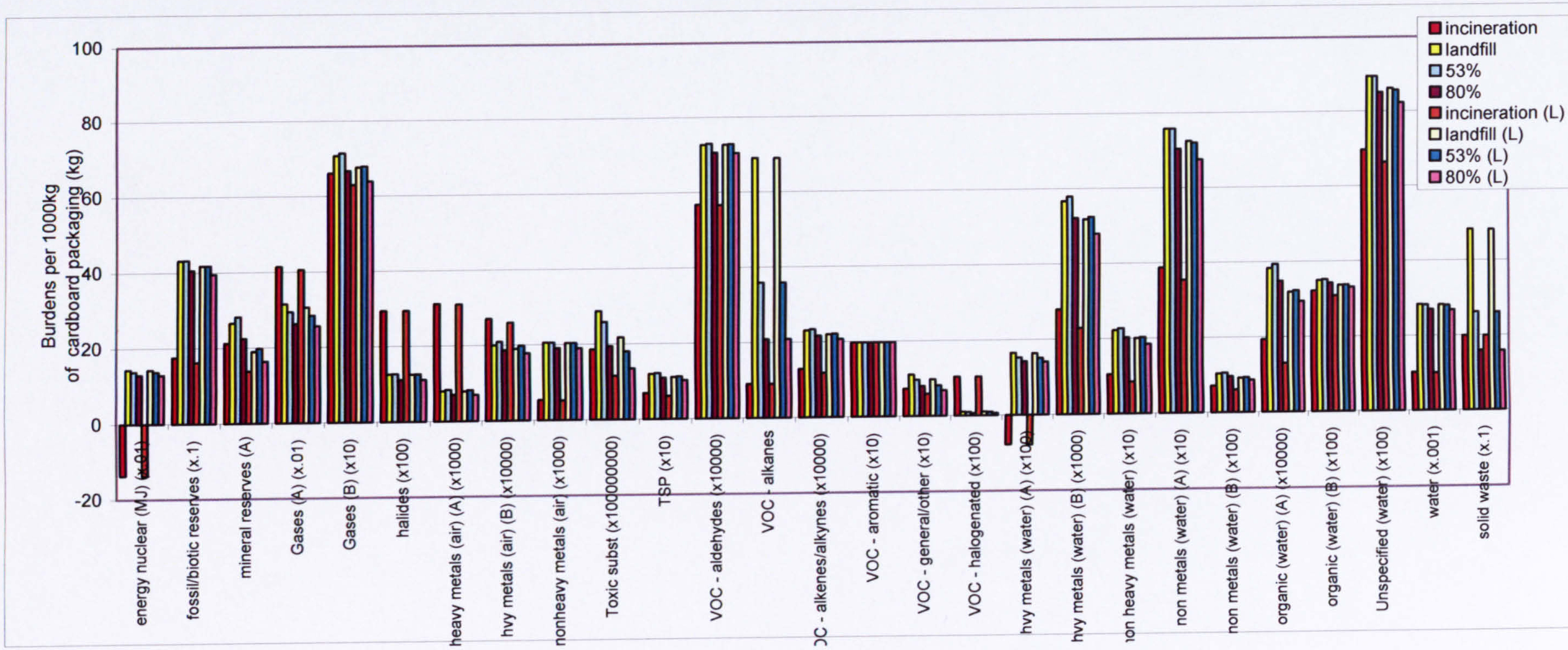

Figure 5.33: Inventory summary

5.6.1.2 Characterisation

Figure 5.34 presents at a glance the percentage variation from the original scenario the new transport assumption has for each environmental impact. It can be noted that all environmental impacts are reduced if all transport returns are laden rather than empty. The results presented here show the sensitivity (in percentages) of the results when testing the transport assumption used in the standard scenarios. From figure 5.35 it can be noted that it is energy (by up to 11%), acidification (by 3%) and human toxicity (by 5%) that are the most sensitive to the new assumption.

The following sections present a more detailed analysis of the results for the impacts the most affected by the new assumption.

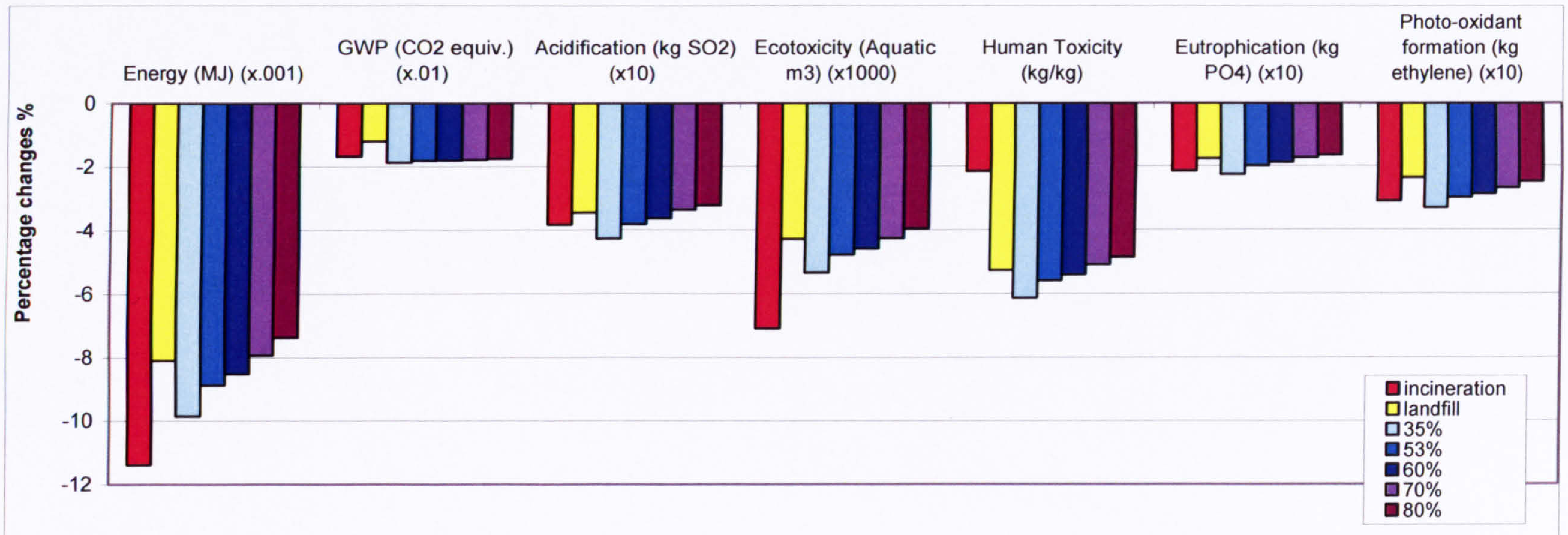

Figure 5.34: Summary characterisation assuming transport returns are laden

5.6.1.3 Human toxicity

Figure 5.35 shows that all emissions contributing to human toxicity are either reduced or unchanged when all return transport are laden. Overall the human toxicity impact is reduced by 6% at 53% recycling and by 5% at 80 when comparing the standard scenario to the new assumption. It can be noted that non-methane VOC are the emissions most sensitive to the new assumption with emissions reduced by 18%; these emissions are reduced by less than 2% when comparing 53% to 30% recycling in the standard scenarios. This indicates that transport is a main contributor to non-methane VOC, but that transport is only a small contributor to the overall impact of the system.

