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Environmental Improvement Potentials of Meat and Dairy Products

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PREFACE

This report on 'Environmental improvement potential of meat and dairy products' is a scientific contribution to the European Commission's Integrated Product Policy framework, which seeks to minimise the environmental degradation caused throughout the life cycle of products. A previous study coordinated by the JRC (EIPRO study) had shown that food and drink is responsible for 20 % to 30 % of the environmental impact of private consumption in the EU, with meat and dairy products contributing most.

This report first presents a systematic overview of the life cycle of meat and dairy products and their environmental impacts, covering the full food chain. It goes on to provide a comprehensive analysis of the improvement options that allow reducing the environmental impacts throughout the life cycle. Finally, the report assesses the different options regarding their feasibility as well as their potential environmental and socioeconomic benefits and costs.

The report focuses on improvement options in three main areas:

- Household improvements, mainly to reduce food losses (wastage) and to reduce car use for shopping;
- Agricultural improvements, mainly to reduce water and air emissions (in particular nitrate, ammonia and methane) and land requirements;
- Power savings in farming, food industry, retail, catering, and for household appliances.

The study presents the consequences that the adoption of these options might have on a broad range of different environmental issues, including global warming, eutrophication, respiratory health impacts, etc. It shows that when all environmental improvement potentials are taken together, the aggregated environmental impacts (external costs) of meat and dairy products may be reduced by about 20 %. The study has also quantified the economic costs and benefits of implementing the different options.

EXECUTIVE SUMMARY

Introduction

The Communication on Integrated Product Policy (COM(2003) 302 final), announced that the European Commission would seek to identify and stimulate action on products with the greatest potential for environmental improvement. This work was scheduled into three phases:

- the first phase consisting of research to identify the products with the greatest environmental impact from a life cycle perspective;
- the second phase which consists in the identification of possible ways to reduce the life cycle environmental impacts of some of the products with the greatest environmental impact;
- in the third phase the European Commission will seek to address policy measures for the products that are identified as having the greatest potential for environmental improvement at least socioeconomic cost.

The first phase was completed in May 2006 with the EIPRO study, which was entrusted to the JRC-IPTS by DG Environment. The study identified the products consumed in the EU having the greatest environmental impact from a life cycle perspective. The study showed that groups of products from only three areas of final consumption – food and drink, private transportation, and housing, which account for some 60 % of consumption expenditure – are together responsible for 70-80 % of the environmental impacts of final consumption.

Based on these conclusions, and on DG Environment's request, three parallel projects were launched by the IPTS, dealing with the environmental improvement of products (IMPRO, respectively IMPRO-car, IMPRO-meat and dairy, and IMPRO-buildings).

This is the report of the IMPRO-meat and dairy project.

Objectives

The study first estimates and compares the environmental impacts of meat and dairy products consumed in EU-27, taking into account the entire value chain (life cycle) of these products. It then identifies and analyses the potentials of improvement options for the processes in the value chains that contribute most to the environmental impacts, focusing on options with proven technological feasibility and short to medium-term implementation horizon. Finally, it assesses the socioeconomic impacts of the improvement options, their relations to autonomous developments and current policies, and their feasibility of implementation. Targets and measures for the implementation of the improvements are suggested.

Environmental impacts

The study finds that the consumption of meat and dairy products contributes on average 24 % of the environmental impacts from the total final consumption in EU-27, while constituting only 6 % of the economic value. For the different impact categories, the contribution of meat and dairy products varies from 6 % to 47 % of the impacts from the total final consumption in EU-27. See Figure 1.1.

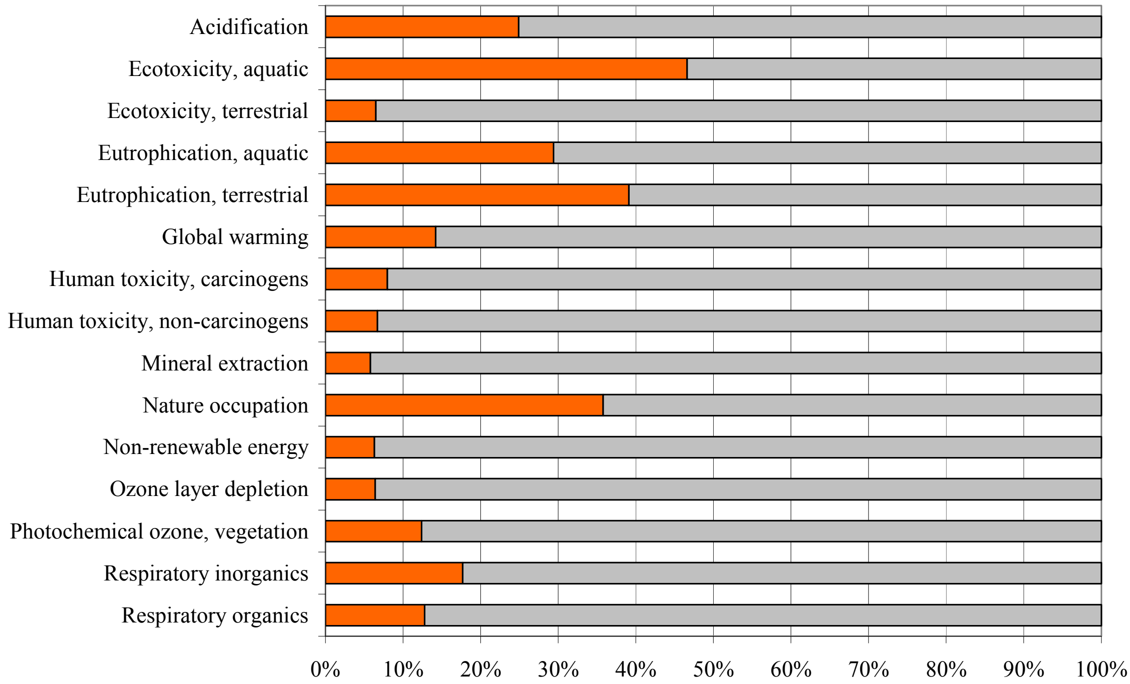

Figure 1.1: **Percentage contribution of meat and dairy products to the environmental impacts of EU-27 total final consumption.**

The monetarised environmental impacts (externalities) are of considerable size compared to the private costs of the products (from 34 % of the private costs for pork to 112 % of the private costs for beef). The large uncertainty on the monetarisation implies that this proportion may be an order of magnitude smaller or larger.

The four main product groups (dairy, beef, pork and poultry products) contribute respectively 33-41 %, 16-39 %, 19-44 %, and 5-10 % to the impact of meat and dairy products consumption in EU-27 on the different environmental impact categories.

Per kg slaughtered weight, there is a clear difference between the three types of meat, with beef having four to eight times larger environmental impacts than poultry and up to five times larger than pork. These differences are less pronounced when comparing the environmental impact intensity (impact per Euro spent) of the three types of meat, where pork has the lowest impact intensity for most of the environmental impact categories (down to 40 % of the impact of poultry and 23 % of the impact of beef).

The relevant environmental impacts related to imports into the EU are included in all these figures.

Improvement options

Improvement options have been identified in three main areas:

- Household improvements, mainly to reduce food losses (wastage) and to reduce car use for shopping;
- Agricultural improvements, mainly to reduce water and air emissions (in particular nitrate, ammonia and methane) as well as land use;
- Power savings in farming, food industry, retail, catering, and for household appliances.

More specifically the improvement options include:

- Planting catch crops during winter (to reduce nitrate leaching, N_2O and ammonia emissions);
- Improved growing practise and intensification of cereal production where yields are low today (to reduce land use and ammonia emissions);
- Optimised protein feeding in pig and dairy farming (to reduce ammonia emissions and nitrate leaching);
- Liquid manure pH reduction (to reduce ammonia emissions);
- Tightening the rules of manure application (to reduce nitrate leaching and N_2O emissions);
- Copper reduction in dairy cattle and pig diets (to reduce copper emissions);
- Methane-reducing diets for dairy cattle (to reduce methane emissions);
- Biogasification of manure from dairy cows and pigs (to reduce methane and N_2O emissions);
- Home delivery of groceries (to reduce greenhouse gas and other air emissions related to car driving);
- New cold appliances only A+ or A++ (to reduce electricity consumption);
- Power saving in farming, food industry, retail, and catering (to reduce electricity consumption);
- Household meal planning tools (to reduce food losses and all environmental interventions throughout the life cycle).

When all the identified environmental improvement potentials are taken together, the total improvement amounts to a reduction of 17 % for nature occupation, around 25 % for global warming and respiratory inorganics, 31 % for acidification and terrestrial eutrophication, 43 % for aquatic eutrophication, to 68 % for aquatic ecotoxicity (when rebound effects and synergies have been accounted for). Since the first three impact categories make up 95 % of the aggregated (monetarised) environmental impact, the aggregated improvement potential amounts only to about 20 % of the total environmental impact of meat and dairy products in EU-27 (and significantly less if rebound effects were not accounted for). Figure 1.2. shows how much the environmental impacts may be reduced for the main environmental impact categories.

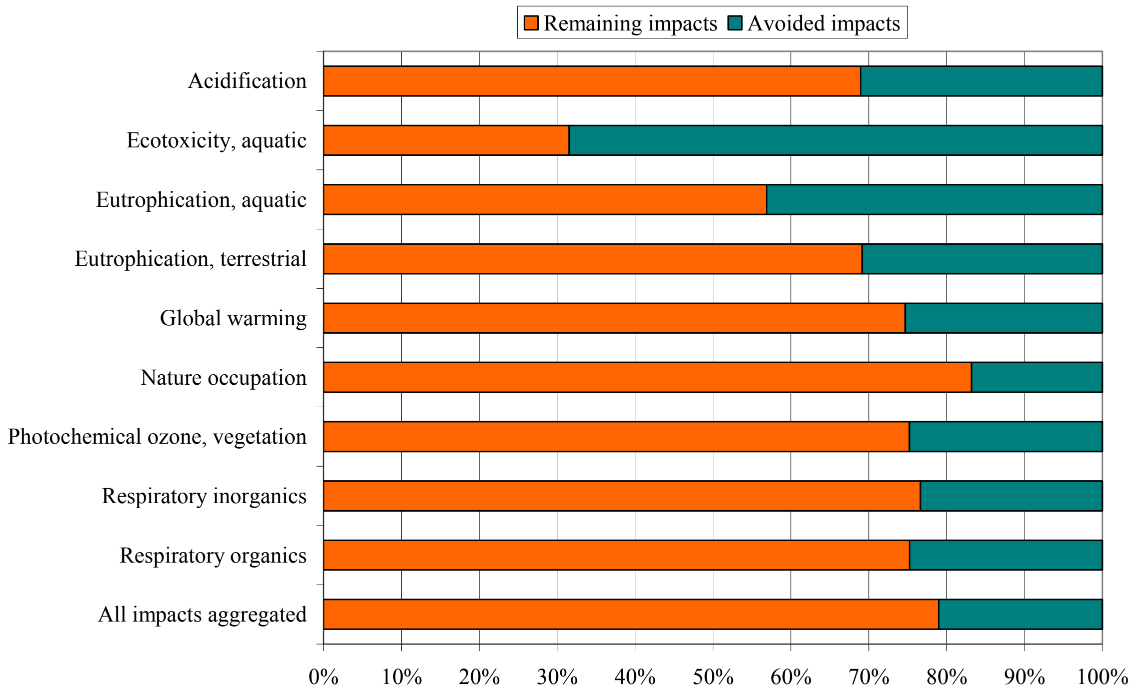

Figure 1.2: Remaining and avoided environmental impacts of meat and dairy products if all identified improvement options are implemented together. Rebound effects as well as synergies and dysergies between different improvement options are considered.

Noting that the aggregated impact from meat and dairy products amount to 24 % of the overall impact of EU-27 total final consumption, this implies that after all improvement options have been successfully implemented, the impact from meat and dairy products would still amount to 19 % of the aggregated impact of EU-27 total final consumption. This seems to suggest that large reductions in the overall impacts from meat and dairy products cannot be obtained from the identified improvement options alone, but would require targeting the level and mode of consumption as such. One of the proposed improvement options may be applicable also for this purpose, namely household meal planning tools.

Impact assessment

The impact assessment shows that the household improvement options have, in addition to relatively important environmental potentials, also possible socioeconomic benefits through household time saving and positive health effects. However, the uncertainties of the assessment are especially high for these types of improvement options.

The agricultural improvement options show a diverse picture, as some have direct economic benefits while others come at direct costs.

Without negative effects or even with direct economic benefits, aquatic eutrophication may be reduced by 30 % by tightening the regulation of manure application, area use for meat and dairy production may be reduced by nearly 5 % through intensification in cereal production, and respiratory inorganics and global warming impacts from meat and dairy production may be reduced by 4 % and 2.5 % respectively, through methane-reducing animal diets.

All other agricultural improvement options involve a trade-off between economic costs and environmental benefits. Although involving direct economic costs, there is an overall net benefit of more widespread use of catch crops, which may reduce aquatic eutrophication from meat and dairy production by nearly 4 %. Acidification and terrestrial eutrophication from meat and dairy production may be reduced by 8-9 % through optimised protein feeding and by 14-16 % through liquid manure pH reduction. Copper reduction in dairy cattle and pig diets may reduce aquatic toxicity from meat and dairy production by two thirds. Biogas plants for liquid manure treatment may reduce the global warming potential from meat and dairy production by nearly 5 %. Large uncertainties apply to the monetarisation of these impacts, so although these improvement options show a net benefit on average, this cannot be shown at 95 % confidence level.

Power savings have both economic and environmental benefits, although the environmental gains are relatively small compared to the other improvement options.

Possible policy targets and measures

Regarding household improvements, the study stresses that household decision-making and its environmental implications are largely unresearched areas, which imply that it is problematic to put up definite policy targets. It therefore suggests initiating comprehensive research in this area, covering issues like:

- The household decision-making processes with respect to diet choices, meal planning, food shopping, meal preparation and food waste; the actual behaviour, the rationales applied, attitudes and conceptual understanding, and the environmental and dietary health implications. The research should cover the relationships to different lifestyles, socioeconomic characteristics, and geographical differences.
- Logistics of shopping and food distribution, to determine the actual potential savings, including household time saving, the traffic rebound effect and the options for preventing this, as well as the extent to which additional shopping is induced by out-of-stock situations in the household.
- The options for improving household decision-making processes and/or their environmental impacts, e.g. through information campaigns and meal planning tools, and what characteristics are essential for their acceptance.
- Rebound effects of changes in household decision-making, e.g. how meal planning and home delivery of groceries affect household behaviour and time allocation, and what value the households assign to the different activities.

To avoid postponing the potentially substantial improvements in expectation of further research results, the study suggests that an important part of the above research be carried out in connection to pilot schemes seeking to implement some of the potential improvements in smaller geographical areas.

Furthermore, more appropriate consumer information may be an important tool to prevent food being discarded because of misconceptions about freshness, colour, texture, and food safety issues. To support this, it may be useful to perform a review of national legislations to identify possible technical barriers that increase food loss or hamper the implementation of technologies that improve shelf life, such as too tight requirements on ‘preferably consumed before’ dates, perverse measuring standards, and demands for what may be labelled ‘fresh’.

For the agricultural improvement options, the following policy targets are implicit in the calculated improvement potentials:

- To reduce nitrate leaching from animal manure to an average 0.64 kg N per pig and 6.3 kg N per Mg milk produced, and from cereal production to an average 6.5 kg N per 1 000 kg cereal (53 %, 39 % and 90 % of current levels, respectively).
- To increase cereal yields to an average 4 500 kg per hectare, thus reducing the area requirement.
- To reduce ammonia emissions from pig and dairy farming to an average 0.72 kg per pig and 5 kg per 1 000 kg milk produced (43 and 69 % of current levels).
- To reduce Cu emissions to soil from pig and dairy farming to an average 2.3 g per pig and 10 g per 1 000 kg milk produced (21 and 44 % of current levels).
- To reduce methane emissions from dairy cows and animal manure to an average 5.3 kg per pig and 18 kg per 1 000 kg milk produced, partly by ensuring that 50 % of all manure from pig and dairy farms is utilised for biogas production.

These targets are expressed relative to the quantities produced, implying that the corresponding absolute emission targets would change with changes in production volume. In this way, the policy targets would not have to be revised if the production volume changes.

Examples of possible measures for achieving such targets include the provision of informational or management tools for dairy farms and integrating regard for methane emissions into the regular feed optimisation procedures. Furthermore, 'license-to-operate' requirements might be placed on agricultural enterprises above a certain size (in terms of acreage and animal units). The study suggests investigating further the different possible measures.

Regarding power savings, the study suggests as a realistic policy target for the household cold appliances that all new appliances sold in Europe be either A+ or A++, as soon as the European industry can supply these. As measures the study suggests:

- The establishment of a standard procedure for calculating lifetime costs of appliances, and adding to the energy-labelling requirements that the lifetime costs of appliances shall be presented alongside the appliance price in the same letter size.
- A European-wide scheme where the consumers can buy A++ appliances at approximately the same price as an average appliance in exchange for returning an old appliance.
- As an alternative, the energy requirements for new appliances could be enforced by direct regulation.

Methodology

The methodology applied in this study is a hybrid life cycle assessment method, which implies a system model that combines the completeness of 'top-down' input-output matrices, based on national accounting statistics combined with national emission statistics (known as NAMEA matrices), with the detailed modelling of 'bottom-up' processes from process-based life cycle assessments. To represent the livestock production in a way that allows to model different improvement options, a range of production systems were modelled, based on well-documented biological input-output relations, such as nutrient balances. These production systems have then been scaled to the level of EU-27, and fitted to the production volume, area, and number of livestock given by Faostat.

For the impact assessment, a flexible model is used that allows results to be presented both in 15 environmental midpoint indicators (global warming potentials, photochemical ozone creation potential, etc.) and in monetary units (Euro). Specifically for this project, a damage model for aquatic eutrophication was developed, since this has until now been missing.

Limitations and uncertainties

The main limitations and uncertainties affecting the assessment of the different improvement options include:

- For the majority of the improvement options, the overall uncertainty on the environmental improvement is dominated by the assumption of the degree to which the improvement option can be implemented, i.e. the area for which catch crops can be implemented, the actual cereal yields that can be achieved, the level of reduction in emissions, the extent of the power saving, and the extent that household behaviour can be affected.
- For improvement options involving large changes in direct production costs, the uncertainty on the cost estimates may contribute significantly to the overall uncertainty. This is particularly the case for cold appliances regulation and for biogasification of liquid manure; the latter also being particularly sensitive to the rate of temporal discounting.
- For some improvement options, the uncertainty on the socioeconomic impacts dominates the overall uncertainty. This is particularly the case for home delivery of groceries and meal planning tools.

Most improvement options show a net benefit at the 95 % confidence level, but due to the large uncertainties in the characterisation factors, this is not the case for the four agricultural improvement options with the largest direct economic costs: optimised protein feeding, liquid manure pH reduction, copper reduction in dairy cattle and pig diets, and liquid manure biogasification. Particularly the benefit of copper reduction is uncertain, since it depends on the impact potential of metal emissions, which may be overestimated in current characterisation models.

A number of impacts have been entirely omitted from the study (impacts from occupation of extensive grazing lands, disruption of archaeological heritage, antibiotic resistance, species dispersal, noise, pesticides transmitted through treated food, depletion of phosphate mineral resources), some have been modelled only very coarsely (all area uses treated equally, despite large differences in biological value) and some have been only qualitatively touched upon (erosion and water balance). These short-comings are likely to mainly bias the results towards a smaller overall impact and smaller overall improvement potentials relative to the result if these impacts had been quantified. It is not expected that inclusion of these impacts would change the overall conclusions.

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PART I: MAIN REPORT

1 INTRODUCTION

1.1 Background

The Communication on Integrated Product Policy (COM(2003) 302 final), announced that the European Commission would seek to identify and stimulate action on products with the greatest potential for environmental improvement. This work had been scheduled into three phases:

- the first phase consisting of research to identify the products with the greatest environmental impact from a life cycle perspective consumed in the EU;
- the second phase which consists in the identification of possible ways to reduce the life cycle environmental impacts of some of the products with the greatest environmental impact;
- in the third phase the European Commission will seek to address policy measures for the products that are identified as having the greatest potential for environmental improvement at least socioeconomic cost.

The first phase was completed in May 2006 with the EIPRO study led by the IPTS (JRC) in cooperation with ESTO research network organisations. The study identified the products consumed in the EU having the greatest environmental impact from a life cycle perspective. In that project, the final consumption had been grouped into almost 300 product categories and assessed in relation to different environmental impact categories, such as acidification, global warming, ozone depletion, etc.

The study showed that groups of products from only three areas of consumption – food and drink, private transportation, and housing – are together responsible for 70-80 % of the environmental impacts of private consumption and account for some 60 % of consumption expenditure.

The EIPRO project conclusions thus suggested initiating the second phase of the work scheduled in the Integrated Product Policy (IPP) communication on these three groups of products. To this end, three parallel projects were launched late 2005-early 2006, coordinated by the IPTS. These projects deal with the Environmental IMprovement of PROducts (IMPRO, respectively IMPRO-car, IMPRO-meat and dairy, IMPRO-buildings).

This is the final report of the IMPRO-meat and dairy project. The project was commissioned by the IPTS and carried out by 2.-0 LCA consultants, with the University of Aarhus, Faculty of Agricultural Sciences (formerly The Danish Institute of Agricultural Sciences) and the Technical University of Denmark as subcontractors. The project was carried out in the period 1 August 2006 to 19 February 2008.

1.2 Objectives

The objectives of the study are:

1. To estimate and compare the environmental impact potentials of meat and dairy products consumed in EU-27, taking into account the entire value chain (life cycle) of these products.

2. To identify and estimate the size of the main environmental improvement options for the products mentioned under 1.
3. To assess the socioeconomic impact potentials of the improvement options mentioned under 2.

The target groups of the study are public authorities and other stakeholders involved in European environmental and agro-industrial policy-making.

1.3 Reading guide

The report gives detailed account of the project results as well as of the methodologies and data used. Readers who are mainly interested in the results of the study are especially referred to Part I (Main Report, Chapters 1-7). The methodologies and data used are documented in Part II (Methodology, Chapters 8-10). Additional details are presented in the annexes (Chapters 12-15).

2 SCOPE AND METHODOLOGY

2.1 Overview

The study aims to quantify the environmental impacts related to meat and dairy products, to identify improvement options, to quantify their potentials to reduce the environmental impacts, and to assess the socioeconomic impacts and feasibility of implementing the improvement options.

The report approached these tasks in the following steps:

- Definition of the scope of the analysis and of the terminology. This is described further on in this chapter.

- Establishment of a system model of the production and consumption of meat and dairy products, including the associated environmental interventions (emissions, use of natural resources). This model is needed as a basis for quantifying the environmental impacts of the different product types and of the contributing processes throughout the food chain (or the life cycle). It is also needed to quantify the changes in environmental impacts when improvement options are implemented. In LCA terminology, this model represents the life cycle inventory. The general principles of the model are described further on in this chapter, and the details are provided in Chapter 8 and Annex I.

- Choice and definition of environmental impact assessment methods. The life cycle inventory delivers information on emissions of different types of substances to air, water and soil, as well as on the amounts of natural resources used (land use, extraction of minerals, etc.). Per se these do not describe or quantify environmental impacts. The environmental impact assessment methods allows to quantify the impacts of the environmental interventions in terms of different impact categories (such as acidification or global warming) and to aggregate the impacts according to different categories into a limited set of high level indicators or even a single impact score. The details of the impact assessment methods used are described in Chapter 9 and Annexes II-IV of the report.

- Calculation of the current environmental impacts of meat and dairy products. This is done by applying the system model and impact assessment methods described above. The results are given for all meat and dairy products together, separately for the main product types (beef, pork, poultry and dairy products), and for the main contributing processes throughout the life cycles. The results are presented at three levels of aggregating environmental impacts (15 midpoint impact categories, three end-point or damage categories, and as single aggregated scores). The environmental impacts are also represented as external environmental costs in terms of euro. The results on the environmental impacts of meat and dairy products are presented in Chapter 3.

- Identification of environmental improvement options and quantification of environmental improvement potentials. To focus the research efforts, improvement options are investigated only for those processes that currently contribute more than 10 % to the environmental impacts of all meat and dairy products for at least one of the disaggregated impact categories. Only options with proven technological feasibility and that can be implemented in the short to medium term are included. The improvement options are determined on the basis of current variation between technologies and by relying on expert knowledge in different forms (including from published technological studies, technology roadmaps, action plans, expert consultations, etc.).

The improvement potential of each option is quantified as changes in the yearly environmental impacts of meat and dairy products as a consequence of full implementation of the improvement option. The implementation of the improvement options is modelled as changes in the average yearly inputs and outputs of the processes compared to the current situation. Changes in environmental impacts are modelled both with and without different types of rebound effects. Results are presented for each improvement option individually, and for improvement options combined (including synergies and dysergies). Furthermore, comparisons are made to assumed autonomous developments.

The methodology for identifying and quantifying improvement options is described in Chapter 4.1. The individual improvement options, how they are modelled, the data used, and the size of the environmental improvement potentials are presented in Chapter 4.2-4.6. Combined effects of several improvements and rebound effects are discussed in Chapter 4.7, and the results of improvement potentials of all investigated improvement options are summarised in Chapter 4.8.

- Socioeconomic impact assessment. The purpose is to allow an overall judgement on the desirability of implementing an environmental improvement option, including the trade-offs in changes of environmental impacts and the other socioeconomic impacts (economic costs, dietary health, etc.). The first step of the assessment is a qualitative screening to identify which of a wide range of socioeconomic parameters are likely to be affected by each improvement option, resulting in a matrix of improvement options and socioeconomic indicators with scores of none-low-medium-high relevance. For combinations of indicators and improvement options assigned high or medium relevance, the indicator values are quantified. Finally, the different types of environmental and socioeconomic costs and benefits, expressed in euro, are compared for each of the improvement options. The socioeconomic impact assessment methodology and data are presented in detail in Chapter 10 and the impact assessment results in Chapter 5.

- Policy alignment analysis, feasibility analysis and assessment of policy instruments. The different existing policies relevant to the improvement options are discussed in Chapter 6.1. Chapter 6.2 assesses the feasibility of implementing the improvement options and makes suggestions on possible further measures for supporting the implementation of the improvements.

- Final conclusions and recommendations of the study are presented in Chapter 7.

- The main limitations and uncertainties that may influence the study results are analysed and, as far as possible, quantified throughout the report.

2.2 Main model characteristics and data sources

A new system model of the production and consumption of meat and dairy products has been developed for this study. It distinguishes 110 processes and quantifies for each of the processes the production outputs, the environmental interventions (such as emissions or use of natural resources), and the inputs received from the other processes. Among the processes included in the model there are 15 agricultural processes (including different livestock production systems as well as feed production systems), 20 food and feed industry sectors, four household processes (such as food storage and cooking) and seven waste management processes.

These specific processes are embedded into the framework of environmentally extended input-output matrices for EU-27 (known as NAMEA matrices). The NAMEA matrices have been obtained by combining national accounting statistics with national emission statistics.

The resulting system model is used in the study to calculate the environmental interventions caused by the different types of meat and dairy products, as well as by the different contributing processes (current situation). It is also used to model the changes in the environmental interventions by implementing the improvement options. Finally the system model serves also for the quantification of costs and benefits in the socioeconomic impact assessment of improvement options.

In LCA terminology, the matrices of the described system model represent the life cycle inventory. The method of combining 'top-down' input-output matrices, based on national accounts and emission statistics, with the detailed modelling of 'bottom-up' processes from traditional process-based life cycle assessments is often called hybrid life cycle assessment.

The data for the agricultural processes are derived from detailed production models, including all relevant inputs and outputs. For example, for each of the five dairy farming systems the production model includes the specifications for different types of land use, herd composition, input of different types of feed, production output (milk, beef, cereal surplus), fertiliser application and nitrogen balance. Well-documented biological input-output relations, such as nutrient balances, have been used to specify the agricultural production models. Data on production volume, area, number of livestock by Faostat have been used to scale the production models up to the level of EU-27.

Also the food-specific transport processes, the household processes, the fertilizer production, the waste management processes and some others are defined using specific process data such as from life cycle inventory databases.

The data for the remaining processes, including the food and feed industry processes, energy production, non-food-specific transport, machinery and other equipment, the different types of services, etc., come directly from the EU-27 NAMEA matrices.

The model design and data used are described in detail in Chapter 8.

2.3 Scope and functional unit

The study covers meat and dairy products consumed in EU-27, and the entire life cycle of these products, whether inside or outside EU-27. (This means the environmental impacts of the relevant imports are included.)

The database for the project (see Chapter 8.2) covers the entire economy. Therefore, the system boundary for the inventory is identical to the boundary between the technosphere and nature, and it has not been necessary to apply any cut-off rules. The boundary between the analysed system and the rest of the technosphere is implicitly defined by the reference flows provided in Table 2.1.

The study is based on data from the most recent year for which consistent data are available. For consumption of meat and dairy products in EU-27, this is 2004. The base year for data on production and household processes and their environmental exchanges is 2000, which is the most recent year for which statistical data were consistently available for 98 % of the production in EU-27.

Emissions from processes are assumed to occur at present time without discounting future impacts. The implications of applying discounting are treated in a separate discussion in Chapter 5.4.

For improvement options, modern technology (BAT) is applied, as well as existing new technologies that are expected to be introduced during the next five years (i.e. before 2012) and implemented before 2020.

The functional unit of the study is the entire annual consumption of meat and dairy products in EU-27. Meat and dairy products are defined as those commodities covered by COICOP ([1]) categories 01.1.2 and 01.1.4, with the exception of eggs.

The total consumption includes both household preparation of meals and catering meals prepared in restaurants and in institutions such as hospitals.

The corresponding reference flows are listed in Table 2.1.

Rebound effects, e.g. derived changes in production and consumption when the implementation of an improvement option liberates or binds a scarce production or consumption factor, as well as synergies or dysergies when simultaneously introducing several improvement options, are separately treated in Chapter 4.7.

([1]) See http://data.un.org/unsd/cr/registry/regcs.asp?Cl=5&Lg=1&Co=01.1

Table 2.1: Reference flows of the annual consumption of meat and dairy products in EU-27.

Product/process	Unit	Amount	Amount per capita	More details in Chapter
Beef and beef products	kg meat (slaught. weight)	6.42E+09	14	8.1.
Dairy products	kg raw-milk-equivalent	1.14E+11	237	8.1.
Pork and pork products	kg meat (slaught. weight)	1.53E+10	32	8.1.
Poultry and poultry products	kg meat (slaught. weight)	9.26E+09	19	8.1.
Restaurant/catering, not incl. food	EUR2000	6.12E+10	127	8.4.7.
Car purchase and driving, for shopping	vehicle-km	8.04E+10	187	8.4.9.
Public transport by road, for shopping	EUR_{2000}	1.90E+09	4	8.4.9.
Storage of food in household	EUR_{2000}	8.91E+09	18	8.4.10.
Cooking in household	EUR_{2000}	6.14E+09	13	8.4.10.
Dishwashing in household	EUR_{2000}	4.54E+09	9	8.4.10.
Tableware & household utensils	EUR_{2000}	8.95E+09	19	8.4.10.
Waste treatment – Food	kg food waste	9.30E+09	19	8.4.12.
Waste treatment – Meat packaging	kg packaging	4.66E+09	10	8.4.12.
Waste treatment – Dairy packaging	kg packaging	3.43E+09	7	8.4.12.

2.4 Terminology

Country abbreviations follow the ISO 2-digit standard.

Currencies: In ISO three-digit code. Most data refer to EUR for the currency Euro. As currencies change values over time, it is often necessary to apply a subscript to indicate the year that the currency refers to, e.g. EUR_{2000} or EUR_{2003}. When no specific mention is given, EUR should be assumed to refer to EUR_{2000}.

'Environment' and 'environmental': In this study, these terms are applied in their narrow sense, i.e. covering only impacts from biophysical stressors, not impacts from social and economic stressors, since these impacts are reported in a separate socioeconomic assessment.

IO: Input-Output. Usually used to designate the national accounting matrices constructed by combining industry supply- and use-tables.

ISO: International Organisation of Standardisation. Reference to individual standards are given by numbers, e.g. ISO 14040.

NAMEA: National Accounting Matrices with Environmental Accounts. The extension of the national input-output tables with data on environmental exchanges per industry.

No: Number.

Number format: Decimal point. Scientific notation with the use of E+ to signify the power of 10, e.g. E+02 = 10^2 = 100. This has been chosen to make it easier to transfer data from the tables of the report directly to common spreadsheets.

Units: As far as practical, SI-units have been applied, with the SI-prefixes shown in Table 2.2. An exception is the traditional area measure ar (a), as in hectar (ha), which should not be confused with the SI-prefix atto- or the popular abbreviation for year. Popular units have also been applied for time (year, month, week, day, hour), written out fully, since multiples of the SI-unit seconds (s) appears awkward. Also, the impact indicators have particular units, which are explained in Chapter 9.1.

Weight: All weights have been expressed in the ISO-unit gram (g) with the SI-prefixes.

Table 2.2: SI-prefixes used in this report.

P	peta-	1.0E+15
T	tera-	1.0E+12
G	giga-	1.0E+9
M	mega-	1.0E+6
k	kilo-	1.0E+3
h	hecto-	1.0E+2

3 ENVIRONMENTAL IMPACTS OF MEAT AND DAIRY PRODUCTS – RESULTS

In this study the environmental impacts of meat and dairy products are calculated at three levels: 15 midpoint impact categories, three endpoint impact categories, and a single overall impact value. The results at midpoint level are presented in Chapter 3.1. The results at endpoint level and as single overall values are presented in Chapter 3.2. Chapter 3.3 identifies which product groups and processes in the food chain contribute most to the environmental impacts. The impact categories at the different levels and the characterisation models used to calculate the impact indicators for the different categories are described in more detail in Chapter 9.1.

3.1 Midpoint results for meat and dairy products

The total environmental impact for the full functional unit (i.e. the total annual consumption of meat and dairy products in EU-27) for each of the environmental midpoint impact categories is shown in Table 3.1. The table presents both characterised results, i.e. in the units of the reference substances for each impact category, and normalised results, i.e. relative to the environmental impact caused by the total annual final consumption in EU-27.

The coefficient of variation is determined from the uncertainty of the statistical data combined with the uncertainty of the midpoint characterisation models (as given in Table 9.4). Since these uncertainties also apply to the normalisation reference, the data on the relative importance of meat and dairy products are not affected.

Table 3.1: Environmental impact of the total annual consumption of meat and dairy products in EU-27 (the functional unit of the study) expressed in the specific units of each impact category and relative to the impact of EU-27 total final consumption.

Impact category	Unit	Amount	Coefficient of variation	Relative to the impact of EU-27 total final consumption (%)
Acidification	m^2 UES	9.49E+10	0.9	24.9
Ecotoxicity, aquatic	kg-eq. TEG water	1.43E+14	2.7	46.6
Ecotoxicity, terrestrial	kg-eq. TEG soil	6.03E+11	2.7	6.5
Eutrophication, aquatic	kg NO_3-eq.	8.86E+09	0.2	29.4
Eutrophication, terrestrial	m^2 UES	3.88E+11	1.2	39.1
Global warming	kg CO_2-eq.	6.69E+11	0.1	14.2
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	1.38E+09	2.7	8.0
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	1.14E+09	2.7	6.7
Mineral extraction	MJ extra	5.26E+09	1.6	5.8
Nature occupation	m^2 arable land	9.76E+11	1.1	35.8
Non-renewable energy	MJ primary	8.76E+12	0.1	6.3
Ozone layer depletion	kg CFC-11-eq.	1.91E+05	1.5	6.4
Photochemical ozone, vegetation	m^2*ppm*hours	6.66E+12	1.7	12.4
Respiratory inorganics	kg PM2.5-eq.	8.51E+08	1.9	17.7
Respiratory organics	person*ppm*hours	7.22E+08	2.2	12.8

For comparison to the relative impacts from Table 3.1, it may be interesting to note that the consumption of meat and dairy products constitutes 6.1 % of the economic value of the total final consumption in EU-27. For many impact categories, meat and dairy products thus have larger impact intensities than an average product. On the other hand, it can be noted that compared to other products, meat and dairy products do not contribute particularly to terrestrial ecotoxicity, human toxicity, ozone layer depletion and extraction of minerals and non-renewable energy resources, and that it therefore may not be particularly relevant to look for improvement potentials for these impact categories.

The main product groups and processes that contribute to the results in Table 3.1 are analysed in Chapter 3.3.

3.2 Endpoint results for meat and dairy products

Here, the total environmental impact of the full functional unit (i.e. the total annual consumption of meat and dairy products in EU-27) is expressed as aggregated values for each of the three damage categories (see Table 3.2), as well as aggregated across all damage categories (see Table 3.3). The endpoint results were obtained by applying the characterisation factors from Table 9.1 to the midpoint results from Table 3.1.

Table 3.2: Environmental impact of the total annual consumption of meat and dairy products in EU-27 (the functional unit of the study) expressed in the units of each damage category as well as in GEUR (1 000 000 000 EUR).

Impact category	Impact on ecosystems		Impacts on human well-being		Impacts on resource productivity
	Species-weighted m^2* years	GEUR	QALY	GEUR	GEUR
Acidification	5.19E+09	0.73			
Ecotoxicity, aquatic	7.19E+09	1.01			
Ecotoxicity, terrestrial	4.77E+09	0.67			
Eutrophication, aquatic	6.38E+09	0.90			
Eutrophication, terrestrial	3.44E+10	4.83			
Global warming	3.89E+11	54.7	14 112	1.04	-0.24
Human toxicity, carcinogens			3 877	0.29	0.089
Human toxicity, non-carcinogens			3 195	0.24	0.073
Mineral extraction					0.021
Nature occupation	8.59E+11	121			
Ozone layer depletion			201	0.01	0.0046
Photochemical ozone, vegetat.	4.39E+09	0.62			1.87
Respiratory inorganics			595 407	44.06	13.69
Respiratory organics			1 907	0.14	0.044
Total impact	1.31E+12	185	618 698	45.8	15.5
Coefficient of variation	0.8	1.5	3.1	3.3	3.3
Impact relative to the impact of EU-27 total final consumption	24 %		17 %		17 %

Table 3.3 also shows the aggregated results if the average Ecoindicator 99 weights from Goedkoop & Spriensma (1999) are applied instead of the characterisation factors of Table 9.1. The Ecoindicator weighting (40 % to impacts on ecosystems, 40 % to impacts on human well-being, 20 % to impacts on resource productivity) has been applied to the normalised endpoint results from Table 3.2, using the endpoint

normalisation references from Table 9.3. The results with Ecoindicator99 weighting gives less weight to impacts on ecosystems, more weight to impacts on human health and more than double the weight to impacts on resource productivity, compared to the monetarisation.

Table 3.3: Aggregated environmental impact of the total annual consumption of meat and dairy products in EU-27 (the functional unit of the study) expressed as a monetarised value in GEUR (1 000 000 000 EUR) and using the Ecoindicator99 weights. The overall aggregated value can also be expressed in QALYs by dividing the values in the second column by the conversion factor 74 000 EUR/QALY.

Impact category	All impacts aggregated			
	GEUR	Contribution (%)	with Ecoindicator99 weights	Contribution (%)
Acidification	0.73	0.3	3.8E-04	0.2
Ecotoxicity, aquatic	1.0	0.4	5.3E-04	0.3
Ecotoxicity, terrestrial	0.67	0.3	3.5E-04	0.2
Eutrophication, aquatic	0.90	0.4	4.7E-04	0.2
Eutrophication, terrestrial	4.8	2.0	2.5E-03	1.3
Global warming	56	22.6	3.0E-02	15.0
Human toxicity, carcinogens	0.38	0.2	6.2E-04	0.3
Human toxicity, non-carcin.	0.31	0.1	5.1E-04	0.3
Mineral extraction	0.02	0.01	4.5E-05	0.02
Nature occupation	120	49.2	6.4E-02	31.9
Ozone layer depletion	0.02	0.01	3.2E-05	0.02
Photochemical ozone, veg.	2.5	1.0	4.3E-03	2.2
Respiratory inorganics	58	23.5	9.5E-02	48.0
Respiratory organics	0.18	0.1	3.1E-04	0.2
Total	250	100.0	2.0E-01	100.0
Relative to the impact of EU-27 total final consumption	22 %		20 %	

Table 3.3 confirms the notion from Table 3.1 that terrestrial ecotoxicity, human toxicity, ozone layer depletion and extraction of minerals are relatively unimportant impact categories for meat and dairy products.

The aggregated result is dominated by three impact categories: nature occupation (32-49 %), respiratory inorganics (24-48 %) and global warming (15-23 %). The agricultural land area that contributes to 'nature occupation' is only that intensively used, either for fodder crops or intensive grazing. Out of the total European pasture area of more than 75 million ha, the livestock model used only includes 21.6 million ha. Thus, it is this area that contributes to the 32-49 % of the total damages in Table 3.3. At least 54 million ha extensive grazing lands are *not* included in this study, i.e. it is regarded as being maintained as extensive grazing land independent of the demand for meat and dairy products.

3.3 Main contributing product groups and processes

3.3.1 Product groups

Table 3.4 provides a breakdown of the environmental impact results on the four main product groups (dairy products, beef, pork and poultry) as absolute values. Table 3.5 and Table 3.6 provide the results per weight unit and per EUR consumption expenditure. The uncertainty on these breakdowns is estimated to be less than +/- 10 % of the values shown due to a high degree of co-variation, while the absolute uncertainty on these values are the same as for the overall results in Table 3.1 and Table 3.2.