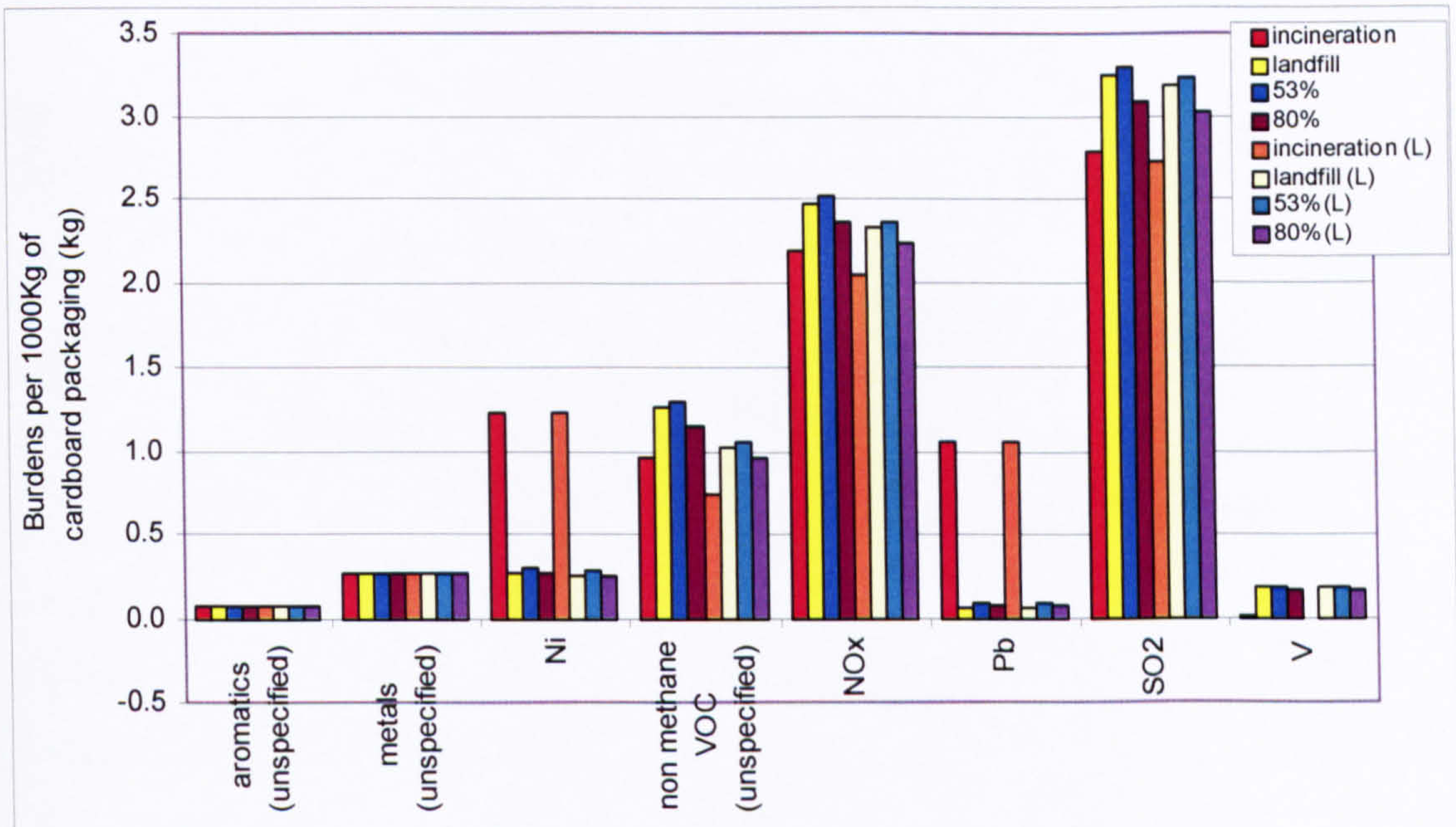

Figure 5.35 : Human toxicity

5.6.1.4 Acidification

The acidification impact sees a reduction of around 3% overall with the new assumption. As can be seen figure 5.36 NOx and SO_2 are the largest contributors to acidification impacts and are the most sensitive to the transport assumption. Yet the sensitivity levels are relatively small with 6% for NOx and 2% for SO_2. This strengthens the findings in section 5.2.3, where acidification reduces with higher recycling, and this is mainly linked the manufacturing processes rather than any transport.

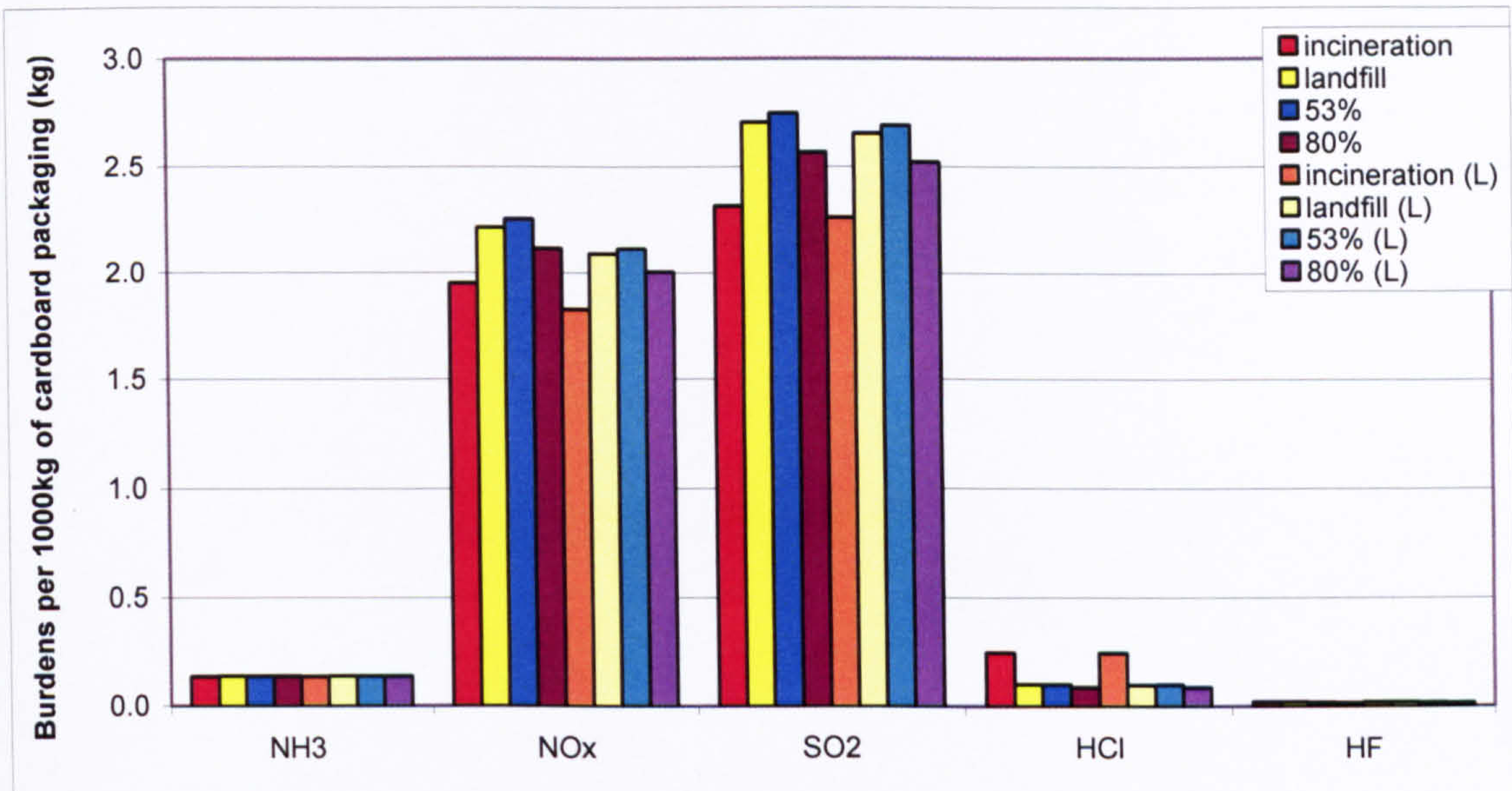

Figure 5.36: Acidification (kg SO_2 equivalent)

5.6.1.5 Energy resources

As expected energy resources are sensitive to the transport assumption by up to 11% for the incineration (reduced) overall. Gas reserves and oil reserves are the main contributor to this impact. Gas reserves are uncorrelated to transport assumption, whereas oil reserves are reduced by a quarter with new transport assumptions. This is explained by the use of gas energy for a majority of the manufacturing and recycling processes.

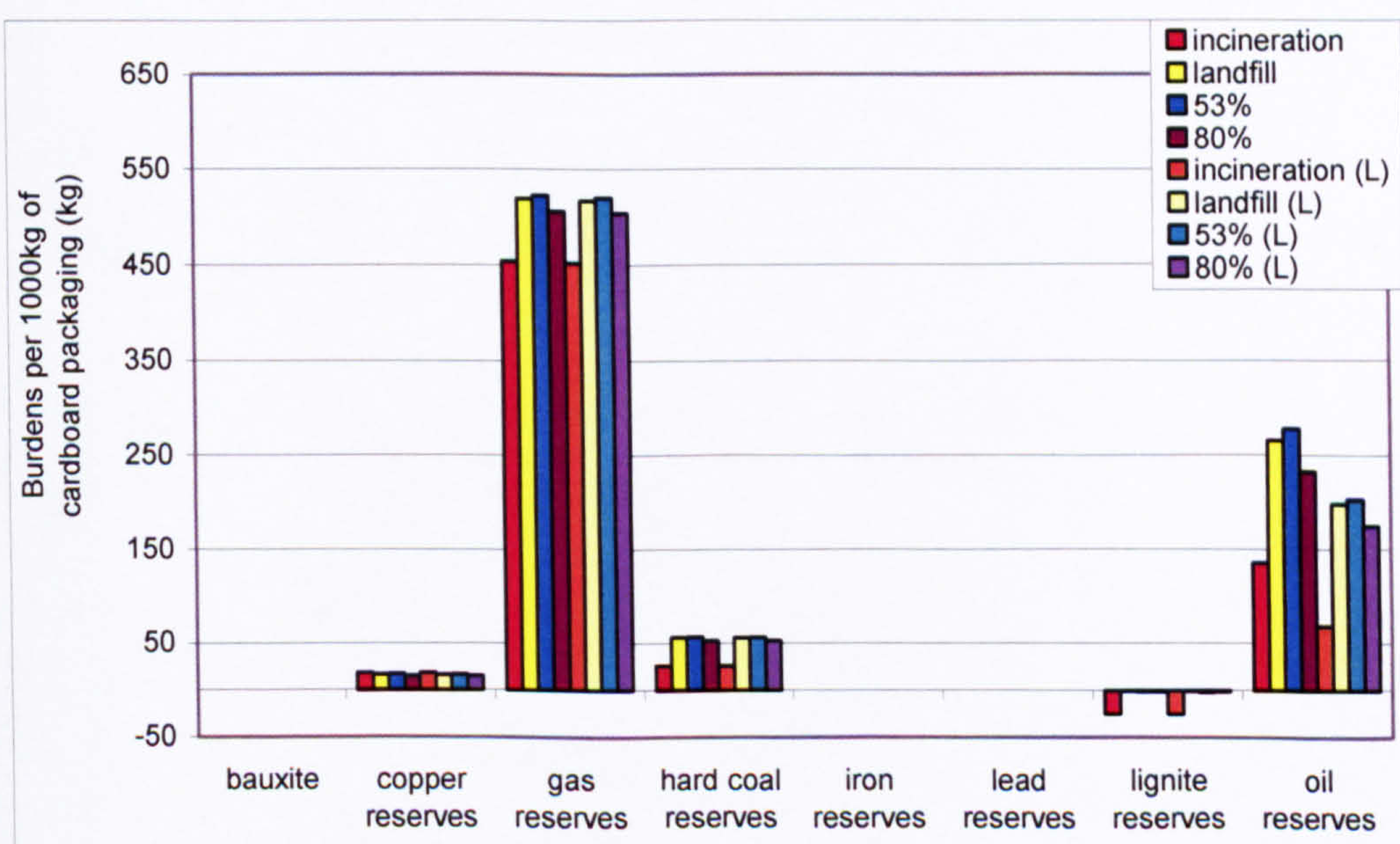

Figure 5.37 Energy resources (MJ)