Table 3.4: Relative contribution (%) of the four main product groups.

Impact category	Dairy products	Beef	Pork	Poultry	All four product groups
Midpoint categories:					
Acidification	36	29	25	10	100
Ecotoxicity, aquatic	36	16	44	5	100
Ecotoxicity, terrestrial	39	20	33	8	100
Eutrophication, aquatic	40	24	28	8	100
Eutrophication, terrestrial	36	31	24	10	100
Global warming	41	28	26	5	100
Human toxicity, carcinogens	36	29	26	9	100
Human toxicity, non-carcinogens	39	21	32	8	100
Mineral extraction	40	19	34	7	100
Nature occupation	33	39	19	9	100
Non-renewable energy	39	20	34	7	100
Ozone layer depletion	39	23	28	9	100
Photochemical ozone, vegetation	39	28	28	5	100
Respiratory inorganics	36	31	23	9	100
Respiratory organics	39	28	28	5	100
Endpoint (damage) categories:					
Impact on ecosystems	36	35	22	8	100
Impacts on human well-being	36	31	23	9	100
Impacts on resource productivity	36	31	24	9	100
All impacts	36	34	22	8	100

Table 3.5: Impact per weight unit for the four main product groups.

Impact category	Unit	Dairy products	Beef	Pork	Poultry
		per kg raw milk equivalent	**per kg slaught. weight**	**per kg slaught. weight**	**per kg slaught. weight**
Midpoint categories:					
Acidification	m^2 UES	0.30	4.32	1.55	0.98
Ecotoxicity, aquatic	kg-eq. TEG water	447	3471	4073	815
Ecotoxicity, terrestrial	kg-eq. TEG soil	2.1	18.9	12.8	5.2
Eutrophication, aquatic	kg NO_3-eq.	0.031	0.325	0.164	0.075
Eutrophication, terrestrial	m^2 UES	1.2	18.6	6.0	4.1
Global warming	kg CO_2-eq.	2.4	28.7	11.2	3.6
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	0.004	0.062	0.023	0.014
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	0.004	0.037	0.023	0.010
Mineral extraction	MJ extra	0.018	0.153	0.117	0.042
Nature occupation	m^2 arable land	2.8	58.9	12.2	9.5
Non-renewable energy	MJ primary	30	276	193	65
Ozone layer depletion	kg CFC-11-eq.	6.5E-07	7.0E-06	3.5E-06	1.8E-06
Photochemical ozone, vegetation	m^2*ppm*hours	23	288	121	37
Respiratory inorganics	kg PM2.5-eq.	0.0027	0.0417	0.0127	0.0086
Respiratory organics	person*ppm*hours	0.0025	0.0318	0.0129	0.0038
Endpoint (damage) categories:		0	0	0	0
Impact on ecosystems	Species-weighted m2*years	4.1	71	18	11
Impacts on human well-being	QALY	2.0E-06	3.0E-05	9.3E-06	6.2E-06
Impacts on resource productivity	EUR	0.05	0.75	0.24	0.15
All impacts	EUR	0.77	13.00	3.52	2.16

Table 3.5 shows that per kg slaughtered weight, there is a clear difference between the three types of meat, with beef having a significantly larger environmental impact, and poultry having the smallest.

These differences are less pronounced in Table 3.6, showing the environmental impact intensity (impact per EUR spent), where pork appears generally to have the smallest impact intensity.

From Table 3.6, it can also be seen that the monetarised externalities (the row 'all impacts') are of considerable size compared to the private costs of the products (34 - 112 % of the private costs). The large uncertainty on the monetarisation implies that this proportion can be an order of magnitude smaller or larger.

Table 3.6: Impact per EUR consumption expenditure for the four main product groups. Note that consumption expenditure includes all life cycle costs, i.e. also costs for shopping and meal preparation, and thus more than just the price of the products.

Impact category	**Unit**	**Dairy products**	**Beef**	**Pork**	**Poultry**
Midpoint categories:					
Acidification	m^2 UES	0.21	0.37	0.15	0.30
Ecotoxicity, aquatic	kg-eq. TEG water	305	298	389	252
Ecotoxicity, terrestrial	kg-eq. TEG soil	1.4	1.6	1.2	1.6
Eutrophication, aquatic	kg NO_3-eq.	0.021	0.028	0.016	0.023
Eutrophication, terrestrial	m^2 UES	0.83	1.60	0.57	1.27
Global warming	kg CO_2-eq.	1.65	2.47	1.07	1.12
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	0.0030	0.0053	0.0022	0.0043
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	0.0026	0.0032	0.0022	0.0031
Mineral extraction	MJ extra	0.013	0.013	0.011	0.013
Nature occupation	m^2 arable land	1.94	5.06	1.16	2.93
Non-renewable energy	MJ primary	20	24	18	20
Ozone layer depletion	kg CFC-11-eq.	4.5E-07	6.0E-07	3.4E-07	5.7E-07
Photochemical ozone, vegetation	m^2*ppm*hours	15	25	12	11
Respiratory inorganics	kg PM2.5-eq.	0.0018	0.0036	0.0012	0.0027
Respiratory organics	person*ppm*hours	0.0017	0.0027	0.0012	0.0012
Endpoint (damage) categories:					
Impact on ecosystems	Species-weighted m^2*years	2.8	6.1	1.8	3.4
Impacts on human well-being	QALY	1.3E-06	2.6E-06	8.9E-07	1.9E-06
Impacts on resource productivity	EUR	0.034	0.064	0.023	0.046
All impacts	EUR	0.53	1.12	0.34	0.67

3.3.2 Processes

Figure 3.1 shows all processes that contribute with more than 10 % of the total environmental impact per midpoint impact category, as well as the relative impact of poultry farming, the food industry and transport processes.

For ozone layer depletion, not shown in Figure 3.1, the only remaining contribution in Europe is related to industrial cooling equipment, and regulation is already in place to eliminate this contribution. This issue shall therefore not be further discussed in this report.

For non-renewable energy and mineral extraction, not shown in Figure 3.1, the processes involved are, of course, the mineral extraction processes, which is not so relevant from an improvement perspective. The processes of interest are rather those that use energy and minerals, and where options for reducing consumption can be identified. The processes that use energy can roughly be assessed as the same as those contributing to global warming. The processes with a large mineral use are those using large amounts of machinery and construction materials, with a relatively low capacity utilisation, notably intensive animal husbandry, retail trade, private cars, and household equipment. Also for terrestrial ecotoxicity, not shown in Figure 3.1, the main contribution is from metal mining and processing, for use in machinery and equipment.

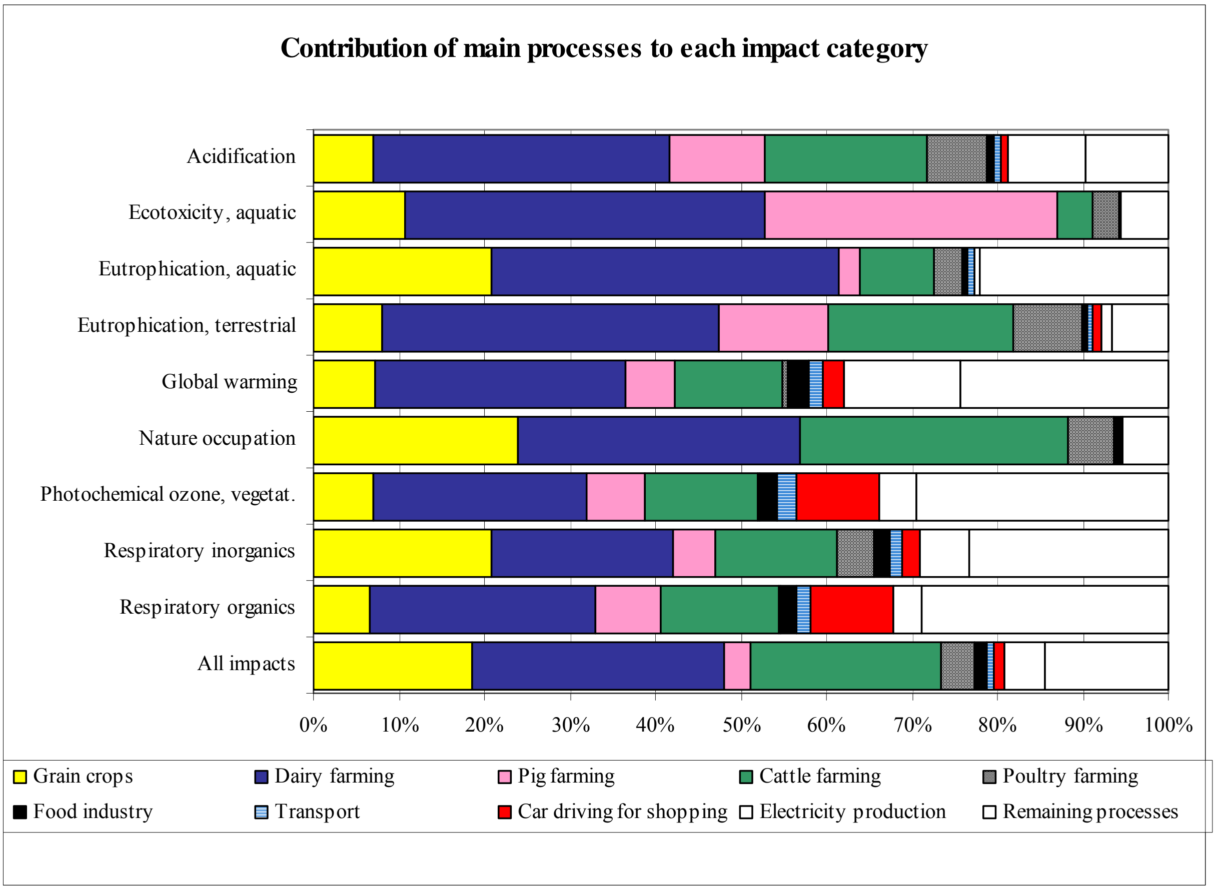

Figure 3.1: **Percentage contribution of main processes to each impact category. Terrestrial ecotoxicity, human toxicity, ozone layer depletion and mineral extraction not included in the figure, but included in the aggregated score for all impacts.**

Likewise, also for the other impact categories it may be relevant to look at options for reducing consumption of the output from the main contributing processes, i.e. to include in the focus the processes that use foods and electricity. Losses of food in household processes cause a significant increase in impact throughout the product life cycle. For electricity, the consuming processes are specified in Table 3.7. The total electricity consumption for meat and dairy products is 200 TWh annually. Storage in the household especially stands out as important. The data in Table 3.7 have not been corrected for differences in electricity price between household and industries. Such a correction would imply that values for household and retail consumption should be reduced by 14 % and values for industry consumption should be increased by 27 %.

Table 3.7: **Main electricity consuming processes for meat and dairy products. In % of total electricity consumption for meat and dairy products.**

	Direct	Farming	Food industry	Retail	Other	Sum
Storage of food in the household	23.4			0.1	0.6	24
Dairy products		4.1	7.5	1.7	6.2	19
Pork and pork products		1.8	7.9	2.2	7.0	19
Dishwashing in household	7.9			0.04	0.5	8.4
Restaurants and other catering, not incl. food	6.4			0.1	1.3	7.8
Beef and beef products		0.8	1.9	1.0	3.6	7.2
Cooking in household	7.1			0.1	0.7	7.8
Poultry and poultry products		0.7	0.7	0.4	1.3	3.1
Other					3.3	3.3
Sum	45	7.3	18	5.5	24	100

In their recent review of life cycle assessments of food products, Foster et al. (2006) conclude that packaging may contribute significantly to the primary energy consumption of dairy products, with the example that one-way glass bottles for milk can use as much energy as for the primary production of the milk they contain. Fortunately, this is an extreme example. The average dairy product is provided in laminated cartons, paper or plastics (see Chapter 8.4.12 for a breakdown on packaging types from the Danish packaging statistics). Also, for the average dairy product, packaging plays a smaller role than for liquid milk. From the available data, the total inflows to the European dairies from the paper, plastics, and non-metal mineral industries can be calculated, which would be an upper estimate for the dairies use of paper, plastics and glass packaging, respectively. These three inflows do not contribute more than 1 %, 0.9 % and 0.2 %, respectively, to the total primary energy use of the life cycle of the dairy products.

Based on the discussion above, it has been identified as relevant to suggest improvement options for the following seven groups of processes:

- Cereal crops
- Dairy farms
- Cattle farms
- Pig farms
- Retailing and shopping
- Electricity
- Household processes, notably storage and food losses.

This list includes only the processes that contribute more than 10 % to the total environmental impact per midpoint impact category. While this implies that specific improvement options will not be suggested for protein feed crops, the improvement options for cereal crops will also largely apply to protein crops.

Improvement options regarding food losses are suggested for food loss in the households only, mainly because there are already significant economic incentives to keep agricultural and industrial losses at a minimum, and because losses are more important the further to the end of the life cycle they occur.

3.4 Discussion of geographical variation within EU-27

Data variation between countries involves a large element of structural and technological variation, i.e. that industries under the same classification in reality have very different outputs and use different technologies to produce the same outputs. Truly geographical differences, i.e. differences that are determined by the geographical location of the processes are those caused by differences in natural conditions (as defined by climate, landscape, soil, etc.) or administrative conditions (such as legislation).

In spite of their apparent diversity, meat and dairy products are quite homogeneous products in terms of their raw materials and processing technologies applied. Likewise, administrative conditions are increasingly being harmonised across the EU, leaving natural conditions as the main cause for geographical variation in the production data.

Additionally, differences in consumer spending, behaviour, and preferences can be substantial, for example the degree of ownership of refrigerators and cars, which influences the size of the markets for fresh milk versus UHT-milk, and the emissions from shopping. The per capita consumption of meat and dairy products is generally smaller in the new EU member countries. While this affects the size of the overall impacts, it has little influence on the relative importance of the contributing processes and the impact intensity (the impact per EUR or impact per weight unit) for the different products.

For most processes in the model, data are available or could be estimated at the level of individual countries. However, due to the substantial intra-EU trade, products consumed in one country are only to some extent produced in the same country, and will therefore partly rely on the same average production data as products consumed in neighbouring countries. This is accentuated when focusing on the consequences of changes in demand, where even very geographically disparate changes in demand will often affect the same processes, as long as the affected markets are interconnected. If the analysis were repeated at the national level, it would therefore be expected to provide very similar overall results as the analysis at EU-27 level.

This does not mean that geographical differences are not important and relevant, but rather that these differences should be taken into account and subsumed in the overall analysis. For example, in the modelling of agricultural processes reported in Chapter 8.4, specific differences in technologies were incorporated in the overall models. These differences have geographical elements, but the differences do not follow national borders. Likewise, specific geographical differences are taken into account when assessing the improvement potentials in Chapter 4, where some improvements are only relevant under specific geographical conditions.

4 IMPROVEMENT OPTIONS

In Chapter 3, those processes in the life cycles of meat and dairy products were identified that are most relevant for environmental improvements. In this Chapter, improvement options are identified for each of these processes, the related process changes are described techno-economically, and the environmental improvement potentials quantified (Chapters 4.2-4.6). Rebound effects, synergies and dysergies of the improvement options are treated in Chapter 4.7. Chapter 4.8 finally provides the summary results of the environmental improvement potentials of all investigated improvement options. Before all this is presented, this chapter starts with a brief description of the methodology followed for identifying and quantifying the improvement options (Chapter 4.1).

4.1 Methodology for identifying and quantifying improvement options

The most promising environmental improvement options are identified for the processes that contribute more than 10 % of the total environmental impact per midpoint impact category, with the restriction that only options with proven technological feasibility and less than 12 years implementation horizon are included.

Improvement options are identified by a systematic procedure, covering different options for improvement according to their nature:

- Improvement options determined on the basis of the current variation between technologies, i.e. as an extension of current best practice within local constraints:
    - Improvements in efficiency of use of current inputs (inputs per product output), including reduction of product losses.
    - Improvements in reduction of emissions and wastes factors (emissions and waste per product output).
    - Improvements through increased utilisation of co-products and wastes (e.g. energy production from liquid manure).
- Improvement options determined from published technological studies, technology roadmaps, action plans, etc., and through expert consultations:
    - Improvements through substitutions of current inputs, covering substitution of chemicals (e.g. pesticides and heavy metals), energy sources (from fossil to renewable), and raw materials (e.g. selection of protein sources with lower environmental impacts).
    - Improvements in technologies that go beyond current best practice or substitutions, or that combine several of the above types of improvements, e.g. precision agriculture, micro-filtration of milk to enhance shelf-life, and new technologies to improve feed efficiency and reduce nutrients in manure.

Each improvement option is modelled in terms of its life cycle impacts, using the same data sources and procedures as for the current consumption (see Chapters 8 and 9). The improvement potentials are quantified (in Table 4.20) as the difference in environmental impacts for the functional unit, separately with and without each improvement option. For each improvement option, the improvement potential is expressed as a percentage of the total environmental impact, both relative to the functional unit (the total annual consumption of meat and dairy products in EU-27) and relative to the normalisation reference (the total annual final consumption in EU-27).

4.2 Improvement options for cereal crops

For the impact categories aquatic eutrophication, nature occupation, and respiratory inorganics, the arable farming of cereal crops contributes more than 20 % of the total life cycle impact for meat and dairy products (see Figure 3.1). Cereal cropping is also responsible for the main contribution of pesticide emissions in the life cycle of meat and dairy products.

4.2.1 Improvement options for nitrogen leaching from cereal crops

The eutrophication potential of cereal crops is by and large determined by the leaching from the area, mainly resulting from an imbalance in the timing of the growing season compared to the time when N is available for the cereal crop.

Leaching is especially important if there is no crop in winter to catch the N that is released from the soil pool in the autumn. How much N is actually leached depends then mainly on the rainfall and the water-holding capacity of the soil.

Managing this soil pool is a complicated matter. However, by sowing a catch crop in the autumn, keeping the surface covered with vegetation during winter, and – before sowing the cereal – ploughing in the catch crop, empirically based models have shown that it is possible to retain considerable amounts of the N that would otherwise be lost by leaching. Berntsen et al. (2004, 2006) showed that nitrate leaching be reduced by approximately 25 kg N/ha as an average for spring cereals on sandy and loamy soil, being greater on sandy soils than on loamy soils. Since this N is largely available for the cereal crop, the use of fertiliser can – and should – be reduced correspondingly, since part of the build-up of N in the soil may otherwise be lost though leaching.

This measure can be considered effective for normal fertilised spring cereals grown in areas where there is a precipitation surplus during wintertime. Lacking data for the area currently without winter crops, the potential area for this measure is calculated as the area with barley (a typical spring cereal) in EU-15 countries excluding the dryer countries Spain, Portugal, Italy and Greece. This gives a potential area of seven million ha. An additional three million ha of barley in new EU member countries would also be eligible for this improvement option, but since this area currently has a lower fertilisation level and consequent nitrate leaching, this area has not been included in the current potential area for this measure.

For the seven million ha it is estimated that introduction of a catch crop together with a reduced N fertilisation (under the assumption of an unchanged crop yield) will result in the following improvements per ha per year:

- Artificial N fertiliser requirement is reduced from 130 kg to 105 kg N (19 %);
- N-surplus is reduced from 59 kg to 34 kg N (42 %);
- Nitrate leaching is reduced from 217 kg to 111 kg NO_3 (49 %);
- N_2O emissions are reduced from 5.7 kg to 4.2 kg N_2O (26 %);
- Ammonia emissions are reduced from 11 kg to 10 kg NH_3 (9 %).

Implementing this option on seven million ha is expected to reduce the leaching from the overall EU-27 cereal production by 9 % with only minor impacts on other emissions. In addition, this option tends to improve soil organic matter, although the effect is anticipated to be small because of the concomitant reduction in N fertilisation.

In Table 4.1, the improvement potential of introducing catch crops on seven million ha is compared to the total impacts for meat and dairy products and the total impacts in EU-27. The calculation includes the saved artificial fertiliser manufacture. Formally, only 35 % of this improvement potential is related to meat and dairy products consumed in Europe, since the rest of the EU cereal production is used for other purposes. The 65 % of the improvement potential in Table 4.1 is thus a rebound effect (see Chapter 4.7). In addition to the quantified impacts in Table 4.1, catch crops may reduce erosion and affect the water balance by increased evapotranspiration. These effects have been assumed insignificant and no attempt at quantification has been made.

The improvement potential for aquatic eutrophication is 2.7 % of the total amount for meat and dairy products, but contributes only 6 % of the aggregated result in Table 4.1. This improvement potential has a low uncertainty, assessed to be +/-20 %, dominated by the uncertainty on the assumption of the size of the area for which catch crops can be implemented. The aggregated improvement potential of establishing catch crops is dominated by the reduction in N_2O emission (contributing 60 %) and the reduction in respiratory inorganics (contributing 21 %). The uncertainty on the N_2O emission is high (coefficient of variation 1.5) and the characterisation factors for global warming and respiratory inorganics are also very uncertain; see Table 9.4. The uncertainty on the aggregated result is therefore rather high (annual impact reduction between EUR 100 million and EUR 1 200 million).