5.6.1.6 Conclusion

The transport assumptions within the standard scenarios are sensitive to a few of the impacts. The main one being energy as expected. However the sensitivity analysis help highlighting that transport is not a big contributor to overall impacts.

The sensitivity analysis conclusion is that the transport assumption did not change the overall outcome of the model. The environmental impacts are reduced by higher levels of recycling in both assumptions, which indicated that the conclusions of the standard scenarios are robust.

5.6.2 Waste final disposal management

The next sensitivity analysis is focused on the final disposal of waste. At present the UK situation for waste final disposal is 93% are landfilled and 7% are incinerated. This is the assumption used in the model. However a new European Directive that came into force in 2002, and implemented in the UK legislation through several regulation, limits the amount and the type of waste going to landfill. In April 2005, the UK government introduced the Landfill Allowance Trading Scheme (LATS), which set limits to all local authorities in England on how much biodegradable waste they can landfill, in an effort to comply with the Directive. The aim is to increase recycling level, however a lack of investments to increase recycling facilities means that part of the targets will be achieved by increased use of incineration with energy recovery. Thus it can be assumed that in the near future more biodegradable waste (such as cardboard packaging waste) could be incinerated rather than landfilled.

5.6.2.1 The assumption

Here the assumption concentrates on the impacts of final waste disposal (incineration an landfill). Where ever in the models waste is disposed of, the following assumptions are tested: 10% incineration with 90% landfill, 15% incineration with 85% landfill, and also a 5% incineration and 95% landfill.

Figure 5.38 presents the percentage variation from the original scenarios for 53% recycling scenario and in figure 5.39 for 80% recycling scenario.

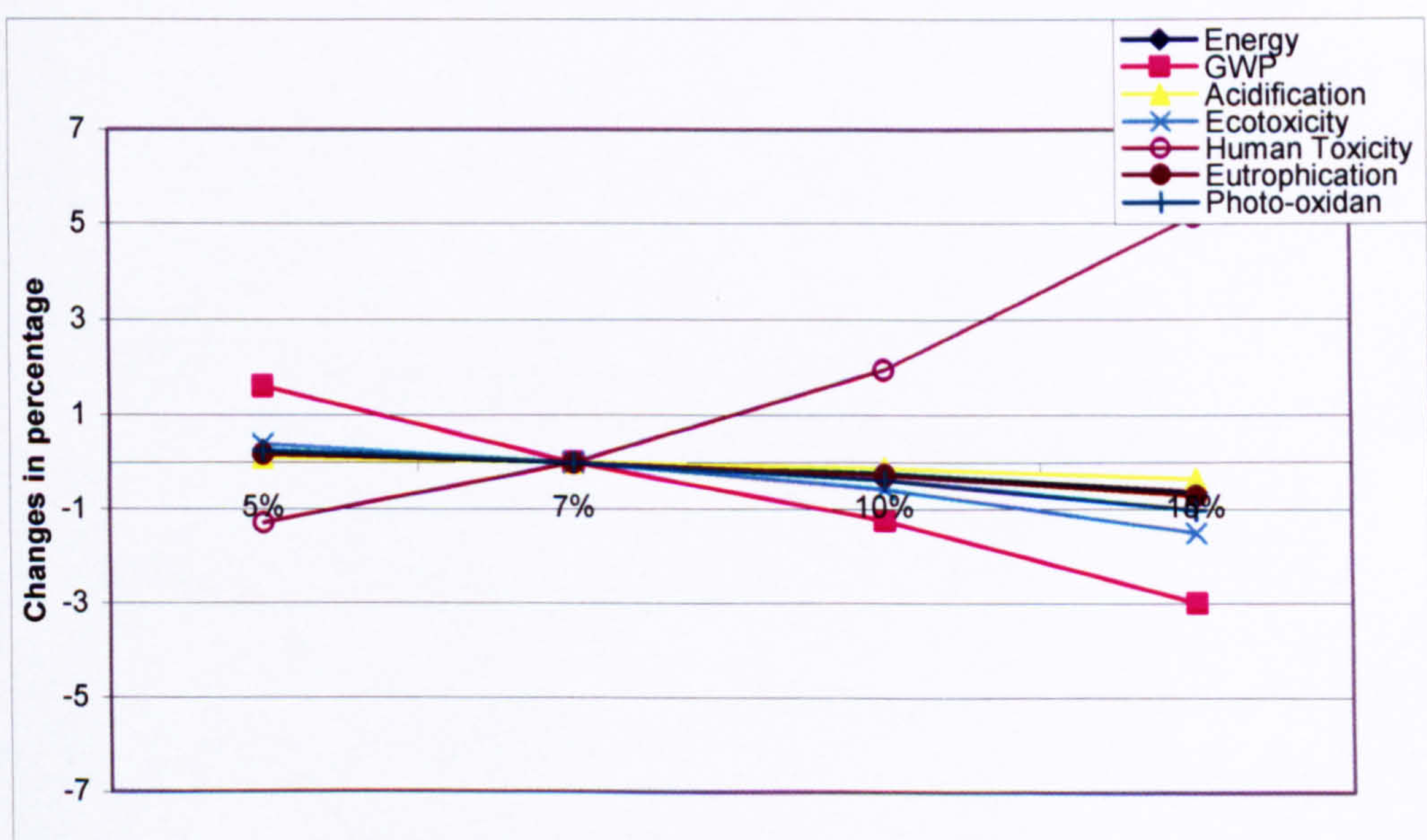

Figure 5.38: Level of incineration as final waste disposal (scenario with 53% recycling)

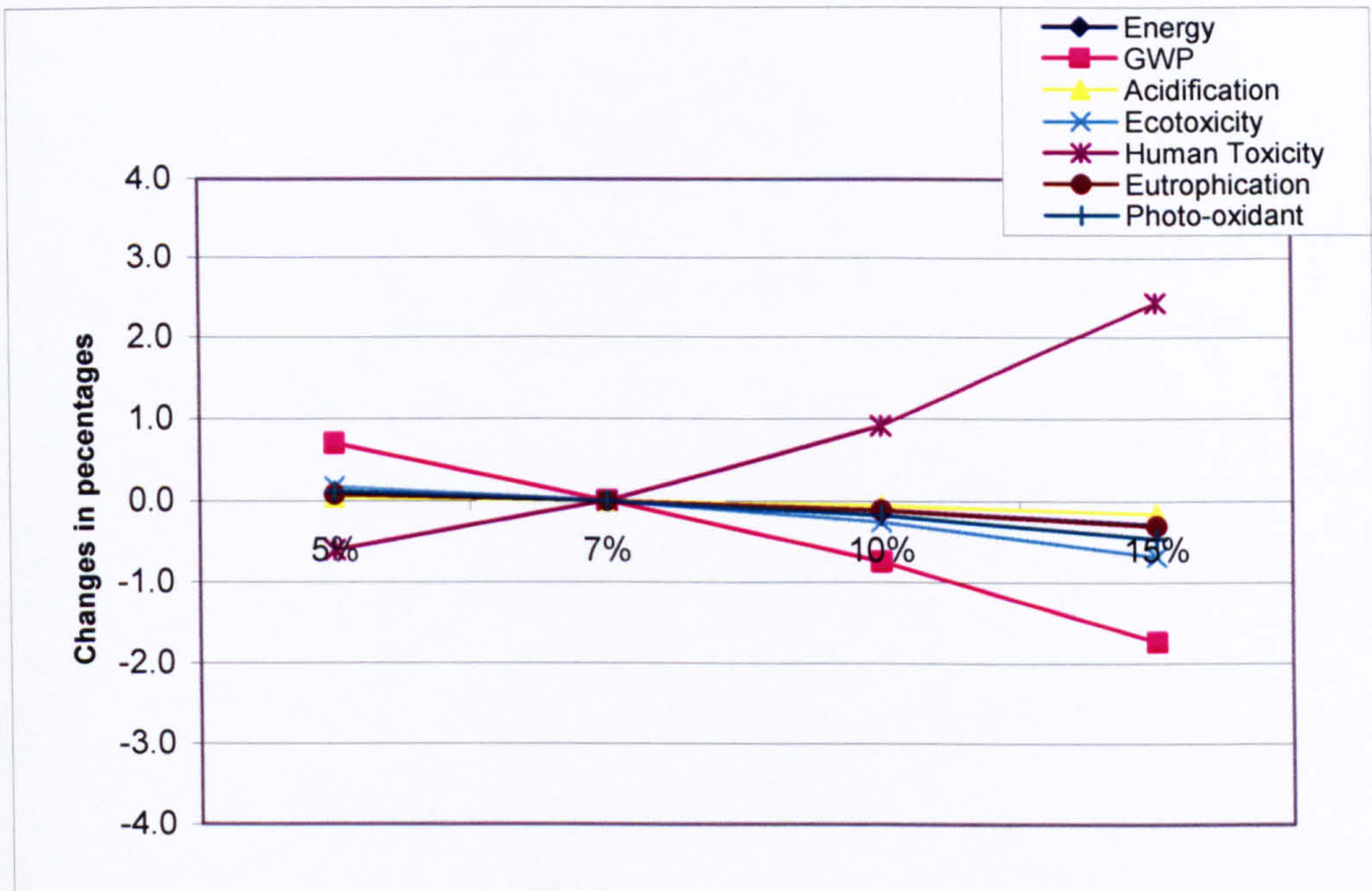

Figure 5.39: Level of incineration as final waste disposal (scenario with 80% recycling)

5.6.2.2 Outcome

Only two impact categories are sensitive to the new assumption, human toxicity and global warming. Figure 5.38 and figure 5.39 highlights that more landfill in general results in overall higher impacts (except for human toxicity) compared to incineration. Higher incineration results in higher human toxicity impact. At 53% recycling with 15% incineration results in a further increase by 5% of human toxicity compared to the standard scenario, for the 80% scenario it is increased by 2.5%.

The environmental impact most reduced by the new assumption is global warming, at 53% recycling, global warming is further reduced by 3% if 15% incineration is used. This was expected as 100% landfill standard scenario generates 27% more global warming than 100% incineration.

Overall this supports the findings in section 5.2.2.

5.6.2.3 Increased emissions

The emissions that are the most sensitive to changing the level of waste landfilled or incinerated are arsenic (As) and lead (Pb):

- Arsenic emissions rise by more than 50% when 15% of final waste is incinerated. However if only 5% of final waste is incinerated the arsenic emissions are then reduced by 13%. These are for 53% recycling model.
- Nickel emissions are increased by 13% when 15% of final waste is incinerated, compared to a 3% reduction when only 5% are incinerated.

This compare with the findings in section 5.2.2.3.

5.6.2.4 Reduced emissions

A few emissions are also reduced when incineration replaces landfilling:

- When looking at which emissions within global warming is most affected by incineration or landfill, CO_2 emissions are reduced by 12% in the 53% scenarios if 15% of the waste is incinerated. This support the findings in section 4.2.2.2 and section 4.2.3
- Methane, volatile organic compounds (VOCs) and Nitrogen emissions are all reduced by 6% when 85% of final waste is landfilled compared to 93% landfilled. These are for 53% recycling model
- Hydrogen sulphide (H_2S) and hydrocarbon are both reduced by 5%, when 85% of final waste is landfilled compared to 93% landfilled. These are for 53% recycling model.

5.6.2.5 Conclusion

Human toxicity impact is the most sensitive to the final disposal management of waste. It has been highlighted that few emissions are greatly increased when incineration level rises. This is explained by the level of contribution of these individual emissions to the impacts, though this is not to say they are not greatly damaging at low level of occurrence.

The sensitivity analysis conclusion is that final waste disposal assumption did not change the overall outcome of the model. The environmental impacts are reduced by higher levels of recycling in both assumptions, which indicates that the conclusions of the standard scenarios are robust

5.6.3 Valuation

The sensitivity analysis characterisation results were applied weighting (valuation) using the CML methodology and the Eco-indicator 99, as previously (see section 4.5).

5.6.3.1 Transport

CML valuation

Figure 5.40 presents the valuation results using CML methodology. The new assumption that all transports are laden rather than empty results in an overall lowered environmental impacts of 2% (for 100% landfill) - 3% (53% recycling). The order of magnitude in changes is relatively small. It is still the case that 80% recycling is better than 53% recycling, 100% incineration or 100% landfill. 100% landfill is still the worst and 100% incineration is still better than 53% recycling but worst than 80% recycling (see section 5.5.1). This indicates that the original scenarios were robust.

Photo-oxidant formation is still the dominating environmental impact; its overall reduction is of 3% for the incineration scenario and 2.5% for recycling.

Global warming is also still the second largest impact, and benefits from a 2% reduction overall.

It is 100% incineration scenario that benefits from the highest reduction of one specific impact: ecotoxicity is reduced by 7% when transport returns are laden; this is higher than for the recycling scenario at 4%.

Eco indicator 99

Figure 5.41 presents the valuation results using Eco-indicator 99 methodology. The eco-indicator 99 value the incineration scenarios as the worst with still the main contributor being carcinogenic (see section 5.5.2), which is unaffected by the new transport assumptions.

It is ozone depletion impact that benefits from the biggest reduction, 21% compared to the initial assumption. However it has to be emphasised that ozone depletion represents less than 1% of the overall valuation score.

Resource fossil, as expected sees a reduction of 9% at 53% and 8% at 80% recycling.

The 80% recycling scenario is still the better scenario having less environmental impact overall.