Table 4.1: Improvement option 1: Annual improvement potential by establishing catch crops on seven million ha. Negative values signify an improvement (reduced impact). Only midpoint categories contributing more than 0.01 % change are shown.

Impact category	**Unit**	**Improvement potential**		
		In units of each impact category	**In % of total impacts for meat and dairy products**	**In % of total impacts in EU-27**
Midpoint categories:				
Acidification	m^2 UES	-2.13E+08	-0.22	-0.06
Eutrophication, aquatic	kg NO_3-eq	-2.40E+08	-2.70	-0.80
Eutrophication, terrestrial	m^2 UES	-9.81E+08	-0.25	-0.10
Global warming	kg CO_2-eq	-3.27E+09	-0.49	-0.07
Respiratory inorganics	kg PM2.5-eq	-1.22E+06	-0.14	-0.03
Endpoint (damage) categories:				
Impact on ecosystems	Species-weighted m2*years	-2.20E+09	-0.17	-0.04
Impacts on human well-being	QALY	-9.31E+02	-0.15	-0.03
Impacts on resource productivity	EUR	-1.96E+07	-0.13	-0.02
All impacts aggregated	EUR	-3.97E+08	-0.16	-0.04

Other options are related to very particular management options like extent and timing of soil tillage, timing of sowing, etc. While the effects are documented, the overall effects on the present background cannot be estimated and an implementation in practise is not suggested at the current time.

A future option, currently being investigated, is new technologies for precision farming, where fertiliser application is differentiated over the fields, either according to the expected growth of cereal or based on the actual N status of the cereal plant. It is expected that the N application to the crops could be reduced resulting in a reduced leaching. While promising from a theoretical point of view, the overall benefit remains to be proven in practice.

4.2.2 Improvement options for reducing area use by cereal crops

Nature occupation is the consequence of area occupied. The area for cereal crops could be reduced by options that raise the yield of cereals per ha.

However, the cereal production is not entirely market driven but is affected by a complex set of agricultural support systems and environmental regulations. A range of schemes has been implemented in different countries to limit intensification of cereal area. Also, in some countries, land use is linked to the livestock production by regulation, in order to reduce over-fertilisation with animal manure. Thus, disposal of animal manure becomes an important purpose of having the land, rather than an efficient crop production.

Where in many countries the yearly increase in cereal yield amounted to 3 % until 1990, the increase thereafter was less than 1 % (EC 2006b). It is unclear whether this slowdown of growth rate is due to the aforementioned regulation or due to a technological barrier for continued growth.

Looking at differences in crop yield among EU-15 countries and new EU member countries there is a huge gap in yield. The most important of the new EU member countries regarding cereal production (Poland and Hungary) obtain only half the yield per ha compared with EU-15 countries (Faostat 2006). Likely explanations for the lower cereal yield in the new EU member countries are smaller inputs of N and plant protection agents, restructuring of the farms, and similar socioeconomic reasons. Rabbinga & van Diepen (2000) modelled the biophysical production potential and its regional distribution within Europe and concluded that it should be possible to raise the yields in Eastern Europe by intensifying farming practise to reach the West European levels.

On this basis, it is assumed that cereal yields in the new EU member countries can be increased to be comparable with the yields in EU-15 following an increased input of fertiliser and plant protection agents, and better management. The area with barley and wheat in new EU member countries amounts to approximately 10 million ha with an average yield of 2.8 Mg/ha. On this area, it is assumed that an increase of yield to 5.2 Mg/ha (the average yield in EU-15 less Spain, Portugal, Italy and Greece, where crop yields are lower) can be achieved by an increase in N fertilisation from 70 to 130 kg/ha, thus reducing the area necessary for the production of one Mg cereal from 3 570 m2 to 1 923 m^2. While emissions per ha will increase as a result of the increased fertilisation, the emissions per ton of cereal produced will decrease (see Table 4.2).

The consequences of the improvement for the inputs and outputs of the cereal production are reported in Table 4.2 and the impact at EU-27 level in Table 4.3. The overall effect on the total EU cereal production will be a 9 % reduction in land use and ammonia emissions with only small changes in other emissions. Technically, only 35 % of this improvement potential is related to meat and dairy products consumed in Europe, since the rest of the EU cereal production is used for other purposes. The 65 % of the improvement potential in Table 4.3 is thus a rebound effect (Chapter 4.7).

In addition to the quantified impacts in Table 4.3, intensification may reduce erosion through the increased amount of crop residues. This effect has been assumed insignificant and no attempt at quantification has been made.

The uncertainty of the quantified improvement potential is assessed to be +/-15 % for the midpoint categories nature occupation and global warming, and is dominated by the uncertainty on the assumptions about the actual cereal yields that can be achieved and the area for which this is possible. Together with respiratory inorganics, these impact categories contribute 98 % of the overall impact potential. The uncertainty on the aggregated result is rather high (annual impact reduction between 2 500 and 22 000 MEUR) due to the high uncertainty on the characterisation factors for these impact categories; see Table 9.4.

Table 4.2: Improvement option 2: Changes in inputs and outputs following intensification of cereal crop growing. In addition to the listed changes, fuel input and fuel related emissions are expected to follow the area, implying a 46 % reduction per kg cereal for the affected area or a 9 % reduction per kg for the average cereal production.

	Original average	**Sub sample appropriate for intensification (10 million ha)**	**Improved growing practise/intensification**		**New average after improvements**	
	Per ha and 4 100 kg cereal	**Per ha and 2 800 kg cereal**	**Per ha**	**Per 2 800 kg cereal**	**Per ha**	**Per 4 100 kg cereal**
Input, kg:						
Fertiliser-N	88	58	108	58	97	88
Manure-N	12	12	22	12	13	12
Fertiliser-P	16	12	22	12	17.5	16
Fertiliser-K	52	35	66	36	57	52
Cropped area, ha:	1	1	1	0.54	1	0.91
Output, kg:						
Cereals	4 100	2 800	5 200	2 800	4 500	4 100
Ammonia	9.7	8.6	10.8	5.8	10.0	9.1
N_2O	4.6	3.5	5.7	3.1	5.0	4.5
Nitrate	199	155	261	141	215	196

Table 4.3: Annual improvement potential by intensification of cereal crop growing in EU-27. Negative values signify an improvement (reduced impact).

Impact category	Unit	Improvement potential		
		In units of each impact category	In % of total impacts for meat and dairy products	In % of total impacts in EU-27
Midpoint categories:				
Acidification	m^2 UES	-1.19E+09	-1.26	-0.31
Ecotoxicity, aquatic	kg-eq. TEG water	-4.85E+10	-0.03	-0.02
Ecotoxicity, terrestrial	kg-eq. TEG soil	-1.84E+09	-0.30	-0.02
Eutrophication, aquatic	kg NO_3-eq.	-7.04E+07	-0.79	-0.23
Eutrophication, terrestrial	m^2 UES	-5.42E+09	-1.40	-0.55
Global warming	kg CO_2-eq.	-5.26E+09	-0.79	-0.11
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	-2.89E+07	-2.09	-0.17
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	-2.99E+06	-0.26	-0.02
Mineral extraction	MJ extra	-7.28E+06	-0.14	-0.01
Nature occupation	m^2 arable land	-4.82E+10	-4.94	-1.77
Non-renewable energy	MJ primary	-1.21E+11	-1.38	-0.09
Ozone layer depletion	kg CFC-11-eq.	-3.88E+02	-0.20	-0.01
Photochemical ozone, vegetation	m^2*ppm*hours	-1.36E+11	-2.05	-0.25
Respiratory inorganics	kg PM2.5-eq.	-1.28E+07	-1.50	-0.26
Respiratory organics	person*ppm*hours	-1.41E+07	-1.96	-0.25
Endpoint (damage) categories:				
Impact on ecosystems	Species-weighted m^2*years	-4.61E+10	-3.52	-0.86
Impacts on human well-being	QALY	-9.18E+03	-1.48	-0.26
Impacts on resource productivity	EUR	-2.45E+08	-1.57	-0.26
All impacts aggregated	EUR	-7.42E+09	-3.02	-0.66

4.2.3 Improvement options for emissions of respiratory pollutants from cereal crops

Approximately equal shares of ammonia, nitrogen oxides, and particulates contribute to the overall impact of inorganic respiratory pollutants from cereal farming. The respiratory pollutants with photochemical ozone creation potential are dominantly carbon monoxide and nitrogen oxides.

The ammonia emission from cereal production is partly related to the use of fertiliser and partly to the N turnover in the soil. Thus, a reduced N fertilisation as suggested through use of catch crops (see Chapter 4.2.1) will slightly reduce the N emission related to fertiliser, but since the soil N pool increase, the total reduction is small. The type of fertiliser may also influence the rate of ammonia released at application, where calcium nitrate seems superior to other types. However, quantitatively a reduction from change of type of fertiliser is expected to be modest.

Nitrogen oxides, particulates, and carbon monoxide are all results of incomplete fuel combustion. By placing the same emission requirements on off-road equipment, such as tractors and harvesters, as currently placed on vehicles for road transport, significant emission reductions can be achieved.

EU Directive 2000/25/EC already regulates the maximum allowable emissions of nitrogen oxides, particulates and carbon monoxide from new tractor engines, with emission standards entering into force in two stages (I and II; see Table 4.4). When fully effective, this regulation is expected to lead to more than 50 % reduction in these emissions compared to those reported in the year 2000 NAMEA, due to implementation of, for example, electronic engine control.

EU Directive 2005/13/EC requires a further tightening of emission limits of new tractor engines for nitrogen oxide emissions, tightening limits by 30-40 % relative to stage II, and for the smallest engine category (18-37kW) reductions of 20 % in NOx and 25 % in PM10 relative to stage II. The technology involved is expected to be engine modifications, common rail injection, air-air charge cooling, limited, uncooled exhaust gas recirculation, as well as electronic engine control for small engines (DfT 2006). Stage IIIB tightens PM limits by around 90 % relative to Stage II, and is expected to force the adoption of diesel particulate filters and sulphur-free fuel, while giving a slight reduction in fuel efficiency, estimated as a 0.5 % increase in CO_2 emissions from agriculture. Finally, Stage IV tightens NOx limits by 75 % on >75kW engines, expected to force the adoption of Selective Catalytic Reduction (SCR) de-NOx treatment of the exhaust gas. These systems rely on adding urea to reduce the NOx over a catalyst, and therefore depend on the user to keep urea tanks filled.

It seems unrealistic to force the implementation of these technologies more than is prescribed by the Directives already in force.

Table 4.4: Emission limits for new non-road engines according to EU Directives 2000/25/EC and 2005/13/EC.

Engine size kW net power	Legislative stage	Implementation from	Emissions			
			CO	HC	NOx	PM
130-560	I	Jan 1999	5	1.3	9.2	0.54
	II	Jan 2002	3.5	1	6	0.2
	IIIA	Jan 2006	3.5	4[a]		0.2
	IIIB	Jan 2011	3.5	0.19	2	0.025
	IV	Jan 2004	3.5	0.19	0.4	0.025
75-130	I	Jan 1999	5	1.3	9.2	0.7
	II	Jan 2003	5	1	6	0.3
	IIIA	Jan 2007	5	4[a]		0.3
	IIIB	Jan 2012	5	0.19	3.3	0.025
	IV	Oct 2014	5	0.19	0.4	0.025
56-75	I	Apr 1999	6.5	1.3	9.2	0.85
	II	Jan 2004	5	1.3	7	0.4
	IIIA	Jan 2008	5	4.7[a]		0.4
	IIIB	Jan 2012	5	0.19	3.3	0.025
	IV	Oct 2014	5	0.19	0.4	0.025
37-56	I	Apr 1999	6.5	1.3	9.2	0.85
	II	Jan 2004	5	1.3	7	0.4
	IIIA	Jan 2008	5	4.7[a]		0.4
	IIIB	Jan 2013	5	4.7[a]		0.025
18-37	II	Jan 2001	5.5	1.5	8	0.8
	IIIA	Jan 2007	5.5	7.5[a]		0.6

a) HC + NOx

4.2.4 Improvement options for pesticide use in cereal crops

Besides copper emissions to soil, pesticide emissions are the most important contributors to aquatic ecotoxicity. The pesticide that comes out as most important in the EU-27 NAMEA is Atrazine, a herbicide used in cereal production, and especially for maize crops. In 2003, following a review, the EU has decided not to re-register Atrazine. Use was withdrawn on 10 September 2004 in all EU Member States except in those that requested essential uses (Ireland, United Kingdom, Spain, Portugal), where use was authorised until 30 June 2007.

Without reducing effectiveness, the overall impact from pesticides can be reduced by choosing the least environmentally harmful pesticide for a given application, and by improved application techniques.

A study by DEFRA (1998) models the effect of a 50 % tax premium on the price of cereal grass weed herbicides, with tax exemptions for pesticide used with specified minimisation techniques, to reduce herbicide usage by 20 % to 25 %. A case study by Giupponi (2001) on substitution of the herbicide Atrazine by herbicides from the sulphonylurea family refers to a reduction in toxicity impact to 12-31 % of the original level.

As the current usage of pesticides (after the phasing out of Atrazine) is not well known, it has not been attempted to quantify the improvement potential in terms of aquatic toxicity equivalents.

4.3 Improvement options in animal husbandry

Animal husbandry has a large share of the total life cycle impact for meat and dairy products for practically all impact categories in Figure 3.1. Most of the below improvement options are relevant for both dairy farming and pig production, and therefore these improvement options are treated together, under the same headings. The analysis also considers beef production, but generally it appears that the identified improvement options are not applicable to beef production.

4.3.1 Improvement options for ammonia emissions in animal husbandry

Ammonia is the dominating pollutant of the impacts from animal husbandry for the impact categories acidification, terrestrial eutrophication, and respiratory inorganics. The main contribution originates from processes related to manure production and handling. Generally, the rate of ammonia emission from manure facilities (liquid manure) increases with NH_4^+ concentration, temperature, pH and evaporation surface (Rom 2002, Nicks 2006, Gustafson & Jeppsson 2006).

In the pig farming systems considered for EU-27, the total emission of NH_3 estimated for the northern system was 1.26 kg NH_3 per pig produced, increasing to 1.75 and 2.62 in the eastern and southern systems, respectively. This difference is related to N in feed (and in manure) and for the southern system an expected larger ammonia loss in stables and storage, partly as a function of the manure handling facilities and partly as a function of higher temperature.

In the dairy systems representing EU-27 milk production, the overall N utilisation is in general only modest at the animal level. An improvement of this can also affect the NH_3 emission.

Proven interventions to reduce ammonia losses from animal husbandry are:

- Optimised protein feeding by reducing protein (N) supply in feed (Frank et al. 2002, Swensson 2003, Poulsen et al. 2003, Dourmad & Jondreville 2006, Aarnink et al. 2007). Poulsen et al. (2003) estimated that for typical Danish (Northern Europe) pig production, the N excretion per pig could be reduced from 5.3 kg N per pig produced to 3.9 kg N, by using two feed mixtures for sows (differing in N content) and reducing the N concentration in slaughter-pig feed by 5 % and instead adding synthetic amino acids. This alone would reduce ammonia emission by 22 %, i.e. from the current 1.26 kg ammonia to 0.98 kg. Frank et al. (2002) and Swensson (2003) observed that a 25 % lowered N supply to dairy cows did not impact milk yield, and reduced ammonia emission in the stable by more than 65 %.
- Reducing the pH of the liquid manure (Pedersen 2004, Gustafsson & Jeppsson, 2006). This can reduce the ammonia emission from stables and storage facilities by 60-70 %. For the total chain of manure handling, the ammonia emission will thus be reduced by 40 %.
- Reduced surfaces for liquid manure through improved construction of manure channels (Rom, 2002).
- Cooling of the storage facilities of liquid manure.

These are all elements considered in the most recent BAT reference document (BREF 2003). For the present purpose the quantification of their implementation have been substantiated through use of more recent literature sources.

Accordingly, the following improvement options are considered:

- Optimised protein feeding in all pig farming systems, and in the central, UK-type and lowland dairy farming systems, including an upgrading of the technical efficiency of the central-eastern pig farming system to northern standards. By a reduced concentration of N in feed, a smaller amount of N is present in manure and thus a smaller amount is lost as ammonia. Supplementation of amino acids maintains a proper balance among the amino acids necessary for optimal protein utilisation and pig growth, so that the overall production results are not affected. In the western and eastern-southern type of dairy production, the diets already have moderate N contents and it is not realistic that the N supply can be reduced further without impairing the milk yield. Also in the beef systems, it is unrealistic to reduce the dietary N; in the beef fattening units because the N supply is already optimised, and in the grazing animals (suckler herds and steers) because of the very small amounts of complementary protein rich feed provided.
- Reduction of pH in liquid manure in all pig and dairy systems except dairy south, in which the overall N utilisation is good. In the beef fattening systems, this measure is only realistic regarding beef fattening units (with liquid manure), since suckler herds and steers are almost entirely kept on grassland or with solid manure.

The options of reducing surfaces for liquid manure through improved construction of manure channels, and of cooling of the storage facilities, have not been modelled here, since they are more difficult to implement than the above proposed improvements and since their importance is reduced once the above measures are implemented.

The lower ammonia emissions mean a larger content in the manure of ammonia N available for plant fertilisation (thus saving input of fertilisers).

For all pig farming systems, implementation of optimised feeding is expected to reduce the overall N excretion in manure by 32 %. Assuming no change in emission factors from stable and storage, the resulting ammonia emission is reduced by 25 %. This is a conservative estimate of the reduction potential, when strictly implemented, since the lower N in manure is expected to reduce the emission factor as well (Aarnink et al, 2007). However, in practice, fluctuations in diet composition will appear, that will reduce the actual level of optimisation. These two effects are assumed to cancel each other out.

In the dairy systems, an optimised feeding (going from 17 % crude protein in dry matter to 14 %) in the relevant systems (central, UK-type, and lowland systems) reduces the overall N excretion from the cattle by approximately 48 kg per cow and year. This is expected to reduce the ammonia emission by 35 % in the systems considered.

The changes in feeding also have an important effect on nature occupation due to the differences in the amount of land used for grain and protein crops.

A pH reduction in the liquid manure through application of sulphuric acid is assumed to reduce the overall ammonia emission from pig stable and storage of pig manure by 60 %, having relatively largest effect in the southern systems, where the emission factors are largest. For all dairy systems, except the east-southern system, reduction of the pH of liquid manure is considered for the manure deposited in stables. Ammonia losses from dairy stables and storage of dairy cattle manure are estimated to be reduced from 14 % (the default value) to 7 %, i.e. a 50 % reduction.

The changes in inputs and outputs of the modelled EU-27 production systems, when implementing optimised protein feeding and liquid manure pH reduction, are reported in Table 4.5 for pigs and in Table 4.6 for dairy cattle. The overall impact for EU-27 is reported in Table 4.7 for optimised feeding and in Table 4.8 for liquid manure pH reduction.

Table 4.5 and Table 4.6 show that optimised protein feeding reduces not only the ammonia emission, but also the leaching potential, due to a lower overall N excretion in manure. A pH reduction of the liquid manure markedly reduces the ammonia emission, but at the same time, leaching of N is expected to increase slightly. From a biological point of view, the collected extra ammonia N is fully available for the plants and therefore could fully substitute fertiliser N after accounting for ammonia emissions in relation to spreading. Therefore, if taken into account in the manure and fertilisation administration, the benefit of reduced ammonia emission through pH reduction could be achieved without noticeable increase in leaching. However, due to the current administration of manure N, where only part of the N in manure is assumed to replace fertiliser N (60 % in NW European systems and 40 % in the eastern and southern systems), an increase in leaching is expected.

It is interesting to note that for pig systems, a pH reduction of the manure (and thus a smaller ammonia emission) can also be obtained through changed feeding. Adding acidifying salts or using slowly fermentable carbohydrates in the pigs' diet can reduce the ammonia emission by up to 70 % (Aarnink et al. 2007), i.e. to a level comparable with an acidification through treatment with sulphuric acid.

The uncertainty on these improvement potentials is assessed to be +/-20 % around the mean values for the ammonia emission and area use, and is dominated by the uncertainty of the assumptions on the reductions that can be achieved and the degree of implementation. Together with the changes in potential global warming impacts, the changes in ammonia emissions and area use contribute more than 90 % of the overall impact potentials. The 95 % confidence interval for the aggregated impact reduction for optimised protein feeding is 1 000 to 16 000 MEUR, while for pH reduction of liquid manure it is 600 to 34 000 MEUR. The larger uncertainty for pH reduction is due to the larger relative role of ammonia emissions in the overall result. While the estimates of emission reductions for ammonia have low uncertainty (+ /- 20 %), the characterisation factors for the involved damage categories are very uncertain, especially for respiratory inorganics; see Table 9.4.