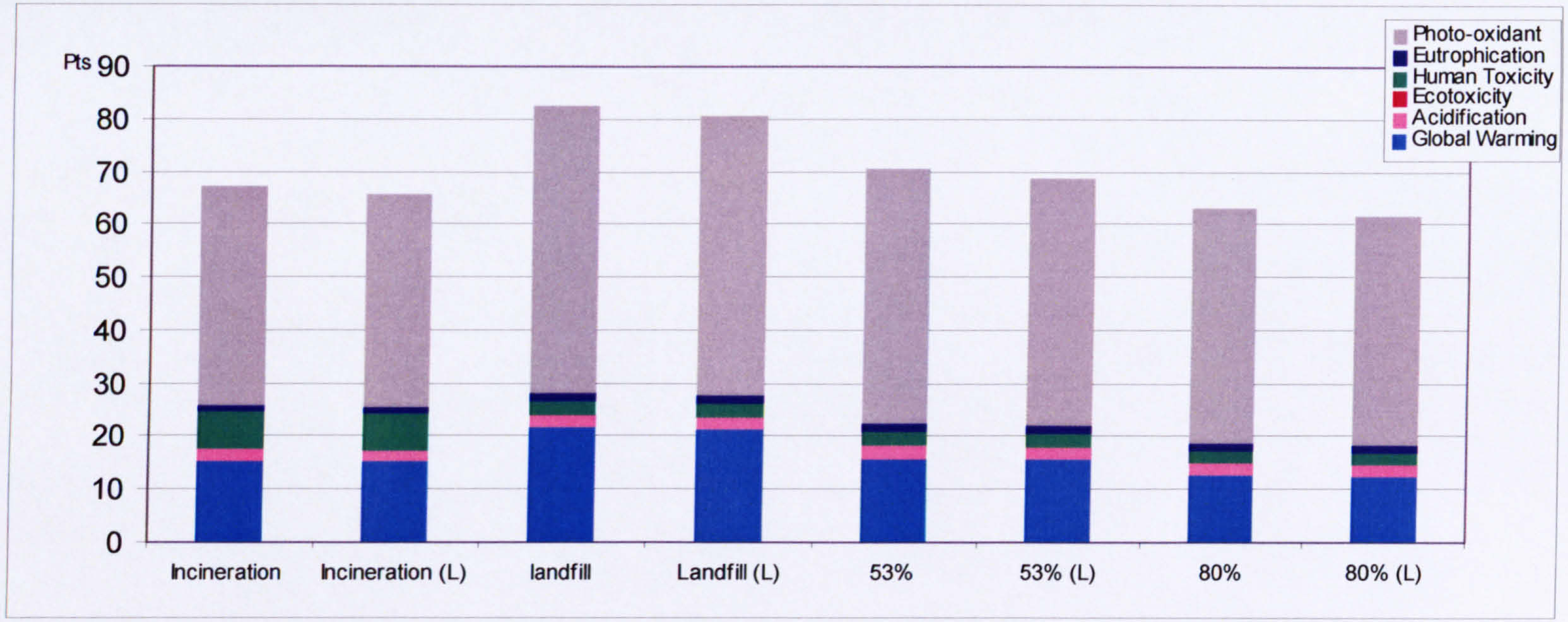

Figure 5.40 : Valuation of transport assumption using CML

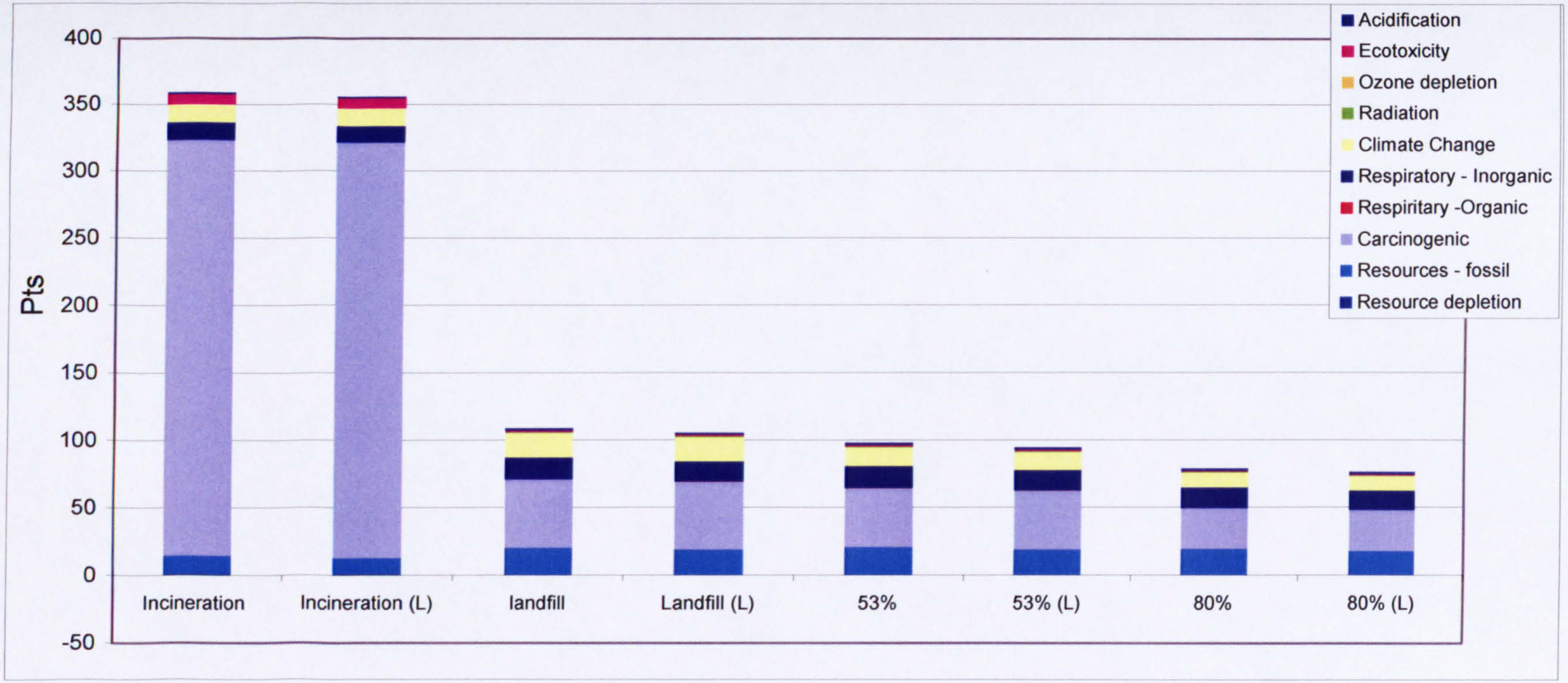

Figure 5.41: Valuation of transport assumption using Eco-indicator 99

5.6.3.2 Final waste disposal

CML valuation

The valuation results using CML methodology indicates little changes in the magnitude of the impacts (figure 5.42 and 5.43). Thus the overall score is left unchanged, this is mainly due to photo – oxidant impact representing more than 65% of the overall score, and is unaffected by the new assumption.

Most environmental impacts are little affected by the new assumptions, but see a small sensitivity in their score. It is global warming that benefits the most from the assumption with a reduction of 2.5% when 15% is incinerated compared to 5%. Human toxicity, which is the only impact to increase (+6%) with higher incineration.

The overall conclusion is consistent to that in section 5.5.1, which indicates that results are robust.

Eco indicator 99

Eco indicator 99 valuation results in a higher overall when more incineration is taking place (figure 5.44 and 5.45). Carcinogenic is still the main contributor to the overall score (see section 5.5.2). It is also the impact with the highest increase in scoring with higher incineration (+22%) at 15% incineration in the case of 53% recycling, and by (14%) at 15% incineration in the 80% recycling scenario.

Ecotoxicity also see a major increase in its score, equivalent to those describe above for carcinogenic. However ecotoxicity impact is only a minor contributor to the overall score.

The overall conclusion that higher recycling results in lower environmental impact is still valid as 80% recycling has an overall score 20% lower than at 53%.

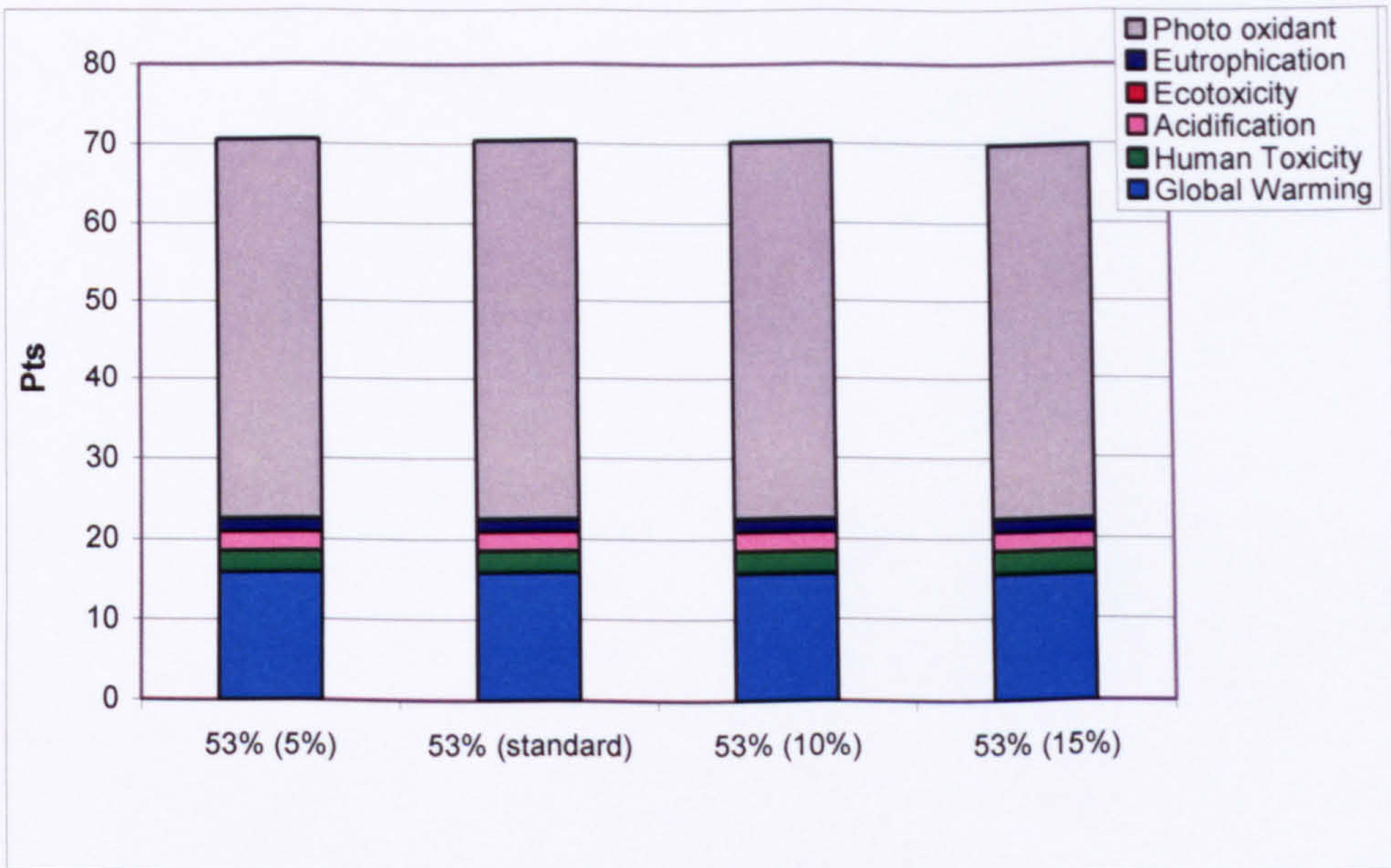

Figure 5.42: Valuation using CML for 53% recycling scenarios, final waste disposal