Table 4.5: Effect of optimising protein feeding and liquid manure pH reduction on input and emissions from pig production (per 10 pigs produced).

	Average pig farming model EU-27	Improved feeding in all systems	Liquid manure pH reduction in all systems	Combined effect of the two measures
Input:				
Cereals, kg	2 499	2 845	2 499	2 845
Soy meal, kg	549	100	549	100
Synthetic amino acids, kg	17	40	17	40
Output:				
Replacement of artificial fertiliser, kg N	22.1	16.3	25.5	18.9
Ammonia, kg	16.8	12.7	9.5	7.2
Nitrate, kg	72.9	54.2	85.0	62.8
N_2O, kg	0.99	0.74	1.03	0.76

Table 4.6: Main changes in inputs and emissions in dairy production following improvement options (kg per 10 Mg milk in EU-27 average).

	Average dairy model EU-27	Optimised feeding in the central, UK-type, and lowland systems (affecting 11.7 million cows in total)	Liquid manure pH reduction for all systems except east-south (affecting 17.3 million cows in total)	Combined effect of the two measures
Inputs, kg:				
Cereals	547	1 316	547	1 316
Soy meal	1 076	430	1 076	430
Fertiliser N	269	281	262	276
Emissions, kg:				
Ammonia	74	62	60	51
N_2O	23.9	22.7	24.2	22.9
Nitrate	520	470	540	488

Table 4.7: Improvement option 3: Annual improvement potential by optimising protein feeding in pig and dairy farming. Negative values signify an improvement (reduced impact). Only midpoint categories contributing more than 0.01 % change are shown.

Impact category	Unit	Improvement potential		
		In units of each impact category	In % of total impacts for meat and dairy products	In % of total impacts in EU-27
Midpoint categories:				
Acidification	m^2 UES	-7.42E+09	-7.82	-1.95
Ecotoxicity, aquatic	kg-eq. TEG water	3.34E+12	2.34	1.09
Ecotoxicity, terrestrial	kg-eq. TEG soil	6.29E+09	1.04	0.07
Eutrophication, aquatic	kg NO_3-eq.	-4.07E+08	-4.59	-1.35
Eutrophication, terrestrial	m^2 UES	-3.50E+10	-9.01	-3.52
Global warming	kg CO_2-eq.	7.31E+09	1.09	0.15
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	3.98E+07	2.87	0.23
Mineral extraction	MJ extra	7.67E+07	1.46	0.08
Nature occupation	m^2 arable land	-2.01E+10	-2.06	-0.74
Non-renewable energy	MJ primary	1.93E+11	2.20	0.14
Ozone layer depletion	kg CFC-11-eq.	4.24E+03	2.22	0.14
Photochemical ozone, vegetation	m^2*ppm*hours	1.02E+11	1.53	0.19
Respiratory inorganics	kg PM2.5-eq.	-2.61E+07	-3.07	-0.54
Respiratory organics	person*ppm*hours	8.93E+06	1.24	0.16
Endpoint (damage) categories:				
Impact on ecosystems	Species-weighted m^2*years	-1.69E+10	-1.29	-0.31
Impacts on human well-being	QALY	-1.80E+04	-2.91	-0.50
Impacts on resource productivity	EUR	-3.91E+08	-2.52	-0.42
All impacts aggregated	EUR	-4.10E+09	-1.67	-0.37

Table 4.8: Improvement option 4: Annual improvement potential by pH reduction of liquid manure in pig and dairy farming in EU-27. Negative values signify an improvement (reduced impact). Only midpoint categories contributing more than 0.01 % change are shown.

Impact category	Unit	Improvement potential		
		In units of each impact category	In % of total impacts for meat and dairy products	In % of total impacts in EU-27
Midpoint categories:				
Acidification	m^2 UES	-1.30E+10	-13.72	-3.43
Ecotoxicity, terrestrial	kg TEG-eq s	-5.17E+09	-0.86	-0.06
Eutrophication, aquatic	kg NO_3-eq	2.57E+08	2.89	0.85
Eutrophication, terrestrial	m^2 UES	-6.08E+10	-15.64	-6.11
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq	-2.31E+07	-2.02	-0.14
Respiratory inorganics	kg PM2.5-eq	-5.31E+07	-6.25	-1.11
Endpoint (damage) categories:				
Impact on ecosystems	Species-weighted m^2*years	-5.68E+09	-0.43	-0.11
Impacts on human well-being	QALY	-3.73E+04	-6.02	-1.04
Impacts on resource productivity	EUR	-8.58E+08	-5.52	-0.92
All impacts aggregated	EUR	-4.41E+09	-1.80	-0.40

4.3.2 Improvement options for nitrogen leaching from animal farms

The aquatic eutrophication impacts from meat and dairy products are dominated by nitrogen leaching.

In the dairy systems representing EU-27 milk production, the overall N efficiency (N in outputs relative to N input) is only modest, both at the animal level and the field level, resulting in an N surplus of 83-265 kg N/ha. There is room for an optimised protein feeding and reduction of pH in the liquid manure as described in the previous section. In addition, the overall utilisation efficiency of available N at the farm level can be considered. In particular, the N in the part of the manure that falls on pasture is poorly utilised. In the systems considered, this N efficiency is estimated to 20 %, corresponding to conventions used in N planning in Denmark. However, since the overall N supply on dairy farms is abundant, measures could be taken by which a higher N efficiency is obtained. For instance, it is shown that a high grass intake can be secured even with a substantial reduced grazing time, allowing a larger proportion of the manure to be collected in the barn and therefore available for redistribution (Kristensen et al. 2007, Oudshorn et al. 2007).

Even an efficiency of manure nitrogen of 60 % (after loss of ammonia in stables and storage) relative to artificial fertiliser is not high. For instance, in Denmark, where excessive fertiliser application is penalised by administrative fines and farmers are required to make fertiliser plans and keep fertiliser accounts, it is mandatory since 2006 to calculate N in liquid manure in the fertiliser plan at utilisation efficiencies relative to artificial N of 70 % for cattle and 75 % for liquid manure from pigs. Thus, a utilisation efficiency of 70 % means that the total N in the animal manure (after loss of ammonia in stables and storage) is equivalent to 70 % of commercial fertiliser N. Together with a maximum allowed amount of 'usable' N per ha for a specific crop, an increased assumed N utilisation efficiency will reduce the overall N application per ha.

These required utilisation efficiencies are expected to be attainable by reducing ammonia losses in connection to the application in the fields. On areas with a relatively large livestock density like dairy and pig farms, it is assumed that a comparable reduction in fertiliser N input will be possible without reducing crop yield due to a large N pool in the soil. Under this assumption, an increased utilisation of manure N will directly result in a smaller import of fertiliser and correspondingly less leaching of N. In addition, N_2O emission related to use of fertiliser and leaching would be reduced.

Table 4.9 shows the estimated potential for reducing N leaching following a tightening of the regulation for overall fertilisation on dairy and pig farms and farms that use considerable amounts of pig manure. Table 4.10 shows the resulting annual improvement potential.

Because of the large global warming potential of N_2O, the reduction in N_2O emissions contribute more than 60 % of the aggregated improvement potential in Table 4.10. Aquatic eutrophication contributes 14 % and respiratory inorganics, 12 %.

Table 4.9: Potential for reduced use of N fertiliser and leaching in dairy and pig production systems in EU-27 following a tightening of the regulation of manure application.

	Fertiliser Tg N/year	Leaching Tg N/year	Leaching Tg nitrate/year	N_2O emission Tg/year
Present livestock model (dairy and pigs)	4.12	2.28	10.1	0.365
Reduction potential by limiting total N fertilisation and requiring manure N to be calculated at a utilisation efficiency relative to artificial N of:				
a) 70 % in dairy systems	-0.842	-0.842	-3.73	-0.025
b) 75 % in pig systems	-0.293	-0.293	-1.30	-0.009
a) and b) combined	-1.14	-1.14	-5.03	-0.034

Table 4.10: Improvement option 5: Annual improvement potential by tightening the regulation of manure application. Negative values signify an improvement (reduced impact). Only impact categories contributing more than 0.01 % change are shown.

Impact category	Unit	Improvement potential		
		In units of each impact category	In % of total impacts for meat and dairy products	In % of total impacts in EU-27
Midpoint categories:				
Acidification	m^2 UES	-2.48E+08	-0.26	-0.07
Ecotoxicity, terrestrial	kg-eq. TEG soil	-5.07E+09	-0.84	-0.05
Eutrophication, aquatic	kg NO_3-eq.	-2.66E+09	-29.98	-8.82
Eutrophication, terrestrial	m^2 UES	-1.06E+09	-0.27	-0.11
Global warming	kg CO_2-eq.	-1.68E+10	-2.51	-0.35
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	-2.67E+06	-0.19	-0.02
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	-5.37E+06	-0.47	-0.03
Mineral extraction	MJ extra	-2.19E+07	-0.42	-0.02
Non-renewable energy	MJ primary	-2.82E+10	-0.32	-0.02
Ozone layer depletion	kg CFC-11-eq.	-9.13E+02	-0.48	-0.03
Photochemical ozone, vegetation	m^2*ppm*hours	-2.12E+10	-0.32	-0.04
Respiratory inorganics	kg PM2.5-eq.	-3.35E+06	-0.39	-0.07
Respiratory organics	person*ppm*hours	-1.68E+06	-0.23	-0.03
Endpoint (damage) categories:				
Impact on ecosystems	Species-weighted m^2*years	-1.19E+10	-0.91	-0.22
Impacts on human well-being	QALY	-2.73E+03	-0.44	-0.08
Impacts on resource productivity	EUR	-5.45E+07	-0.35	-0.06
All impacts aggregated	EUR	-1.93E+09	-0.79	-0.17

The uncertainty on the reduction in nitrate leaching and ammonia emissions is rather low, assessed to be +/-40 % around the mean values, dominated by the uncertainty on the assumptions on the reductions that can be achieved and the degree of implementation. The reduction in N_2O emissions is highly uncertain (coefficient of variation 1.5) and the characterisation factors for global warming, aquatic eutrophication and respiratory inorganics are also very uncertain; see Table 9.4. The uncertainty on the aggregated result is therefore rather high (annual impact reduction between EUR 650 million and EUR 5 700 million).

4.3.3 Improvement options for copper emissions to soil from animal husbandry

The main contribution to aquatic ecotoxicity from pig and dairy farming is copper emissions to soil.

In pig diets, copper is often oversupplied either because large safety margins are applied or because it is used as growth promoter as explained in Chapter 8.4.3. Consequently, pig manure has a high concentration of this element (Dourmad & Jondreville 2006). EU legislation from 2003 (EC 2003a) limits the concentration of Cu to be used in pig diets, but the acceptable level of Cu in feed mixtures is still from three to more than ten times the estimated nutrient requirement, depending on growth stage of the pigs. Since less than 1 % of Cu is retained in the pig carcass, this large Cu supply leads to high concentrations in the manure.

In dairy production, copper has until recently been used routinely in footbaths for maintaining hoof health. The remains from the footbaths end up in the manure and consequently the concentration in cattle manure is found to be considerable higher than expected from the nutritional requirements of the cows.

In EU-27 livestock models, the Cu in pig and dairy manure was estimated from recent Danish measurements (Landskontoret for Planteavl 2006), showing for pig production an emission of Cu six times larger than theoretically calculated from the nutrient requirements. For dairy cattle, the difference is a factor of three. As explained in Chapter 8.4.3, there are good reasons to believe that the use of Cu adheres to the maximum stipulated by EU legislation. Thus, there are good opportunities to reduce this emission, most easily in the pig production, using different probiotics; it becomes more difficult in dairy production. However, in dairy production the Cu can be reduced through better general hygiene and limiting the use of footbaths to short periods, instead of throughout the year.

The estimates in Table 4.11 show that the overall Cu emission from dairy and pig farming can potentially be reduced to one third following tighter regulation on its use as feed additive and other bactericidal uses. The decrease in Cu as feed additive requires an increased attention to Cu content and Cu bioavailability in the feed components, as well as the addition of alternative probiotics. The environmental and economic effects of this addition have been included in the further modelling of this improvement option.

Table 4.11: Cu emissions and nutritional requirements for dairy cattle and pigs in EU-27.

	Dairy cattle Mg/year	Pig farming Mg/year	Total Mg/year	Total in % of current emissions
Current emissions	3 640	3 260	6 900	100
Minimum requirement	1 270	540	1 810	26
Minimum + 25 %	1 590	680	2 270	33
Reduction potential	2 050	2 580	4 630	67

The main uncertainty regarding this improvement option is not related to the size of the possible emission reduction, but rather to its importance (see Chapter 9.1) relative to the costs of implementation (see Chapter 5.2.6).

4.3.4 Improvement options for methane emissions from animal husbandry

Of the total global warming potential from dairy and cattle farming, emissions of carbon dioxide contribute only 6-17 %, while dinitrogen oxide and methane contribute approximately equal shares of the remaining 83-94 %. For pig farming, carbon dioxide contributes 23-25 %, dinitrogen oxide 29-32 %, and methane 43-46 %. Dinitrogen oxides are indirectly linked to the N turnover, which means that improved N efficiency and reductions in emissions of other N compounds is also the key to reducing N_2O emissions.

Besides its role as a greenhouse gas, methane is also responsible for 58-84 % of the photochemical ozone creation potential from dairy and cattle farming, the rest being caused by CO and NOx emissions, which have been dealt with under cereal production in Chapter 4.2.3.

The methane from enteric rumen fermentation in cattle is to a large extent sensitive to manipulation through dietary means. The methane emission can be reduced through a more concentrate rich diet at the expense of roughage, and especially through a higher concentration of fat in the diet. Following Kirchgessner et al. (1995), it can be estimated that increasing the concentration of dietary fat from 3 % to 5 % in the dairy cow diets will reduce the methane emission by 17 %.

It is well known that the use of unsaturated fatty acids can reduce the methane emission even further. However, using this measure is a delicate matter, since milk composition in some cases may be unacceptably altered and cow health may be impaired. Therefore, there is need for further technological development before this option can be recommended in general.

As an improvement option, it is therefore suggested to provide the farmers with informational and management tools that allow them to calculate the methane emission potential of the different diets and thereby integrate a regard for methane emissions in the regular feed optimisation procedures. This can be combined with a maximum allowed calculated methane emission potential corresponding to what is known to be achievable without detrimental effects, i.e. corresponding to the above mentioned 17 % reduction. The overall impact of this is shown in Table 4.12.

The uncertainty on the emission reduction is assessed to be low, +/-30 % around the mean value, and is dominated by the uncertainty on the assumptions on the reductions that can be achieved, and the degree of implementation. The large uncertainty on monetarising ecosystem impacts from global warming implies that the aggregated improvement potential is nevertheless quite uncertain (EUR 500 million to EUR 4 400 million annually).

The second improvement option suggested to reduce methane emissions is the use of liquid manure for biogas production. This will have a threefold effect in relation to global warming potential. According to Sommer et al. (2001), the methane emission from the manure will be reduced by 40 % or 1.1 kg methane per Mg manure, the N_2O emissions will be reduced by 14 g per Mg manure, and at the same time, the methane produced will substitute energy from fossil sources and thereby reduce the overall contribution to global warming.

The EU-27 dairy and pig farming models include a total of 55 million livestock units, producing 830 Tg manure annually, of which it is estimated that 50 % may be available for biogas production (i.e. excluding farms with solid manure and farms that are too small and/or too distant from other larger farms). With a production of 22 m^3 biogas per Mg manure (Birkmose 2000), an energy content of 23MJ/m^3 biogas and a 37 % efficiency of electricity generation (Nielsen 2004), an electricity production potential of 52 kWh per Mg manure or 21.6 TWh/year is obtained for the utilisable manure in EU-27. With a gross value of electricity (before distribution) of 54 EUR/MWh this gives an income of EUR 1 170 million/year.

The required investment is EUR 750 million/year at a cost of 5 000 EUR/kW capacity (Walla & Schneeberger 2003) assuming a capacity utilisation of 80 % and a lifetime of 20 years. This value has been applied to account for the necessary contribution from the construction industry.

The biogas is assumed utilised in a combustion plant and the emissions from this have been taken from Nielsen (2004). The reduction in emissions from the manure amounts to 460 Gg CH_4 and 5.8 Gg N_2O per year. The resulting improvement potential is summarised in Table 4.12.

The uncertainty on the global warming reduction is assessed to be +/-60 % around the mean value at midpoint. The aggregated value is further influenced by the large uncertainty on monetarising ecosystem impacts from global warming, resulting in a span between EUR 1 200 million and EUR 7 500 million annually for this improvement option. The investment and net operating costs is another important source of uncertainty for the overall benefit of this improvement option, which is further discussed in Chapters 5.2.8 and 5.4.

Another option to consider is an overall intensification of milk and beef production, relying more on concentrate to stimulate milk yield and growth, and consequently reducing the number of animals. This option seems most relevant for the dairy production. The overall EU milk yield per cow is 5 900 kg/year. In Denmark and Sweden, milk yield per cow and year is 8 500 kg and the predicted yield in 2010 is approximately 10 000 kg. This is a result of improved breeding, feeding and overall management. There is no reason to believe that similar milk yields cannot be obtained elsewhere, in particular if the milk quota system is unrolled.

Table 4.12: Improvement options 7 and 8: Annual improvement potential by methane reducing diets for dairy cows and gasification of pig and dairy cow liquid manure in EU-27. Only midpoint categories contributing more than 0.01 % change are shown. Negative values signify an improvement (reduced impact).

Impact category	**Unit**	**Improvement potential**					
		In units of each impact category		**In % of total impacts for meat and dairy products**		**In % of total impacts in EU-27**	
		Diet change	**Biogas**	**Diet change**	**Biogas**	**Diet change**	**Biogas**
Midpoint categories:							
Acidification	m^2 UES		-1.01E+09		-1.06		-0.27
Ecotoxicity, terrestrial	kg TEG-eq s		-8.94E+09		-1.48		-0.10
Eutrophication, aquatic	kg NO_3-eq		-6.97E+06		-0.08		-0.02
Eutrophication, terrestrial	m^2 UES		-9.51E+08		-0.24		-0.10
Global warming	kg CO_2-eq	-1.70E+10	-2.93E+10	-2.54	-4.37	-0.36	-0.62
Human toxicity, carcinogens	kg C_2H_3Cl-eq		-2.97E+06		-0.21		-0.02
Human toxicity, non-carc.	kg C_2H_3Cl-eq		-4.44E+06		-0.39		-0.03
Mineral extraction	MJ extra		-6.80E+06		-0.13		-0.01
Non-renewable energy	MJ primary		-3.21E+11		-3.66		-0.23
Ozone layer depletion	kg CFC-11-eq		-4.93E+02		-0.26		-0.02
Photochemical ozone, veg.	m^2*ppm*hours	-2.29E+11	-2.04E+11	-3.44	-3.06	-0.43	-0.38
Respiratory inorganics	kg PM2.5-eq		-6.59E+06		-0.77		-0.14
Respiratory organics	pers*ppm*h	-2.81E+07	-2.25E+07	-3.89	-3.12	-0.50	-0.40
Endpoint (damage) categories:							
Impact on ecosystems	species-weighted m^2* years	-1.01E+10	-1.74E+10	-0.77	-1.33	-0.19	-0.32
Impacts on human well-being	QALY	-4.33E+02	-5.31E+03	-0.07	-0.86	-0.01	-0.15
Impacts on resource productivity	EUR	-5.97E+07	-1.54E+08	-0.38	-0.99	-0.06	-0.16
All impacts aggregated	EUR	-1.51E+09	-3.01E+09	-0.62	-1.22	-0.14	-0.27

In Table 4.13, the consequences of increasing milk yield from the present 5 900 kg/cow to 8 500 kg/cow are estimated, assuming the same on-farm land use, an increased import of feed to the farm, and a marginal biological efficiency of transforming feed into milk of 60 %. The effect on nitrogen turnover in the cows is estimated according to Nielsen & Kristensen (2001).

Per cow, more cereals and concentrates are needed to support the larger milk yield (0.64 kg per kg extra milk produced). This means that instead of being net cereal exporting, the dairy farms become net importers of cereals. The need for fertiliser N decreases due to the larger N import in feed, while this also leads to slightly larger N emissions per cow unit.

Table 4.13: Input-output relations if milk yield increases from 5 900 kg/cow to 8 500 kg/cow.