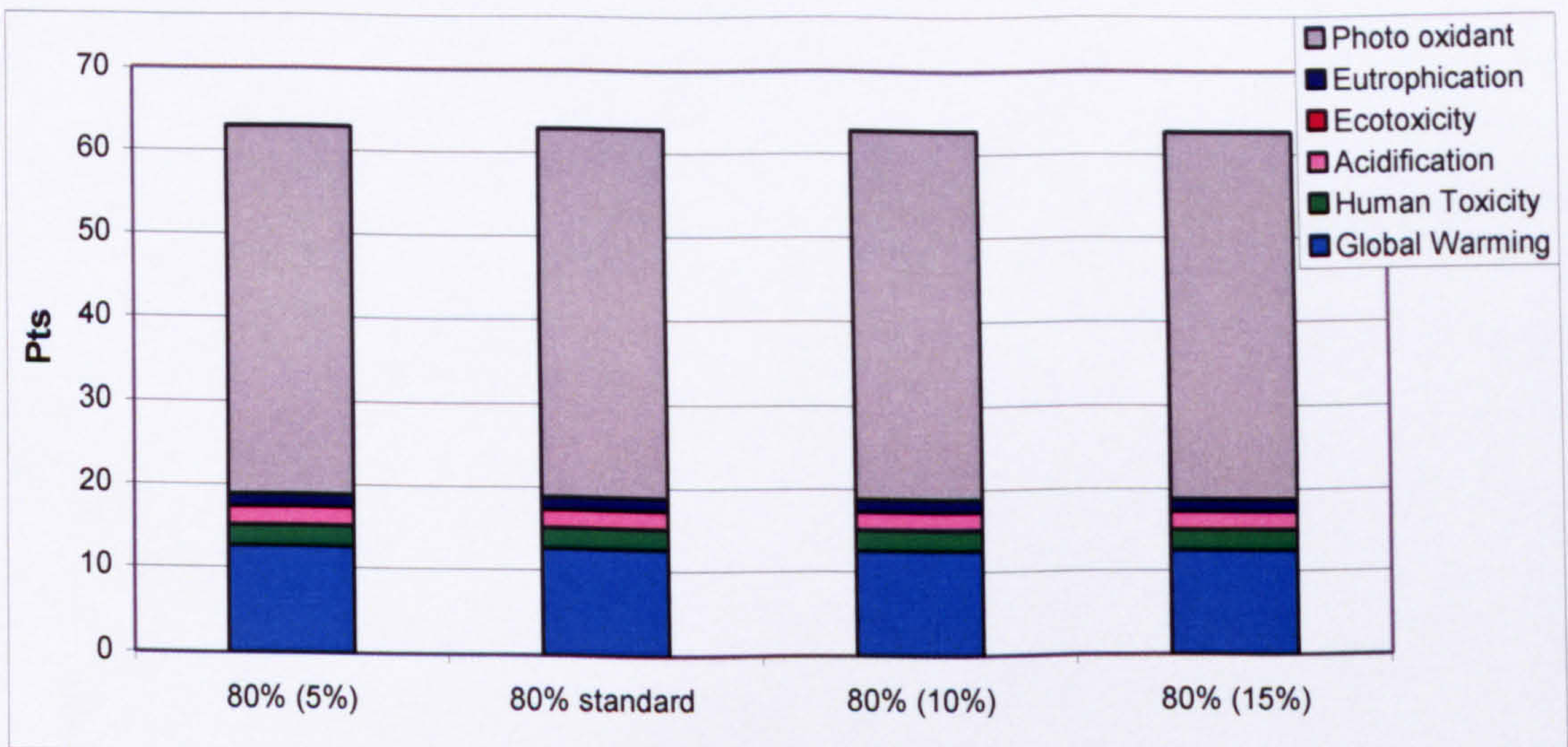

Figure 5.43: Valuation using CML for 80% recycling scenarios, final waste disposal

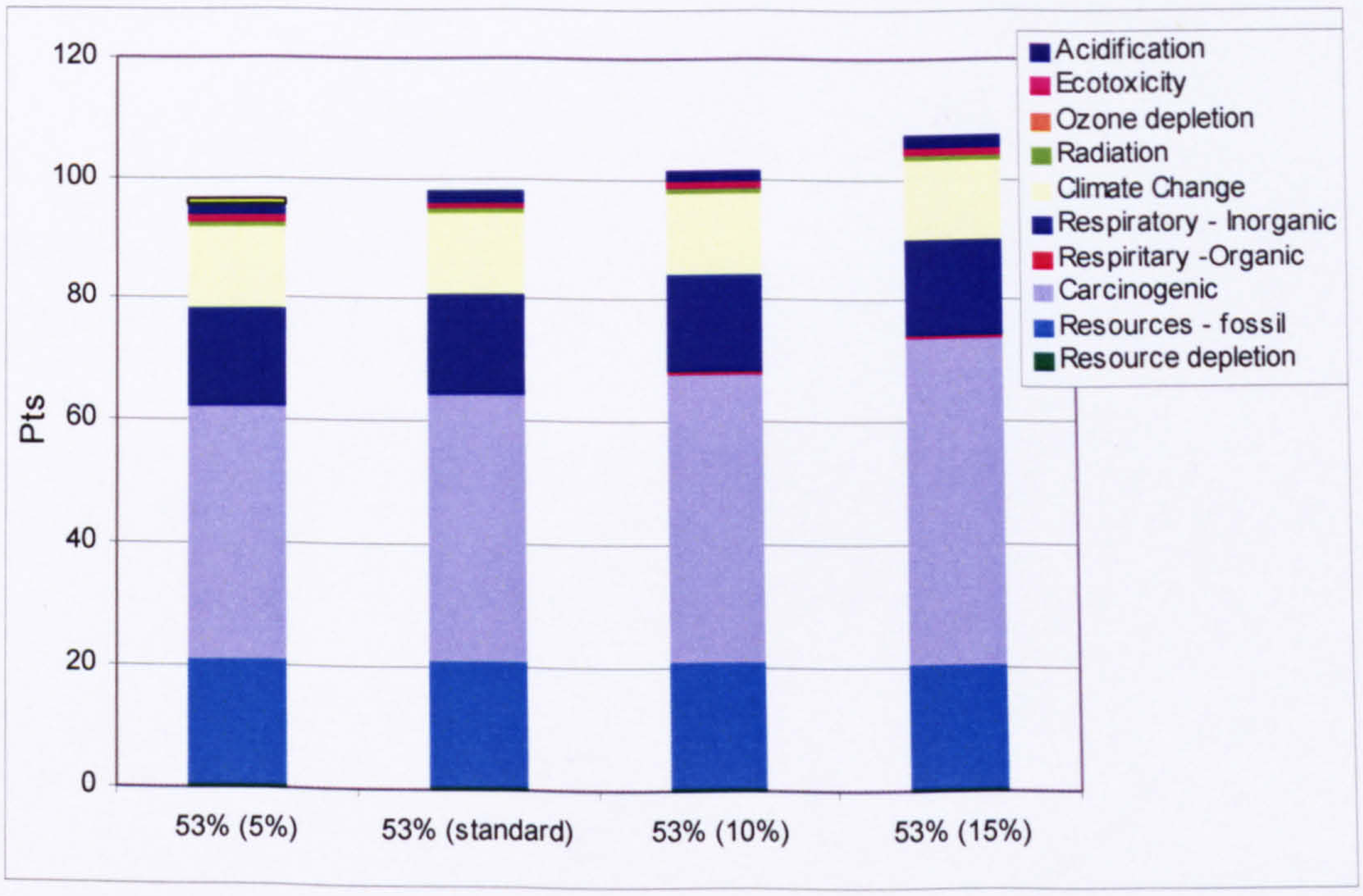

Figure 5.44: Valuation using Eco indicator 99 for 53% recycling scenarios, final waste disposal