	Present (26 million dairy cows)		With larger milk yield (18 million dairy cows)	
	Per cow unit	**Per 10 Mg milk**	**Per cow unit**	**Per 10 Mg milk production**
Land use at dairy farm, ha	1.25	2.1	1.25	1.5
Inputs, kg:				
Cereal	-	-	963	1 133
Soy meal	634	1076	1 055	1 241
Fertiliser N	159	269	150	176
Fertiliser P	5	9	5	6
Mineral P	4	6	3	4
Products, kg:				
Cereals	280	474	-	-
Beef, live-weight	338	573	338	397
Emissions, kg:				
Methane	168	284	182	215
Ammonia	44	74	48	56
N_2O	14.1	23.9	14.4	16.9
Nitrate	306	520	330	388
Phosphate	0.83	1.40	0.89	1.05

However, per kg milk, methane emission from the dairy farms is reduced by 24 %, and land use at the farm is reduced by 29 %. At the same time, emissions of ammonia, N_2O and nitrate per kg milk are reduced. On the other hand, this improvement at the specialised dairy farm is offset by the concomitant increase in feed requirement and reduction in beef output (30 % less beef produced, due to a smaller number of calves born and a smaller number of cows slaughtered), leading to increased emissions from feed production and from the induced additional beef production from suckler cows necessary to keep meat output unaltered. The net effect for methane emissions is a mere 4 % of the emissions from the dairy farms, and this is further counteracted by a net increase in CO_2 emissions, so that the net effect on global warming is negligible. Worse, the reduction in emissions is accompanied by a significant increase in area and energy requirement for feed production, so that for most impact categories the intensification leads to an increase in impacts, as can be seen in Table 4.14.

Table 4.14: Annual change in impacts by intensification of dairy farming. Negative values signify an improvement (reduced impact).

Impact category	Unit	Improvement potential		
		In units of each impact category	In % of total impacts for meat and dairy products	In % of total impacts in EU-27
Midpoint categories:				
Acidification	m^2 UES	-2.80E+08	-0.30	-0.07
Ecotoxicity, aquatic	kg-eq. TEG water	6.66E+12	4.65	2.17
Ecotoxicity, terrestrial	kg-eq. TEG soil	1.10E+10	1.82	0.12
Eutrophication, aquatic	kg NO_3-eq.	-2.54E+08	-2.86	-0.84
Eutrophication, terrestrial	m^2 UES	-2.42E+09	-0.62	-0.24
Global warming	kg CO_2-eq.	-1.79E+09	-0.27	-0.04
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	1.09E+08	7.88	0.63
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	2.23E+07	1.95	0.13
Mineral extraction	MJ extra	1.11E+08	2.11	0.12
Nature occupation	m^2 arable land	6.66E+10	6.83	2.44
Non-renewable energy	MJ primary	4.37E+11	4.99	0.31
Ozone layer depletion	kg CFC-11-eq.	7.35E+03	3.85	0.25
Photochemical ozone, vegetation	m^2*ppm*hours	2.53E+11	3.80	0.47
Respiratory inorganics	kg PM2.5-eq.	2.60E+07	3.05	0.54
Respiratory organics	person*ppm*hours	2.30E+07	3.18	0.41
Endpoint (damage) categories:				
Impact on ecosystems	Species-weighted m^2*years	5.77E+10	4.41	1.07
Impacts on human well-being	QALY	1.86E+04	3.00	0.52
Impacts on resource productivity	EUR	5.00E+08	3.22	0.53
All impacts aggregated	EUR	1.00E+10	4.07	0.89

A further side effect of the increased milk yield is an increased risk of poorer udder health resulting in higher veterinary costs, as well as being an animal welfare issue. According to Østergaard & Neimann-Sørensen (1989), the veterinary costs from a similar increase in milk yield were estimated to increase by 20 % per cow. Calculated per kg milk produced, however, veterinary costs decrease slightly.

As the negative effects of intensification of dairy farming in general outweigh the benefits, this option will not be considered further. Conversely, it could be argued from the results in Table 4.12 that an extensification of the highly specialised Danish and Swedish milk farms would be an environmental improvement option. However, the importance of this is limited by the relatively low proportion of the total EU-27 milk yield that is presently supplied by high-yielding dairy farms.

Nevertheless, the result in Table 4.12 is an argument for restricting further specialisation in dairy farming, or at least to remove any existing intensives for such specialisation.

4.3.5 Improvement options for area use by animal husbandry

The land area included in this study as contributing to 'nature occupation' is only the intensively used area. Besides the land for production of cereal and protein crops to be used as input in the cattle production, the cattle systems use land for intensive grazing and production of roughage. Although extensive pasture also occupies areas that could support natural ecosystems, extensive grazing is generally regarded as a form of landscape maintenance, independent of the demand for beef and dairy products. Thus, it seems less relevant to consider a reduction of area use for beef production based on suckler system or grazing steers.

The dairy systems are often quite intensive systems, where cows are in-house at least part of the day. This opens a theoretical option for intensification of the systems, whereby an increased crop yield can be obtained through intensive cropping of forage rather than grazing (and the area correspondingly reduced). This is in particular an option to consider in the west and central systems where rotational grass is used for grazing. In the UK and the lowland systems, which are based on permanent pasture, this option will be difficult to implement. Also, cow health and welfare aspects may be compromised when omitting grazing. Therefore, such improvement options have not been studied in detail.

4.4 Improvement options for retailing and shopping

Shopping by car contributes to more than 10 % of the life cycle impact of photochemical ozone (the human impacts of which is represented by the impact category 'respiratory organics'). The main contributing substance to the photochemical ozone impact from car driving is carbon monoxide.

Emission reductions from car driving are the subject of the parallel study by JRC-IPTS on environmental improvement potentials of cars (IMPRO-car) ([1]), and will therefore not be dealt with here.

What this study does look at is the options for reducing car driving for shopping by providing alternative distribution systems. Delivery services offered by supermarkets may replace at least some of the shopping trips by private car, if combined with remote ordering (e.g. through Internet). A larger capacity utilisation of the delivery vehicle leads to a net result of fewer kilometres driven per consumer.

Currently, Internet shopping for groceries shows mixed success, and is not yet very widespread. Internet shopping is rapidly gaining ground for household durables, mainly motivated by time and cost savings, and increased options for comparing products. With respect to groceries, particular obstacles are the customers' desire to assess the quality of fresh goods by physical inspection and requirements for speedy delivery and flexible delivery times (Hansen 2005). Particular incentives may therefore be required to accommodate these concerns, such as more standardised quality, and 'satisfaction or money back / new delivery' guarantees.

([1]) http://ipts.jrc.ec.europa.eu/publications/pub.cfm?id=1564

As a possible means to strengthen high-quality service, a requirement to participate in a quality scheme (like the Danish ‘Smiley’ scheme ([2])), possibly with public support, has been suggested, or the setting up of a competing public delivery service. A delivery service that is not bound to one particular supermarket chain (such as a public one) may have the advantage of providing consumers with a larger selection and a stronger price competition, thus providing additional incentives for consumers to participate. To strengthen the acceptance, it might be an option to require delivery services to be offered for free (limited to customers within a certain distance from the shop and, for example, to purchases over a certain value or for a maximum number of deliveries within a period), so that the cost of the service is distributed evenly over all customers.

As an achievable target, if suitable incentives are put into place, it is assumed that 25 % of customer trips can be replaced by a delivery service, taking into account that some consumers will not take part and that small ‘topping-up’ shopping will still be done by personal visits to a local store. Assuming that delivery vans, with a required average emission per vehicle-km not exceeding that of an average passenger car, can serve 50 customers on a 50 km round trip, thereby saving eight trips by public transport and 42 car trips with an average distance of 4 km, a saving of 18 % is obtained on all shopping kilometres by car, and a reduction in public transport for shopping of 25 %.

A delivery service will not apply to meat and dairy products alone, but will cover all food shopping, and the reduction percentage is therefore applied to the overall data for food shopping trips from Chapter 8.4.9, i.e. to 6.7 % of all private car-km and 2.6 % of all vehicle-km by public transport. The resulting environmental impact reduction potential is shown in Table 4.15. The main contributions to the overall impact reduction are from global warming (48 %) and respiratory inorganics (37 %).

Technically, only 42 % of the improvement potential in Table 4.15 is related to meat and dairy products (see Chapter 8.4.8); the rest is a rebound effect for the remaining groceries (see also Chapter 4.7). This also partly explains why the improvement for global warming in Table 4.15 can be nearly 2 % of the total impacts for meat and dairy products, although shopping constitutes only 2.6 % of the global warming impact of meat and dairy products in Figure 3.1. The other part of the explanation is that the improvement in Table 4.15 includes also the upstream processes, such as car manufacture, while the percentages for shopping in Figure 3.1 represent the emissions from the shopping process itself, i.e. the emissions from driving only.

([2]) http://www.uk.foedevarestyrelsen.dk/Inspection/Smiley/

Table 4.15: Improvement option 9: Annual improvement potential by home delivery of groceries in EU-27. Negative values signify an improvement (reduced impact).

Impact category	Unit	Improvement potential		
		In units of each impact category	In % of total impacts for meat and dairy products	In % of total impacts in EU-27
Midpoint categories:				
Acidification	m^2 UES	-6.72E+08	-0.71	-0.18
Ecotoxicity, aquatic	kg-eq. TEG water	-1.69E+11	-0.12	-0.05
Ecotoxicity, terrestrial	kg-eq. TEG soil	-2.60E+10	-4.31	-0.28
Eutrophication, aquatic	kg NO_3-eq.	-1.57E+07	-0.18	-0.05
Eutrophication, terrestrial	m^2 UES	-2.28E+09	-0.59	-0.23
Global warming	kg CO_2-eq.	-1.24E+10	-1.86	-0.26
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	-2.90E+07	-2.09	-0.17
Human toxicity, non-carc.	kg C_2H_3Cl-eq.	-1.82E+07	-1.59	-0.11
Mineral extraction	MJ extra	-2.41E+08	-4.58	-0.26
Nature occupation	m^2 arable land	-1.08E+09	-0.11	-0.04
Non-renewable energy	MJ primary	-3.74E+11	-4.27	-0.27
Ozone layer depletion	kg CFC-11-eq.	-4.01E+03	-2.10	-0.13
Photochemical ozone, vegetat.	m^2*ppm*hours	-3.33E+11	-4.99	-0.62
Respiratory inorganics	kg PM2.5-eq.	-1.17E+07	-1.38	-0.24
Respiratory organics	person*ppm*hours	-3.60E+07	-4.99	-0.64
Endpoint (damage) categories:				
Impact on ecosystems	Species-weighted m^2*years	-8.86E+09	-0.68	-0.16
Impacts on human well-being	QALY	-8.69E+03	-1.40	-0.24
Impacts on resource productivity	EUR	-2.83E+08	-1.82	-0.30
All impacts aggregated	EUR	-2.17E+09	-0.88	-0.19

If home delivery is widespread, this may also involve a reduction in retail space, as warehouses can replace some shops. This has not been included in the above calculations. Also, deliveries can be made more frequently than private shopping, without significant increase in vehicle-kilometres, which may contribute to reduce storage loss in households (further discussed in Chapter 4.6).

Emissions from car driving are larger for short trips with a cold engine than for longer trips, which means that reductions in shopping by car will mean a larger improvement than that shown by the proportional reduction in kilometres. This reduction in cold engine emissions has not been quantified.

The uncertainty on the emission reductions is assessed to be +/-35 % around the mean values, and is dominated by the uncertainty on the assumptions on the reduction in driving that can be achieved, and the degree of implementation. The uncertainty on the aggregated improvement potential is dominated by the uncertainty on the characterisation factors (see Table 9.4) resulting in a span in the annual improvement potential from EUR 1 000 million to EUR 5 000 million. Other important sources of uncertainty for the overall benefit of this improvement option are the direct costs and the value of the time saving in the households, which are further discussed in Chapter 5.2.9.

Even without alternative distribution systems, the frequency of shopping, and thereby the overall distance driven, may be reduced by providing better tools for planning of food purchases and through increased shelf-life of the products. These options are described in more detail in relation to food loss in Chapter 4.6.

4.5 Improvement options for electricity consumption

Electricity production accounts for 15 % of the global warming impact potential from the life cycle of meat and dairy products, see Figure 3.1. This underlines the importance of the electricity supply system, also for the meat and dairy product system. However, the EU electricity supply is the subject of other ongoing policy preparing research by JRC-IPTS and others, and will therefore not be dealt with here.

What this study does look at is the options for reducing electricity consumption related to meat and dairy products.

As shown in Table 3.6, storage of food in the households accounts for approximately 20 % of the total electricity consumption in the life cycle of meat and dairy products.

During 2007, parallel to this study, preparatory studies under the eco-design of energy-using products Directive (EuP Directive) were ongoing. The preparatory studies for cold appliances (*'Preparatory Studies for Eco-design Requirements of EuPs. Lot 13: Domestic Refrigerators & Freezers')* include economic, stock and market analyses and assessment of improvement options. None of the documents from the Preparatory Studies on Refrigerators & Freezers were finished at the moment of writing this study, but a number of draft versions were available at http://www.ecocold-domestic.org/. Design options and technological innovations are described in a first draft version from the project (Presutto & Mebane 2007). The preliminary results of the Preparatory Studies have been used in this report whenever appropriate. Calculations on consumer prices, savings on electricity and the life cycle costs for refrigerators and freezers will be made in the EuP Preparatory Studies for cold appliances (Mebane et al. 2007), but is not included in the draft reports available at the time of writing. This study has therefore performed its own calculations.

Two options were considered: a scheme for early replacement of old appliances by highly energy efficient new appliances (A+/A++), and a requirement that prevents consumers from buying other than new A+/A++ appliances. The combination of the two measures was also considered.

As a starting point, the maximum potential energy savings are calculated from replacing the remaining applications of different age classes still in service by new A++ appliances (Table 4.16).

Table 4.16: **Maximum annual savings potential of replacing a generation of old cold appliances with new A++ appliances**

Year of manufacture	Estimated average annual kWh per 100 litres for new appliances sold in period[1]	Savings potential per 100 litres[2]	Share of generation still in service[3] (%)	Share of total population (%)	Annual savings potential of generation[4] (%)	Accumulated annual savings potential[4] (%)
1975-1980	260	200	4	2.6	3.0	3.0
1981-1985	235	175	13	6.2	6.5	9.5
1986-1990	210	150	31	12.4	11.0	20.6
1991-1995	190	130	56	19.4	15.1	35.6
1996-2000	160	100	79	25.1	15.0	50.6
2001-2005	130	70	93	28.3	11.8	62.4
2006	115	55	99	5.9	1.9	64.3

1. Based on Rüdenauer & Gensch (2005). Average for all types of cold appliances.
2. Relative to the annual consumption of 'average A++ appliances', set to 60 kWh per 100 litres, based on Rüdenauer & Gensch (2005), an estimate for all types of cold A++ appliances (ranging from 54 kWh/100 litres to 84 kWh per 100 litres).
3. Estimate based on age distribution of refrigerators and refrigerator/freezers sent to recycling from Calabrese (2004) combined with the average lifetime of cold appliances of 17 years from Rüdenauer & Gensch (2005).
4. Relative to total annual current consumption of 'average cold appliances' (168 kWh per 100 litres), which is calculated as the sum of the 'estimated average kWh per 100 litres for new appliances sold in period' for each period multiplied by the 'share of total population'.

Figure 4.1 shows the effects of introducing an early replacement scheme and of introducing a requirement that prevents consumers from buying other than new A+/A++ appliances. The calculations are based on replacement of all existing cold appliances from before 2000 by new A+/A++ appliances over a five-year period 2009 to 2013, one-fifth each year, replacing the oldest appliances first, assuming an average energy requirement of 75 kWh/100 litres for A+/A++ appliances. Furthermore, it has been assumed that the efficiency of A++ appliances is close to reaching the optimal technical limit for efficiency, and hence the appliances used in 2040 are assumed to be A++. This is the reason for both curves in Figure 4.1 approaching 60 kWh per 100 litre appliance in 2040. The EuP Directive preparatory studies for cold appliances might show that it will be possible to reach even lower energy consumption than the existing A++ today, but the current draft documents from the study do not contain exact estimates on energy savings for the technology innovations mentioned in the preparatory studies. It should be noted that the figure of kWh per 100 litres varies quite a lot between refrigerators, refrigerator-freezers and freezers. The calculations here are based on an estimated average of cold appliances.

As can be seen from Figure 4.1, the largest reductions in the electricity consumption can be obtained by combining an 'early replacement scheme' with a legislative requirement that prevents consumers from buying other than new A+/A++ appliances (the green line compared to the dark blue line in Figure 4.1).

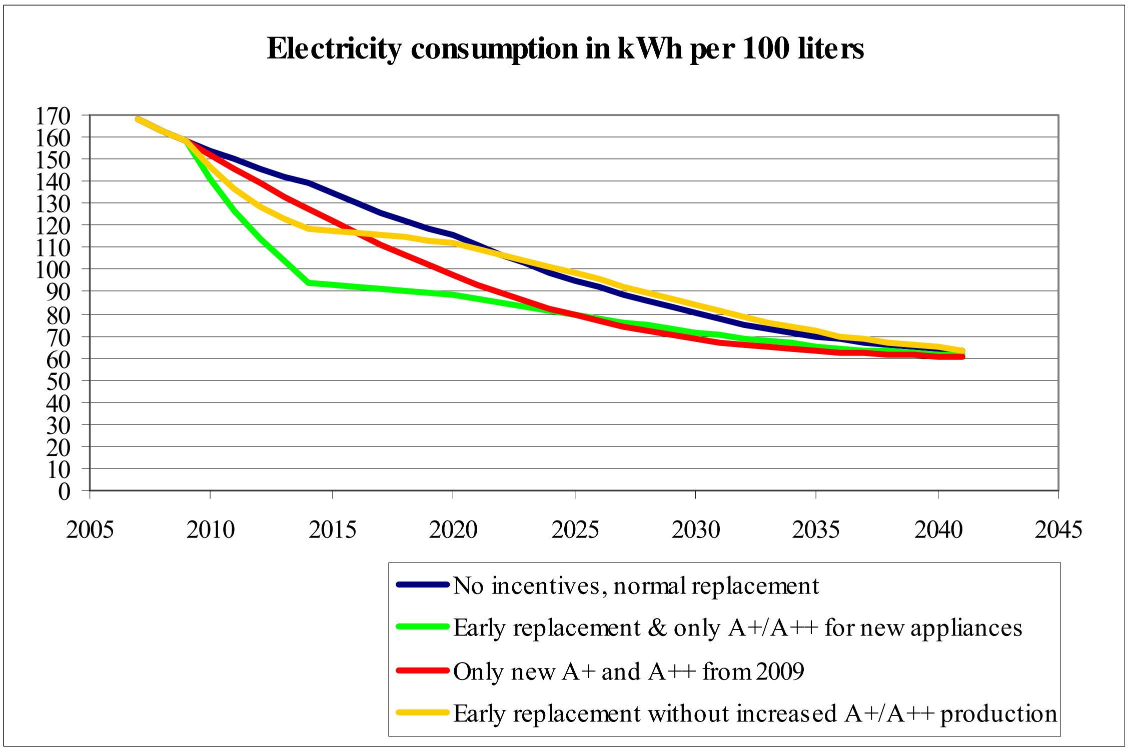

Figure 4.1: **Electricity consumption by cold appliances for four scenarios.**

Introducing an early replacement scheme will require increased production of appliances for a period (in the calculations here a five-year period from 2009-2013). This is shown in Figure 4.2. The 'natural replacement' is calculated using an average lifetime of 17 years (i.e. 1/17 = 5.9 %), assuming that the market for cold appliances in Europe is saturated and not expanding. In the EuP Preparatory Studies for cold appliances (Mebane et al. 2007) the estimated lifetime for refrigerators is set to 15 years (with sensitivity analysis for 10, 12 and 17 years). As can be seen from Figure 4.2, the demand will drop significantly after the five-year period, because most consumers will have highly efficient, new cold appliances with an average lifetime of 14-17 years. It would be a serious challenge for the cold appliance industry to cope first with a significant rise in demand and then a serious decrease, especially if the industry has had large investments in technology in order to transform into production of A+ and A++ appliances only.

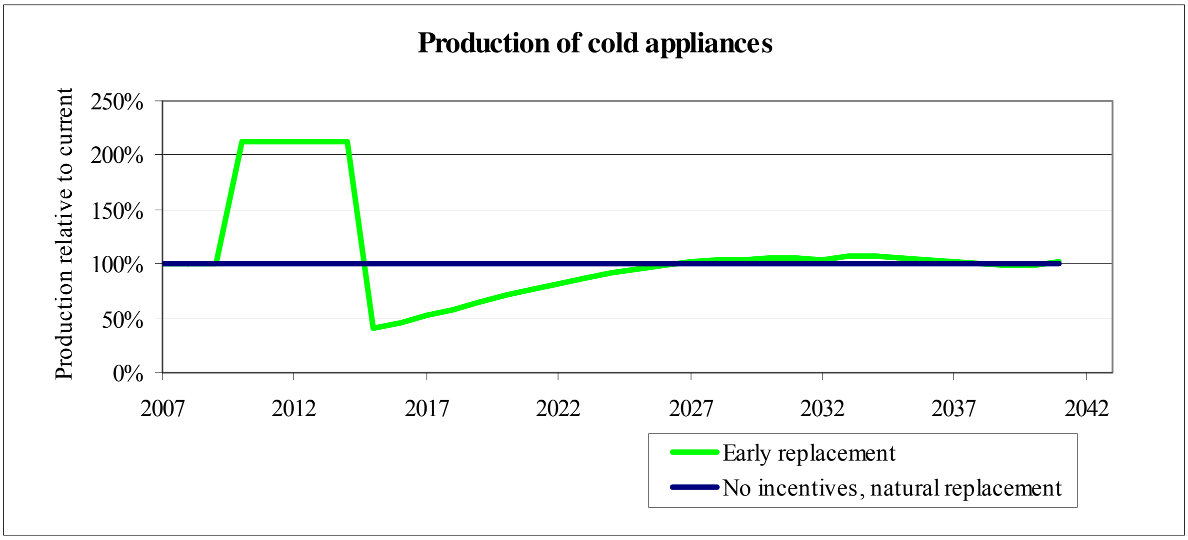

Figure 4.2: Demand for A+/A++ cold appliances in an early replacement scheme compared to natural replacement.

In addition to these possible capacity problems, an 'early replacement scheme' requires an economic incentive for the consumer to discard the appliances 'prematurely'. Since the consumer is in a situation where the alternative is to keep the old appliance, the incentive must be larger than the electricity costs of the current appliance. A 270-litre appliance from the period 1991-1995, with an annual electricity savings potential of 350 kWh (130 kWh/100 litres according to Table 4.16) could give a monthly saving on the electricity bill of EUR 4.1 when exchanged for a new A++ appliance (with a consumer electricity price of 0.14 EUR/kWh, also used by Mebane & Piccinno 2006). This monthly saving could pay for an A++ appliance of EUR 508 (calculated by adding average trade margins and VAT to the EUR 237 EUR production cost of Mebane & Piccinno 2006) in little more than 12 years (with 3 % interest). However, a newer appliance from the period 1996-2000, with an annual electricity savings potential of 270 kWh (100 kWh/100 litres according to Table 4.16) would only give a monthly saving of EUR 3.15 when exchanged for a new A++ appliance. This monthly saving could not pay for the new appliance within its lifetime. Thus, without public economic support, an 'early replacement scheme' can only target appliances from before 1996. Furthermore, the above amortisation calculations do not take into account that the early replacement involves a loss of appliance lifetime, i.e. that the new appliances will need replacement earlier than under 'natural' replacement. Before accepting an early replacement, a rational consumer would also require a compensation for this loss of appliance lifetime. The loss of appliance lifetime is equal to the additional production requirement and can be estimated at 2.7 years (the net area between the two lines in Figure 6.2) or 16 % of a 17 years lifetime. With an appliance cost of EUR 508, this loss has a value of approximately EUR 80. If this compensation were to be added to the appliance price, the payback time for replacing the 1991-1995 appliance would be close to 15 years.

Considering these complicating issues for implementing an 'early replacement scheme', the suggested improvement option is restricted here to an 'A+/A++ appliance scheme', where consumers are offered A+/A++ appliances at approximately the same price as an average appliance, in exchange for returning an old appliance ([3]).

([3]) A cost-neutral approach would be to let the consumers pay the price difference over the electricity bill year by year, corresponding to the cost of the saved electricity. Purchase of A+/A++ appliances instead of 'average new appliances', will give electricity savings of approximately 40 kWh per 100 litres per year (using the same assumptions as for Tables 8.1 and 8.2 above) or 108 kWh per year per appliance (with the average size of 270 litres). With a consumer electricity price of 0.14

The average annual net improvement potential in Table 4.17 is calculated over the period 2010 to 2040, i.e. a 30-year period. The net improvement potential is an accumulated electricity saving of 340 kWh per 100 litres (the area between the 'no incentives' and 'only A+/A++' lines in Figure 4.1) minus a corresponding 10 % increase in raw material inputs and production activities for the more efficient appliances, calculated from the proportions between production costs and electricity savings in Mebane & Piccinno (2006). The aggregated improvement potential is dominated by the contributions from the impact categories global warming (72 %) and respiratory inorganics (25 %).

An 'A+/A++ appliance scheme' will not apply to meat and dairy products alone. Formally, only 42 % of the improvement potential calculated in Table 4.17 is related to meat and dairy products (Chapter 8.4.9); the rest is a rebound effect for the remaining cold storage capacity; see also Chapter 4.7.

The potential emission reductions are well documented and the uncertainty on the percentage reductions in Table 4.17 is estimated to +/-10 % around the mean values. The uncertainty on the aggregated improvement potential is dominated by the uncertainty on the characterisation factors for global warming and respiratory inorganics (see Table 9.4), resulting in a span in the annual improvement potential from EUR 300 million to EUR 1 800 million. Another important source for the overall uncertainty on this improvement option is the direct costs, as discussed in Chapter 5.2.10 and Chapter 5.4.

Table 4.17: Improvement option 10: Average annual improvement potential for the period 2010-2040 of a 'consumers buy only new A+/A++ appliances' scheme in EU-27. Negative values signify an improvement (reduced impact). Only midpoint categories contributing more than 0.01 % change are shown.

Impact category	Unit	Improvement potential		
		In units of each impact category	**In % of total impacts for meat and dairy products**	**In % of total impacts in EU-27**
Midpoint categories:				
Acidification	m^2 UES	-3.42E+08	-0.36	-0.09
Ecotoxicity, terrestrial	kg-eq. TEG soil	-2.21E+09	-0.37	-0.02
Eutrophication, terrestrial	m^2 UES	-4.46E+08	-0.11	-0.04
Global warming	kg CO_2-eq.	-6.59E+09	-0.98	-0.14
Non-renewable energy	MJ primary	-9.48E+10	-1.08	-0.07
Photochemical ozone, vegetat.	m^2*ppm*hours	-2.81E+10	-0.42	-0.05
Respiratory inorganics	kg PM2.5-eq.	-2.74E+06	-0.32	-0.06
Respiratory organics	person*ppm*hours	-2.29E+06	-0.32	-0.04
Endpoint (damage) categories:				
Impact on ecosystems	Species-weighted m^2*years	-3.94E+09	-0.30	-0.07
Impacts on human well-being	QALY	-2.07E+03	-0.33	-0.06
Impacts on resource productivity	EUR	-4.99E+07	-0.32	-0.05
All impacts aggregated	EUR	-7.57E+08	-0.31	-0.07

EUR/kWh (also used by Mebane & Piccinno 2006), the consumer can thus save 15 EUR/year or 1.25 EUR/month by buying a new 'A++ appliance' instead of an 'average new appliance'. The difference in cost of the two appliances can be estimated to EUR 90, by adding average trade margins and VAT to the EUR 42 unit manufacturing costs difference from Mebane & Piccinno (2006). With a 3 % annual interest, and a monthly payment of EUR 1.25, the pay back time of this price difference would be less than seven years.

The average annual net improvement potential of an 'A+/A++ appliance scheme', as outlined in Table 4.17 , is not directly comparable to the impacts from the other more 'permanent' improvement options, since the number of years over which the improvement is calculated is arbitrary, and will influence how well the accumulated improvement will compare to the more 'permanent' improvements.

Compared to the household processes, the industrial processes – farming, food industry, retail, and catering – responsible for 14 %, 19 %, 6 % and 6 %, respectively, of the electricity use in the life cycle of meat and dairy products, should be under a larger pressure for efficient use of electricity. Nevertheless, experience has shown that efficiency improvements are still possible when specifically targeted. Based on experts' assessment, a 5-15 % reduction is regarded as realistic, corresponding to 3-6 % of the overall electricity consumption in the life cycle of meat and dairy products, or 6-12 TWh.

For the calculation of the improvement potential in the industrial processes here, it is assumed that the electricity saving potential is nine TWh.

The main uncertainty on this improvement option is the extent to which reduction options can be identified and implemented. For the aggregated impact reductions, the uncertainty is dominated by the uncertainty on the characterisation factors for global warming and respiratory inorganics.

4.6 Improvement options for household processes

Improvements in electricity consumption for household processes were dealt with in the previous sub-chapter. This section looks at the options for reducing the household loss of meat and dairy products, which is estimated to be in the range of 20 % (see also Chapter 8.4.10). When comparing to a loss of 1.5 % at full-service restaurants (Jones 2005), and considering that a large part of the loss is storage loss, it should be possible to reduce household losses considerably though better planning of food purchases, i.e. ensuring that the right amounts are available in the households at the right time.

According to Unilever (2005), nearly 75 % of dinner decisions are made the same day. Also, many consumers only think a few days ahead when shopping. Only 58 % of all shopping trips are planned, and bigger trips are likelier to be planned (Unilever 2005).

Consumer's planning may be improved by providing personalised shopping lists based on the composition of previous purchases and the known shelf-life of purchased items. The shopping lists could be combined with an Internet-based ordering system and the delivery system mentioned in Chapter 4.4. More advanced, an electronic 'menu planner', with a large number of recipes with options for personalised preferences, may generate shopping lists from chosen menus and may suggest menus based on previously purchased food well in advance of its 'best-before' date. If linked to an Internet-based ordering and delivery system, very little effort would be required of the user.

It is estimated that food loss can generally be halved by application of better planning tools, and that these tools will be accepted by 25 % of consumers, i.e. resulting in an average 12.5 % reduction in food waste, or 2.5 % of the total amount of meat and dairy products purchased by households. The resulting savings in environmental impacts are shown in Table 4.18. These savings also include the savings in treatment of food packaging waste and food waste. The uncertainty is assessed to be +/-50 % around the mean value.

The main contributions to the aggregated impact improvement come from reductions in nature occupation (66 %), respiratory inorganics (17 %) and global warming (14 %).

For the aggregated improvement potential, the uncertainty is dominated by the uncertainty on the characterisation factors for nature occupation and respiratory inorganics; see Table 9.4. The resulting span for the improvement potential is EUR 2 600 million to EUR 11 000 million annually. Other important sources of uncertainty for the overall benefit of this improvement option are the associated cost savings and the possible associated health benefits, which are further discussed in Chapter 5.2.12.

Table 4.18: Improvement option 12: Annual improvement potential by reducing household loss of meat and dairy products in EU-27 by better household meal planning tools (12.5 % reduction equal to 2.5 % of all meat and dairy products). Negative values signify an improvement (reduced impact).

Impact category	**Unit**	**Improvement potential**		
		In units of each impact category	**In % of total impacts for meat and dairy products**	**In % of total impacts in EU-27**
Midpoint categories:				
Acidification	m^2 UES	-1.70E+09	-1.79	-0.45
Ecotoxicity, aquatic	kg-eq. TEG w	-2.01E+12	-1.41	-0.65
Ecotoxicity, terrestrial	kg-eq. TEG soil	-8.45E+09	-1.40	-0.09
Eutrophication, aquatic	kg NO_3-eq.	-1.48E+08	-1.67	-0.49
Eutrophication, terrestrial	m^2 UES	-7.23E+09	-1.86	-0.73
Global warming	kg CO_2-eq.	-1.17E+10	-1.75	-0.25
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	-2.47E+07	-1.78	-0.14
Human toxicity, non-carc.	kg C_2H_3Cl-eq.	-1.73E+07	-1.52	-0.10
Mineral extraction	MJ extra	-7.20E+07	-1.37	-0.08
Nature occupation	m^2 arable land	-2.49E+10	-2.55	-0.91
Non-renewable energy	MJ primary	-1.21E+11	-1.38	-0.09
Ozone layer depletion	kg CFC-11-eq.	-3.13E+03	-1.64	-0.10
Photochemical ozone, veg.	m^2*ppm*hours	-1.16E+11	-1.74	-0.22
Respiratory inorganics	kg PM2.5-eq.	-1.62E+07	-1.91	-0.34
Respiratory organics	pers*ppm*hours	-1.25E+07	-1.74	-0.22
Endpoint (damage) categories:				
Impact on ecosystems	species-weighted m^2*years	-2.98E+10	-2.27	-0.55
Impacts on human well-being	QALY	-1.18E+04	-1.90	-0.33
Impacts on production	EUR	-2.93E+08	-1.89	-0.31
All impacts aggregated	EUR	-5.35E+09	-2.18	-0.48

The options for reducing loss of fresh products by increasing their shelf life were also considered. The proportion of fresh meat is at least 36 % (Danmarks Statistik 2003) and the proportion of fresh milk is around 11 % of all dairy products (42 % of all drinking milk according to Giffel et al. 2006).

For meat, increased shelf life can be obtained by:

- Vacuum packing, which seals the meat in plastic bags from which air has been expelled. The bags minimise both gas and moisture permeability and they act as a barrier to prevent the meat surface from exposure to external oxygen. The method has a limited use for display packages of meat, as the meat looses the 'fresh meat' colour and gets a dark purple colour, which is considered to be an important factor for the attractiveness of the meat to the consumer. The lack of oxygen inhibits the growth of bacteria, which cause the meat to deteriorate. Vacuum packed primal cuts stored at 0°C should have a shelf life of four-eight weeks. Meat held in vacuum pack for long-term storage must be kept at a temperature of 3°C or less.
- Modified Atmosphere Packaging (MAP), where the air in the pack is replaced by a modified atmosphere. The shelf life depends on the composition of the modified atmosphere and experiments for increasing shelf life with different modified atmospheres are ongoing. Meat packed in modified atmosphere will typically have a shelf life of five-eight days if kept under 2°C. Besides the shelf life and colour stability of the meat, a lot of other factors vary with the packaging methods, e.g. tenderness, drip loss and cooking loss.

An essential problem is that the shelf life of meat is very dependent on storage temperature. At the household, the typical temperature in the refrigerator will be around 4-6°C rather than 0-2°C. As the shelf life is significantly reduced with higher temperatures, the consumer will not be able to store meat in refrigerators for 7-10 days, regardless of packaging. Some new refrigerators are constructed with a '0° zone'. According to Miele (2006), the life of milk and fresh meat is increased from three to seven days in their 0° zone. However, the low temperature involves extra electricity consumption by the refrigerator.

For milk, new technologies have made extended shelf life possible. Ultra-pasteurisation and microfiltration are the two major processing technologies for extending the shelf life of chilled dairy products. Citing Giffel et al. (2006): '*For shelf-life extension, various technological opportunities are available, e.g. high temperature-short time heat treatment, microfiltration or bactofugation combined with pasteurisation. These technologies can be used to overcome the off-flavour formation and physical and textural stability problems traditionally associated with high heat processed dairy products.'* According to Giffel et al. (2006), the microfiltered milk in the United Kingdom has a shelf-life of 23 days. If combined with a heat-treatment and/or clean filling, a shelf life of up to circa 32 days can be achieved under refrigerated conditions. Microfiltered milk has captured 11 % of the market in the United Kingdom and is estimated at 20-25 % in Germany (Giffel et al. 2006).

The European Food Information Council (EUFIC) (2006) writes: 'There is also a new development in membrane technology manufacture, which leads to a similar hygienic safety as 'thermisation' of skimmed milk at 50°C. This will allow the commercialisation of new milk, which can be stored at room temperature for six months and with a taste similar to fresh pasteurised milk.'

The mentioned improvements in shelf life of fresh products can generally be expected to be autonomous developments with focus on the retail and consumer preferences. Once the packaging is opened and stored at the household, the increase in shelf life is limited. Due to the relatively low share of fresh products and the uncertainty in establishing an effect on household food losses, this study refrains from quantifying the possible impacts.

Consumer information may also be an important tool to prevent food being discarded because of misconceptions about freshness, colour, texture, and food safety issues. Food industry representatives have indicated to us that in support of this, it may be useful to perform a review of national legislations to identify possible technical barriers that increase food loss or hamper the implementation of technologies that improve shelf life. Examples could be too tight requirements on 'preferably consumed before' dates, perverse measuring standards, and demands for what may be labelled 'fresh'.

Increased durability and better planning may also lead to a reduction in the need for shopping; see Chapter 4.5, especially the 'topping-up' shopping. Also this reduction is expected to be small and has not been quantified, since there is a lack of data on how many of the shopping trips are actually determined by spoiled products, as well as on the distance and transport mode for 'topping-up' trips, which could be expected to be more often shorter and performed by means other than cars, than the average shopping trip.

4.7 Rebound effects, synergies and dysergies of the improvement options

4.7.1 Rebound effects

Rebound effects are the derived changes in production and consumption when the implementation of an improvement option liberates or binds a scarce production or consumption factor, such as:

- money (when the improvement is more or less costly than the current technology);
- time (when the improvement is more or less time consuming than the current technology);
- space (when the improvement takes up more or less space than the current technology); or
- technology (when the improvement affects the availability of specific technologies or raw materials).

Three types of rebound effects can be distinguished:

- Specific rebound effects, where production and consumption of the specific product in question changes;
- General rebound effects, where the overall production and consumption changes;
- Behavioural rebound effects, where the organisation of production and consumption changes, affecting both the specific product in question and other specific products.

Price rebounds occur when the improvement is more or less costly than the current technology. Costs of each improvement option are reported in Table 10.1. The price rebound may be divided in an effect on the consumption of meat and dairy products (the specific price rebound), and an effect on the general consumer expenditure (the general price rebound, also known as the income effect). The specific price rebound (for those improvement options that affect the price of the meat and dairy products) can be determined from the own-price elasticity of these products.

As reported by Seale et al. (2003), the own-price elasticity of meat and dairy products is relatively small, varying from -0.2 in high-income EU countries to -0.5 in the new EU member countries, i.e. implying that for every percentage increase in price of meat and dairy products, demand will fall with 0.2-0.5 %, with a weighted average value of 0.35 for EU-27. Consequently, this leaves 65 % of the change in price for the general rebound effect. The environmental consequences of the price rebound can thus be modelled by a 35 % change in the consumption of meat and dairy products and a 65 % change in average consumption. The environmental impacts of the average EU-27 household consumption are provided in Table 9.3. It would be more correct to use data for the marginal household consumption, but the environmental impacts of average consumption appears to be a reasonable proxy for those of marginal consumption (Thiesen et al. 2006). The change in consumer spending also implies corresponding shifts in production (second order effects), which may affect income distribution. When the total level of consumption and production is not affected, it is normal practice to ignore such second order effect, since the market mechanism evens out these effects in the long run.

Time rebounds occur when the improvement is more or less time consuming than the current technology. This is particularly relevant for changes in shopping patterns, where for example the home delivery of groceries will liberate time previously spent for shopping. The part of this time-saving that can be related to meat and dairy products are estimated in Chapter 10.10 to be 6.6E+9 hours annually in EU-27. Time elasticities, i.e. coefficients of time allocation between different activities when more or less time becomes available, are unfortunately scarce. Gershuny (2002) reports time elasticities based on United Kingdom time series, showing that approximately 40 % of the time released from unpaid work (such as shopping) will be used for other activities that may involve transport (going out, visits, sports), which implies that some of the saved car driving from shopping may be offset by increased car driving for these other activities. However, Gershuny's data cover only the situation where the saved time is released from unpaid work, not the elasticities within unpaid work, when for example the time saved on shopping is instead used to increase the time spent for childcare or cooking. Generally, the time used in leisure has remained relative stable over time (see Hoffstetter & Madjar 2003, and references therein), which implies that own-time elasticities within unpaid work may indeed be much larger than cross-elasticities to leisure activities. Most activities within unpaid work, apart from shopping, do not involve transport or other activities with high environmental intensities. In conclusion, it has been assumed that the time rebound stays within unpaid work and is environmentally neutral. The uncertainty in this assumption is an additional reason for carrying out in-depth pilot studies before implementing home deliveries of groceries as an environmental improvement option.

In addition to the time spent on different activities, there may also be an additional rebound effect from shifting in the timing of activities. The shifting of an activity in time may influence the overall activity pattern, even when there is no change in overall time spent on the specific activity. For example, while shopping is normally done in the daytime, Internet shopping may be done at night. The liberated daytime may have a different alternative use than the additional time used at night. In general, improvements that provide more flexible time usage, such as Internet shopping, are likely to release more time for out-going activities, which accentuates the possibility that the environmental improvement is partly offset by the rebound effect.

Space rebounds occur when the improvement takes up more or less space than the current technology. Relevant examples in the context of this study are the changes in agricultural land use, the additional road space released when car driving for shopping is reduced, the possible saving in kitchen space if home delivery was expanded to also supply ready-made meals, and the additional space requirement for cold storage when using A++ appliances (due to larger insulation thickness):

- The reduction in agricultural land use, or in its intensity of use, leads to increased availability for other land uses, either recreative (landscape accessibility) or commercial. No studies were found that quantify the net effect of increased availability of or accessibility to extensified agricultural land, i.e. what would be the alternative use of the liberated agricultural land, and what changes would occur in recreational behaviour.
- The additional road space released may reduce congestion (and consequent time usage and emissions) and subsequently induce increased traffic (and consequent emissions). As shown in the review article in TDM Encyclopedia (Victoria Transport Policy Institute 2007), generated traffic will in the long-term fill a significant portion (50-90 %) of added urban roadway capacity, thus reducing the environmental improvement proportionally.
- If home delivery of groceries were to include ready-made meals, some households would – when acquiring a new dwelling – possibly consider a smaller dwelling with less kitchen space, such as a simple kitchen niche such as those found for example in some New York apartments. This would probably apply to single person households. This development would not be likely to occur to any considerable extent within the time horizon of the present study (2020).