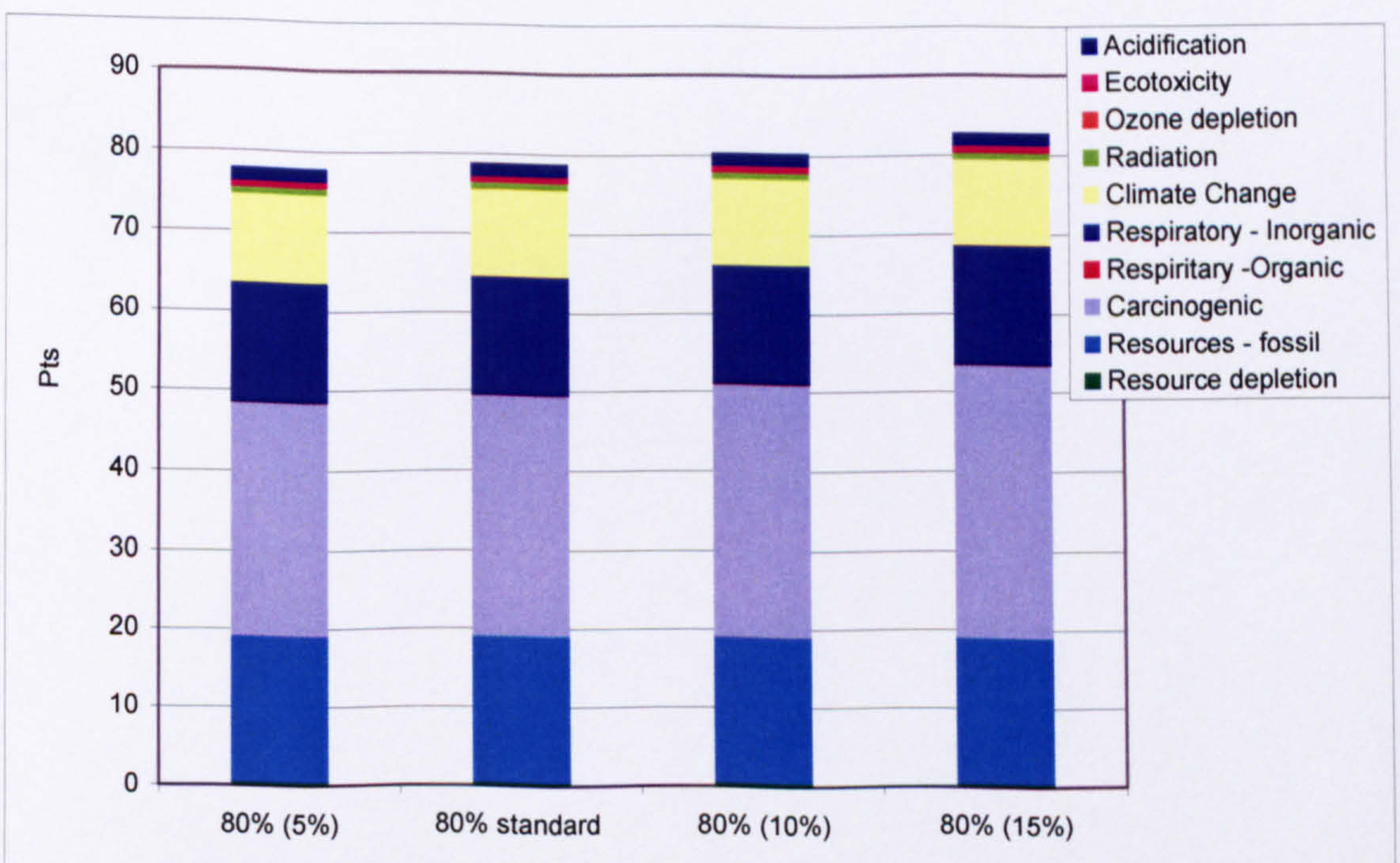

Figure 5.45: Valuation using Eco indicator 99 for 80% recycling scenarios, final waste disposal

5.6.3.3 Valuation Conclusion

It can be concluded from the valuation the results for the new assumption that higher recycling still translate into lower environmental impact overall.

For the new transport assumption using the CML methodology it is global warming and photo-oxidant that are the most reduced by the new transport assumption. The highest reduction was for ecotoxicity in the 100% incineration scenario. When using the eco-indicator 99 methodology it was ozone depletion and resource fuel that benefited from the greatest reduction.

For the new final waste disposal assumptions using the CML methodology, human toxicity are increased with higher recycling, whereas global warming is reduced. The overall scoring is little affected. When eco-indicator 99 is used, it the carcinogenic impact that dominates and sees the greatest reduction.

6 DISCUSSION AND RECOMMENDATIONS

6.1 Discussion

The results of the present study suggest that recycling cardboard packaging waste, at levels of up to at least 80%, is a better option than high levels of landfill or incineration, for the environmental impacts considered. The relationship between environmental benefits and increased levels of recycling is linear, and there does not seem to be a ceiling above which benefits start to decline and/or environmental costs start to become significant. Hence, the upper limits will be dictated by other issues, such as the ones discussed below.

6.1.1 Financial consideration

The optimum level of recycling will be dictated by the financial costs of reaching such recycling targets and society acceptance in changing behaviours. The UK current practice is based on maximising the diversion (recycling) of packaging waste from industrial and commercial sources. This is governed by two facts, the wastes are cleaner and easier to separate and it is also a cheaper approach. This was taken into consideration when developing the models and thus kerbside collections from households were not considered. So far this has been enough to meet the mandatory targets, however to achieve the new higher targets of 55% overall recycling, the packaging industry will need to rely on an ever-growing contribution from household waste. This is mainly true for glass and plastics packaging waste,. The implementation of more kerbside and bring collection schemes will increase the costs per tonne of material recycled. As demonstrated by Hummel (2001) kerbside and bring collections should be targeted at areas where high participation rates can be achieved with high quality separation. For example, kerbside collections in rural areas may be more expensive than in urban areas, but perhaps might result in higher participation and better quality in separated materials.

6.1.2 Secondary markets

Recycling is only likely to be judged sustainable if reliable local markets exist for the secondary materials. The sensitivity analysis highlighted significant reductions in the environmental benefits associated with high levels of recycling when it was assumed that the recycled materials had to be transported into Europe for sale/reuse. This is a particular issue in the context of the UK at present, as secondary market prices are not particularly stable. Indeed, the UK approach to implementing the Producer Obligations Regulation is very different from on the continent, in that it relies on market forces to dictate the prices of secondary materials, and to raise capital to develop collection and recycling facilities (see section 2.1.3.1). However, the markets have been very volatile so far, limiting the scope for substantial investment in additional recycling facilities and making it hard to plan for the future.

6.1.3 Consumer behaviour

Society's attitude towards recycling activities and towards the manufacture and use of products that incorporate significant quantities of secondary materials is critical. On one hand, households have to be engaged to participate in more recycling, on a regular basis and ensure high quality in waste separation. On the other, consumers and business/industry have to orient their choice and demand products with recycled material content. Thus ensuring market viability for the secondary material, they have helped to produce in the first instance (Coggins, 2001).

6.1.4 Political alternatives to recycling

The political attitude at present is to favour recycling as much as possible, as outlined in the National Waste Strategy for England and Wales (DETR, 2000), however over ambitious diversion level from landfill could mean higher use of incineration (see section 6.2 below). The Department of Environment Food and Rural Affairs has set the Waste and Resource Program (WRAP) on one hand to develop secondary market, but also to help local authorities to promote recycling. It is WRAP that is behind the new national Recycle Now campaign. However, as discussed above, this will not work unless there are markets and applications for the secondary materials. Yet, there are a number of laws that prevent the use of recycled materials for certain applications, the main one being food packaging.

The political emphasis regarding packaging should perhaps lie with greater promotion of waste minimisation. At present the regulations that deal with minimisation have been subtle and none have yet set targets to deal with this aspect. Clearly, if the use of packaging is minimized then the amount of waste would be reduced, as would the demand for resources and raw materials. However, packaging has an important function to fulfil in delivering products in a safe, clean and informative way. Reducing waste packaging through minimisation of waste or altering packaging through the use of recycled material might increase product wastage. This would have much more impacts on the environment, as products are primarily packaged to be protected, and their value (in terms of resources and economics) is greater than that of packaging (Monkhouse et all, 2004).

One final comment is that packaging wastes only represent 3% of total solid waste being landfilled in the EU, and in 1999 the cost of recycling 50% of these 3% was estimated to be 10 billion Euros. In this context any environmental benefits that can be reached through higher recycling may seem like a drop in an ocean, and it might be argued that it would be better to tackle waste streams that make a greater impact on our environment.

6.2 Recommendations

The impacts of different levels of cardboard packaging waste recycling targets have been modelled to establish the environmental impacts of different recycling targets. This particular packaging waste stream was selected for analysis because whilst the UK has already reached 53% recycling, a few other Members States have achieved much higher levels and are calling for higher targets within the European Directive (94/62/EC).

The research has considered recycling levels ranging from 35% up to 80% (the present level is around 53%). In the basic analysis it was assumed that the efficiency of the recycling process would not be affected by changes in the level of recycling. In reality, however, it is questionable whether sufficient capacity to recycle or to make use of recycled fibres exists within the UK. For this reason sensitivity analyses were undertaken to evaluate the impacts of transporting material into European markets for recycling. Further sensitivity analyses were undertaken to evaluate the impacts associated with cleaner energy use. Analyses for 100% landfill and 100% incineration have also been undertaken for comparison.

It seems appropriate to highlight the fact that the main part of the life cycle studied here bearing most of the burdens and contributing most to environmental impacts is the *manufacturing* of cardboard. At a recycling level of 80% cardboard manufacturing is responsible for two thirds of acidification and energy-related burdens, and for 62% in the case of ecotoxicity. Recycling activities, however, are the main source of global warming (41%) linked to the energy intensive process, photo-oxidant formation (60%) linked to high release of NOx emission in the process and eutrophication (65%) - related burdens (see section 3.4.2). Thus it might also be beneficial to concentrate on improving manufacturing practices to reduce the use of energy, chemicals and water. One further benefit of recycling is that solid waste disposal is reduced by 50% at 80% recycling compared with 35% recycling, although this particular concern may, alternatively, be addressed by a move towards increased incineration.

Given the current political pressures to address the issue of global warming, and the apparent benefits in this respect that increased recycling has been shown to produce, the data in this thesis presents a strong argument in favour of increasing the target for the recycling of paper packaging (up to 80%). However the current situation in the UK is that the paper and board reprocessing facilities is running at maximum facilities (ERM, 2002).

However there seems to also be an argument to increase the use of incineration with energy recovery as a final disposal route rather than relying so heavily on landfill. Indeed the results before valuation have shown that incineration had lower impact for most categories studied than landfill. The valuation outcome was that incineration has a significant contribution to carcinogenic damages. However, it should also be emphasized that the incineration data used here do not take into account the new emissions limits imposed by the new European Directive on incineration (2000/76/EC), and any improvements in technology that might arise in response to them.

Incineration represents on 7% of final waste disposal at present. The 1990s saw the closure of many incinerators in the UK dealing with municipal solid wastes. This was largely as a result of the introduction of legislation on emissions from incinerators, such as: the Environmental Protection Act 1990, and European Directive 89/429/EEC on air pollution from existing municipal waste incineration plants which took effect in 1996. This led to the closure of many municipal solid waste incinerators with minimal pollution control, which had been built in the late 1960s and early 1970s.

However, after the publication of the National Waste Strategy (DETR, 2000) it has been suggested that between 89 and 166 incinerators would be required (DETR, 2000) to meet the demanding targets of diverting waste from landfill. The turn of the millennium saw several incinerators planned, some of which have already been granted permission; many more are still in the consultation process.

The government would need to ensure that any increase in target levels is supported by appropriate investments to develop facilities and markets locally, to ensure that the environmental impacts are reduced to the optimum level. If diverted cardboard packaging waste had to be exported or transported with much greater distances then the benefits of recycling might be compromised.

The present study only identified the environmental impacts associated with higher recycling targets. No assessment of the economic costs of reaching such targets has been conducted. Since the LCA has not identified an optimum level of recycling, but rather that higher recycling results in reduced levels of environmental impact, the optimum reachable recycling targets will be dictated by economic and social costs. An economic evaluation would also identify the ratio of spending to environmental improvement reached, which would help in setting an optimum recycling level.

7 CONCLUSIONS

This thesis has presented the findings of the research on establishing the environmental impacts of packaging waste and recycling targets, in the context of the European Packaging and Packaging Waste Directive (94/62/EC). The study focused on cardboard packaging waste, and used LCA as the chosen methodology.

After presenting the aim and objectives of the research, an extensive literature review was presented which focused first on the increasing role played by environmental issues in politics and legislation. It outlined the development of the European Packaging and Packaging Waste Directive (94/62/EC) and the lengthy process for it to be implemented in the UK regulations. The Directive has been revised in the last 12 months and the new targets have been decided. LCA was chosen by he sponsor as the most appropriate methodology to conduct the study, due to its unique approach of "cradle to grave". Its principles, applications, methodological issues and limitations have been described. The use of CBA and CEA in combination with LCA to evaluate the economic relevance of a decision was investigated. Economic valuation can be used to convert LCA results into homogenous units, thus simplifying the comparability of results.

The manufacturing of paper and board has been described, including a review of environmental impacts links to the processes. Finally, the context of paper packing was reviewed.

The present study has assessed the potential environmental impact of selected recycling target levels for cardboard packaging waste for the UK based on the CML criteria. The range of studied targets was representative of the current UK situation (53%), and higher levels which are to be achieved by the new Directive targets for 2008, as well as even higher levels that might be enforced in the future. Extreme scenarios, specifically 100% landfill and 100% incineration, have also been considered. Several assumptions underpinned the model. The boundaries of the model exclude the forestry impact and start when wood chips are delivered to a mill. They include raw materials, cardboard manufacturing, and recycling activities. They also include energy and transport. The boundaries stop at the final disposal of waste. It was assumed that 67% of raw materials are recycled fibres, and that the recycling taking place is re-used within the model. Of the portion of waste not recycled, 93% is landfilled and 7% incinerated.

The valuation stage by weighting the characterisation results puts into perspective the importance of impact. The characterisation results and the valuation results can point to different conclusions. Thus, the characterisation stage quantifies the relative contribution of individual environmental emissions to relevant impact categories. Throughout the characterisation stage, incineration came as a better scenario for a number of considered impact categories. However, at this stage the severity of the potential impact were not included. The use of two different methodology using different approaches for the valuation highlighted the difference in results that can be obtained. For the case of incineration, the potential level of damage especially to human health (mainly due to arsenic emissions) is high and thus sees 100% incineration scenario as a high scorer, in the case of Eco-indicator 99 valuation. Yet, when using CML methodology 100% incineration only becomes less desirable scenario once 60% recycling take place. It has to be stressed that this does not mean that landfill or incineration are the better options.

In addition two further analyses were conducted of the scenarios to assess the impact of modifying two parameters. The first variant to be considered was the use of cleaner energy in the recycling process. The second focused on scenarios in which it was assumed that separated waste over and above the current level of 55% was exported to the continent for reprocessing, thereby incurring higher levels of transport-related burdens.

Finally two sensitive analyses were conducted to evaluate the robustness of the model. The transport assumptions underpinning the model assumed all transport returned empty, the sensitive analysis considered the impact if all transport returned fully laden. The second sensitivity analysis focused on final waste disposal option. The balance between landfill and incineration was modified.

The sensitivity analysis confirmed that higher recycling level always resulted into overall lower environmental impacts.

The model developed focused only on the environmental impact and did not address whether the scenarios are technically achievable within the UK context.

The results have been analysed in terms of life cycle inventory, impact characterization and two alternative valuation methodologies (the CML midpoint methodology and the Eco-Indicator 99 endpoint methodology). The results of the life cycle assessment were reviewed for the standard scenarios as well as the clean energy and transport ones.

From the modelling, one conclusion is certain, higher recycling reduces the overall environmental impact of the scenarios. The characterisation exercise based on CML databases identified global warming as the main benefactor with a 20% reduction, if the level of recycling were to be increased from 53% to 80%, energy usage is lowered by 8% and photo-oxidant formation by 8%. 100% incineration appeared to be the better option for most impact categories, apart for human toxicity which is 62% higher than for 80% recycling level. 100% landfill was identified as having a major impact in global warming, 42% higher than at 80% recycling. The use of cleaner energy would result in an overall reduction of environmental impact. The most dramatic reductions are for global warming (42%), and photo-oxidant formation (26%) when comparing 53% to 80% (E). When greater distances were considered for the transportation of diverted waste, the additional burdens were comparatively minor, and the apparent benefits associated with increased levels of recycling were unaffected.

The valuation of the results was achieved using two different methodologies, CML and Eco-Indicator 99. They both conclude that higher recycling would result in an overall reduction of environmental impact. However, conclusions differed on the comparative environmental impacts associated with the 100% landfill and 100% incineration scenarios.

The CML valuation scores 100% landfill as the worst scenario. The score for incineration is comparable with the 60% recycling scenario, with recycling at levels greater than 60% representing the best option.

The Eco-Indicator 99 valuation favours recycling over landfill or incineration, with increased benefits associated with higher levels of recycling. 100% incineration has the worst score, three and half times more than any other scenario, mainly due to the impact on human toxicity. 100% landfill scores similarly to 35% recycling.

The different valuation results emphasize that LCA is only a tool to inform decision-making and cannot provide one definitive answer to a given problem. However, as 80% recycling represents the best option for both valuation approaches, it may be concluded that any move to increase the target level for the recycling of cardboard packaging can most likely be justified on environmental grounds. The present research concentrated on – and the results are only valid for – paper packaging waste, but the same approach can be used for other packaging materials.

While further recycling will reduce environmental impacts, incineration and landfill can be used to a better extent. In the present situation, the UK landfills 93% of the waste paper that is not recycled, and only 7% is incinerated. Yet, the results presented here showed that incineration has lower impact compared to landfill for almost all categories considered. It is only with the use of weighting (Eco-Indicator 99) that incineration appears to be much worse, due to its contribution to carcinogenic damage.

The results of the present study suggest that recycling waste paper packaging, at levels of up to at least 80%, is a better option than high levels of landfill or incineration. The relationship between environmental benefits and increased levels of recycling is linear, and there does not seem to be a ceiling above which benefits start to decline and/or environmental costs start to become significant. Hence, other issues will dictate the upper limits. Some of which are the financial costs to implement the necessary infrastructure and actions to reach such targets, the development of secondary markets, as well as a shift in consumer and industry behaviours regarding recycled product but also in contributing to recycling.

8 SUGGESTIONS FOR FURTHER WORK

The present study concentrated on establishing the environmental impact of cardboard packaging waste only. Further work is needed to establish the optimum level of recycling targets for the Directive.

In order to establish an optimum level of recycling target, an economic assessment for reaching different levels of recycling need to be undertaken, and establish a ratio of economic spending to environmental benefits. This can be achieved using Cost Benefit Analysis methodology. As presented in the literature review CBA and LCA can be combined. This would be the next step after this study.

In the literature review (section 2.3.2) it was identified that CBA is a method used in the development of projects and policies to evaluate their social impacts. CBA establishes whether or not a project or policy is worthwhile from a social welfare perspective. An economic cost benefit analysis would require that all the economic costs involve at every stage of the model under study be considered, this means respecting the same system boundaries and including all stages, the same needs to be done for any economic benefits that arise within the system.

In order to be able to assess the economic cost with the environmental impacts all parameters to be considered will require to be in the same units. An economic valuation can convert the identified environmental impacts into monetary value. However several economic valuation techniques exist. These were presented in the literature review (section 2.3.4) as methodologies concerned with estimating the value that individuals place on non-market goods and services as well as individuals' preferences for environmental improvement or conservation, or individuals' loss of well being because of environmental degradation or from losing an environmental asset a 'value' can be revealed by a consumer's behaviour derived from their stated 'willingness to pay' (WTP) or 'willingness to accept compensation' (WTA) for example.

The results of the economic valuation will have converted the environmental impacts into monetary values, which then can be directly incorporated with the economic costs/benefits findings.

The analysis will then compare the monetary value of benefits with the monetary value of costs. An optimum level of recycling target will be reached when the social benefits outweigh the costs.

The methodology used for this research could be applied to all the materials covered by the Directive. For each individual material an LCA of the whole life cycle need to be conducted (not just packaging waste). This will require identifying the relevant recycling level set to these materials, what has been achieved so far and what are the potentials for improvements. A full life cycle inventory and its impact assessment will be completed and valuation will be used to assess the results. Again as above a CBA and economic valuation can be used to establish optimum level of recycling.

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