Technology rebounds occur when the improvement affects the availability of specific technologies or raw materials. As mentioned previously in this Chapter, many improvement options for the functional unit 'annual consumption of meat and dairy products in the EU-27' also affect meat and dairy products not consumed in EU-27 (i.e. exported products) or food production and consumption activities in general. These can be seen as specific technological rebound effects of implementing the suggested improvement options. These rebound effects are summarised in the notes to Table 4.20. In addition to these, the increased household availability of:

- home delivery of groceries may lead to decreased car-ownership for families where the need for a car for shopping is a determining factor for car-ownership. The consequent reduction in car driving will be subject to the same space rebound effects as described above.
- meal planning tools are likely to have significant effects beyond the modelled reduction in loss of meat and dairy products. Not only can a reduction in losses also be expected for other perishable foods, notably vegetables, but the meal planning tools may also stimulate a more healthy composition of meals in general, either as a simple effect of planning, or as a result of specifically integrating such features in the planning tools (see Huang et al. 2006). The additional reductions in food loss have been modelled by a 2.5 % reduction in the final consumption of 'agricultural products n.e.c.' in EU-27 NAMEA, but have refrained from modelling any impacts from reduction in healthcare expenditures.

In Chapter 4.8, the environmental potentials of all improvement options are summarised, both with and without inclusion of rebound effects.

4.7.2 Synergies and dysergies

The implementation of an improvement option may cause the improvement potential of other improvement options to increase (a synergic influence) or decrease (a dysergic influence). Each of the improvement options suggested in this chapter have been analysed for possible synergic or dysergic influences, and the identified synergies and dysergies are summarised in Table 4.19. The most important of these synergies and dysergies are quantified and the impact on the overall improvement potentials is reported in Chapter 4.8.

Table 4.19: Synergies '+' and dysergies '÷' between the suggested improvement options. Brackets indicate synergies and dysergies that are regarded as too small or too uncertain to quantify.

Influencing improvement option	Influenced improvement option											
	1	2	3	4	5	6	7	8	9	10	11	12
1. Catch crops					(÷)							÷
2. Cereal intensification	+				(+)							÷
3. Optimised protein feeding	(÷)			÷	÷							÷
4. Liquid manure pH reduction	(+)		÷		+							÷
5. Tightening of manure regulation	(÷)			+								÷
6. Copper reduction in animal diets												÷
7. Methane-reducing animal diets												÷
8. Liquid manure biogasification												÷
9. Home delivery of groceries												+
10. Cold appliances regulation												
11. Power saving in industry												
12. Household meal planning									+			

The effect of improvement option 1 (catch crops) will increase by 43 % if improvement option 2 (cereal intensification) is implemented, since the latter increases the area under intensified growing practise, which will benefit from the introduction of catch crops, from seven million ha to 10 million ha (see Chapter 4.2.1). Catch crops have their largest effect on leaching on well-fertilised land and land fertilised with manure (since the release of nutrient from the manure takes places over a longer period, including in autumn where the catch crops can take up the mineralised N).

The effect of improvement option 1 (catch crops) will tend to decrease if improvement option 3 (optimised protein feeding) is implemented, since the latter is already implying a smaller amount of N available for leaching, due to a smaller overall N excretion in manure; see Chapter 4.3.1. This dysergy is only relevant for the part of the manure that is used on areas where catch crops is a relevant measure.

The effect of improvement option 1 (catch crops) will tend to increase if improvement option 4 (liquid manure pH reduction) is implemented, since the latter will slightly increase the N available for leaching. This synergy is only relevant for the part of the manure that is used on areas where catch crops is a relevant measure.

The effect of improvement option 1 (catch crops) will tend to decrease if improvement option 5 (tightening of manure regulation) is implemented, since the latter is in itself an incentive to introduce catch crops and other measures that will reduce the soil N pool, thus leaving less N for the catch crops to take up. This dysergy is only relevant for the part of the manure that is used on areas where catch crops is a relevant measure.

There is a mutual dysergy between improvement options 3 and 4 (optimised protein feeding and liquid manure pH reduction), since both aim at reducing ammonia emissions from the manure. Optimised protein feeding reduces ammonia N in the liquid manure and thus decreases the potential of reducing ammonia emission through a pH-reduction of the liquid manure. This interaction is already modelled in Chapter 4.3.1. (Table 4.5 and Table 4.6). Furthermore, there are mutual relationships between these two improvement options and improvement option 5 (tightening of manure regulation), since the former improvement options influence the soil N pool, and thus influence the amount that can be affected by tightening of the regulation. For example, an optimised protein feeding (improvement option 3) reduces the amount of manure N (and thus reduces the impact of option 5), whereas a liquid manure pH-reduction (improvement option 4) increases amount of manure N (and the impact of option 5). Furthermore, improvement option 4 increases the proportion of $NH_4^+ - N$ in the manure, which improves the value of the manure as fertiliser (possibility of better timing between N available and plant growth) and makes it easier to obtain a high utilisation of the manure N (improvement option 5) without impairing crop yield. At the same time, implementing option 5 improves the impact of option 4. Option 4 is introduced to reduce ammonia-emission, but has as a side effect that more N is retained in the manure and thus subject to leaching since only a part of this is utilised by the crops. Implementing option 5 reduces this side effect, and thus interacts positively.

The effect of improvement option 5 (tightening of manure regulation) will tend to decrease if improvement option 1 (catch crops) is implemented, since the catch crops will already have reduced the N leaching, which is the focus of the manure regulation. This dysergy is only relevant for the part of the manure that is used on areas where catch crops is a relevant measure.

The effect of improvement option 5 (tightening of manure regulation) will tend to increase if improvement option 2 (cereal intensification) is implemented, since on the intensified area there is a larger potential for N leaching, which is the focus of the manure regulation.

There is a mutual synergy between improvement option 9 (home deliveries of groceries) and improvement option 12 (household meal planning), since they both require a more structured shopping behaviour, as in Internet-shopping, and facilitate the correspondence between planned meals and items purchased. It is likely that the willingness to accept the two improvement options is co-dependent, implying that if these improvement options are implemented simultaneously, the calculated improvement potentials are likely to be increased, possibly even doubled.

The effects of improvement options 1 to 8 (the agricultural improvements) will reduce the effect of improvement option 12 (household meal planning), since the agricultural production avoided by the reduced food loss will have less environmental impact.

4.8 Summary of all investigated improvement options

The improvement potentials of all suggested improvement options are summarised in Table 4.20. Only the shares of the improvements that can be ascribed to EU-27 consumption of meat and dairy products have been included, i.e. no rebound effects are included.

In Table 4.21, the same options are summarised, but now including rebound effects. The following rebound effects are quantified:

- The specific technological rebound effects on meat and dairy products not consumed in EU-27 (i.e. exported products) and food production and consumption activities in general, as described in the notes to Table 4.20 (denoted by italics).
- The specific technological rebound effect of implementing meal planning tools on the food loss of food products other than meat and dairy products, calculated as a 2.5 % reduction in the final consumption of fresh agricultural products.
- The traffic rebound effect, calculated at 70 % of the saved car driving for shopping.
- The specific and general price rebound effects resulting from the changes in consumer expenditure related to the costs reported in Table 10.1. The price rebound effects have been calculated on the changes in consumer surplus after accounting for the above-mentioned space and technology rebound effects, as reported in the notes to Table 4.21. The specific rebound effect of changes in the price of meat and dairy products has been modelled by a 35 % change in the consumption of meat and dairy products. The remaining 65 %, as well as price changes that do not affect meat and dairy products specifically, have been modelled by a change in average consumption.

In Table 4.22, all improvement potentials are added up, with and without quantified synergies and dysergies. The following synergies and dysergies are quantified:

- A 43 % increase in the effect of improvement option 1 (catch crops) as a result of having implemented improvement option 2 (cereal intensification), since the latter increases the area under intensified growing practise, which will benefit from the introduction of catch crops, from seven million ha to 10 million ha.
- The mutual dysergy between improvement option 3 (optimised protein feeding) and option 4 (liquid manure pH reduction). Both options reduce ammonia emissions from the manure. The dysergy is described in Chapter 4.3.1: the combined effects in Table 4.5 and Table 4.6 are smaller than the sum of the effects of the two measures. The effects of the dysergy have been divided equally between the two improvement options.
- The effect of improvement option 5 (tightening the regulation of manure application) has been adjusted for the effect of the interactions with improvement options 3 and 4 (optimised protein feeding and liquid manure pH reduction). The latter options influence the soil N pool, and thus influence the amount that can be affected by tightening of the regulation.
- A doubling of the effects of improvement options 9 and 12 (home deliveries of groceries and household meal planning), modelling a mutual doubling of the consumers' willingness to accept the two improvement options when both are implemented.
- A reduction in the effect of improvement option 12 (reduction of food loss through household meal planning) resulting from the smaller environmental

impact of the avoided agricultural production when all the agricultural improvements (options 1 to 8) have been implemented.

Table 4.20: Improvement potentials of the suggested improvement options (without rebound effects), in % of the total impact from consumption of meat and dairy products in EU-27. Only that part of the improvement which can be ascribed to the consumption in EU-27 has been included. Negative values signify an improvement.

	1	2	3	4	5	6	7	8	9	10	11	12
Midpoint categories:												
Acidification	-0.08	-0.44	-6.31	-10.89	-0.22			-0.86	-0.30	-0.15	-0.51	-1.79
Ecotoxicity, aquatic		-0.01	1.86			-52.06			-0.05			-1.41
Ecotoxicity, terrestrial		-0.11	0.79	-0.68	-0.70			-1.20	-1.81	-0.17	-0.74	-1.40
Eutrophication, aquatic	-0.95	-0.28	-3.71	2.28	-24.88			-0.06	-0.07			-1.67
Eutrophication, terrestrial	-0.09	-0.49	-7.27	-12.41	-0.23			-0.20	-0.25	-0.05	-0.16	-1.86
Global warming	-0.17	-0.28	0.84		-2.08		-2.16	-3.55	-0.78	-0.41	-1.39	-1.75
Human toxicity, carcinogens		-0.73	2.28		-0.16			-0.17	-0.88			-1.78
Human toxicity, non-carcinogens		-0.09		-1.60	-0.39	-5.11		-0.32	-0.67		-0.23	-1.52
Mineral extraction		-0.05	1.11		-0.35			-0.10	-1.93			-1.37
Nature occupation		-1.73	-1.55						-0.05			-2.55
Non-renewable energy		-0.48	1.72		-0.27			-2.97	-1.79	-0.46	-1.55	-1.38
Ozone layer depletion		-0.07	1.71		-0.40			-0.21	-0.88			-1.64
Photochemical ozone, vegetation		-0.72	1.21		-0.26		-2.93	-2.48	-2.10	-0.18	-0.61	-1.74
Respiratory inorganics	-0.05	-0.53	-2.48	-4.96	-0.33			-0.63	-0.58	-0.14	-0.47	-1.91
Respiratory organics		-0.69	0.98		-0.19		-3.31	-2.53	-2.09	-0.13	-0.47	-1.74
Endpoint (damage) categories:												
Impact on ecosystems	-0.06	-1.23	-0.98	-0.34	-0.77	-0.28	-0.65	-1.08	-0.28	-0.13	-0.43	-2.27
Impacts on human well-being	-0.05	-0.52	-2.35	-4.78	-0.37	-0.01	-0.06	-0.70	-0.59	-0.14	-0.48	-1.90
Impacts on resource productivity	-0.04	-0.55	-2.03	-4.38	-0.29	-0.01	-0.33	-0.81	-0.77	-0.14	-0.47	-1.89
All impacts aggregated	-0.06	-1.06	-1.30	-1.42	-0.66	-0.21	-0.52	-0.99	-0.37	-0.13	-0.44	-2.18

1. Catch crops on seven million hectares. *35 %* of the values in Table 4.1.
2. Improved growing practise and N intensification for cereal crops. *35 %* of the values in Table 4.3.
3. Optimised protein feeding in pig and dairy farming. *74 %* of the values for pigs and *85 %* of the values for dairy cattle. The remaining products are not consumed inside the EU.
4. Liquid manure pH reduction. *74 %* of the values for pigs and *85 %* of the values for dairy cattle, since the remaining products are not consumed inside the EU.
5. Tightening the regulation of manure application. *74 %* of the values for pigs and *85 %* of the values for dairy cattle, since the remaining products are not consumed inside the EU.
6. Cu reduction in dairy cattle and pig diets. 3 650 Mg (*79 %* of total reduction potential, since the remaining products are not consumed inside the EU).
7. Methane-reducing diets for dairy cattle. *85 %* of value in Table 4.12, since the remaining dairy products are not consumed within the EU.
8. Biogasification of 50 % of the manure from dairy cows and pigs. 337 Tg manure (*81 %* of the value in Table 4.12, since the remaining products are not consumed inside the EU).
9. Home delivery of groceries. *42 %* of values in Table 4.15.
10. New cold appliances only A++. *42 %* of the values in Table 4.17.
11. Power saving in farming, food industry, retail, and catering. In total nine TWh.
12. Household meal planning tools. Loss of meat and dairy products reduced by 2.5 % of total consumption.

Table 4.21: Improvement potentials of the suggested improvement options, when including rebound effects, in % of the total impact from consumption of meat and dairy products in EU-27. Negative values signify an improvement. The included rebound effects are described in the notes below the table.

	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Midpoint categories:												
Acidification	-0.23	-1.21	-8.00	-13.84	-0.19	-0.03		-1.25	-3.06	-0.34	-0.47	-1.49
Ecotoxicity, aquatic		-0.01	2.22	-0.08	0.05	-66.14		-0.12	-1.55	0.01	0.02	-1.31
Ecotoxicity, terrestrial		-0.18	0.73	-1.07	-0.71	-0.05		-1.81	-11.60	-0.28	-0.58	0.49
Eutrophication, aquatic	-2.71	-0.76	-4.74	2.80	-29.92	-0.02		-0.23	-2.48	0.02	0.04	-1.61
Eutrophication, terrestrial	-0.26	-1.36	-9.18	-15.75	-0.20	-0.03		-0.41	-2.05	-0.10	-0.13	-1.77
Global warming	-0.50	-0.72	0.87	-0.07	-2.42	-0.03	-2.54	-4.60	-5.50	-0.94	-1.31	-0.97
Human toxicity, carcinogens		-1.97	2.57	-0.24	-0.06	-0.05		-0.53	-9.42	0.07	0.14	-0.25
Human toxicity, non-carcinogens		-0.14	-0.26	-2.23	-0.34	-6.54		-0.70	-9.89	0.08	-0.08	0.23
Mineral extraction			1.12	-0.31	-0.27	-0.05		-0.48	-12.81	0.10	0.18	0.74
Nature occupation		-4.89	-2.26	-0.14	0.08	-0.03		-0.21	-2.10	0.02	0.03	-2.68
Non-renewable energy		-1.25	1.87	-0.28	-0.18	-0.05		-4.00	-12.53	-0.99	-1.37	0.66
Ozone layer depletion		-0.08	1.89	-0.31	-0.34	-0.05		-0.60	-10.83	0.09	0.16	0.20
Photochemical ozone, vegetation		-1.97	1.29	-0.22	-0.22	-0.04	-3.44	-3.30	-7.27	-0.37	-0.52	-0.77
Respiratory inorganics	-0.15	-1.44	-3.28	-6.39	-0.30	-0.03		-0.99	-4.47	-0.29	-0.40	-1.37
Respiratory organics		-1.88	1.00	-0.20	-0.13	-0.04	-3.89	-3.36	-7.07	-0.27	-0.38	-0.77
Endpoint (damage) categories:												
Impact on ecosystems	-0.17	-3.47	-1.50	-0.57	-0.84	-0.38	-0.77	-1.54	-3.17	-0.28	-0.38	-2.12
Impacts on human well-being	-0.16	-1.42	-3.12	-6.17	-0.35	-0.05	-0.07	-1.08	-4.56	-0.30	-0.42	-1.34
Impacts on resource productivity	-0.13	-1.51	-2.73	-5.67	-0.26	-0.04	-0.38	-1.21	-4.86	-0.28	-0.40	-1.28
All impacts aggregated	-0.17	-2.97	-1.88	-1.94	-0.71	-0.30	-0.61	-1.43	-3.54	-0.28	-0.39	-1.92

1. Catch crops on seven million hectares. Values from Table 4.1, but including 51 MEUR reduced final consumption, of which 12 % is meat and dairy products.
2. Improved growing practise and N intensification for cereal crops. Values from Table 4.3, but with 515 MEUR increased final consumption, of which 12 % meat and dairy products.
3. Optimised protein feeding in pig and dairy farming. Values from Table 4.7, but including 1380 MEUR reduced final consumption, of which 35 % is meat and dairy products.
4. Liquid manure pH reduction. Values from Table 4.8, but including 926 MEUR reduced final consumption, of which 35 % is meat and dairy products.
5. Tightening the regulation of manure application. Values from Table 4.10, but including 580 MEUR increased final consumption, of which 35 % is meat and dairy products.
6. Cu reduction in dairy cattle and pig diets. 4 630 Mg, including 217 MEUR reduced final consumption, of which 35 % is meat and dairy products.
7. Methane-reducing diets for dairy cattle. Values from Table 4.12.
8. Biogasification of 50 % of the manure from dairy cows and pigs. Values from Table 4.12, but with 1 410 MEUR reduced final consumption, of which 35 % meat and dairy products.
9. Home delivery of groceries. Values from Table 4.15, but with only 30 % of the reduction in car-driving, and including the resulting 40 000 MEUR reduced final consumption.
10. New cold appliances only A++. Values from Table 4.17, but including 343 MEUR increased final consumption as a rebound effect of the saved consumer spending.
11. Power saving in farming, food industry, retail, and catering. In total nine TWh, including 650 MEUR increased final consumption as an effect of the saved consumer spending.
12. Household meal planning tools. Loss of meat and dairy products *as well as fresh vegetables* reduced by 2.5 % of total consumption. Including 8 770 MEUR increased final consumption as a rebound effect of the saved consumer spending.

Table 4.22: Improvement potentials with rebound effects, and with specified synergies and dysergies, in % of the total impact from consumption of meat and dairy products in EU-27. Negative values signify an improvement. The included synergies and dysergies are described in the notes below the table.

	1	2	3	4	5	6	7	8	9	10	11	12
Midpoint categories:												
Acidification	-0.33	-1.21	-6.42	-12.26	-0.20	-0.03		-1.25	-6.11	-0.34	-0.47	-2.42
Ecotoxicity, aquatic		-0.01	2.22	-0.08	0.05	-66.14		-0.12	-3.09	0.01	0.02	-1.30
Ecotoxicity, terrestrial		-0.18	0.83	-0.97	-0.75	-0.05		-1.81	-23.20	-0.28	-0.58	0.95
Eutrophication, aquatic	-3.87	-0.76	-5.02	2.52	-28.51	-0.02		-0.23	-4.95	0.02	0.04	-2.33
Eutrophication, terrestrial	-0.37	-1.36	-7.37	-13.95	-0.22	-0.03		-0.41	-4.10	-0.10	-0.13	-2.81
Global warming	-0.71	-0.72	0.85	-0.10	-2.39	-0.03	-2.54	-4.60	-11.00	-0.94	-1.31	-1.80
Human toxicity, carcinogens		-1.97	2.57	-0.24	-0.07	-0.05		-0.53	-18.85	0.07	0.14	-0.51
Human toxicity, non-carcinogens		-0.14	-0.03	-1.99	-0.37	-6.54		-0.70	-19.79	0.08	-0.08	0.43
Mineral extraction			1.13	-0.30	-0.29	-0.05		-0.48	-25.61	0.10	0.18	1.49
Nature occupation		-4.89	-2.26	-0.14	0.08	-0.03		-0.21	-4.21	0.02	0.03	-5.19
Non-renewable energy		-1.25	1.88	-0.27	-0.20	-0.05		-4.00	-25.05	-0.99	-1.37	1.28
Ozone layer depletion		-0.08	1.90	-0.30	-0.36	-0.05		-0.60	-21.67	0.09	0.16	0.41
Photochemical ozone, vegetation		-1.97	1.30	-0.21	-0.23	-0.04	-3.44	-3.30	-14.53	-0.37	-0.52	-1.46
Respiratory inorganics	-0.21	-1.44	-2.56	-5.67	-0.32	-0.03		-0.99	-8.94	-0.29	-0.40	-2.49
Respiratory organics		-1.88	1.01	-0.19	-0.14	-0.04	-3.89	-3.36	-14.13	-0.27	-0.38	-1.46
Endpoint (damage) categories:												
Impact on ecosystems	-0.25	-3.47	-1.45	-0.53	-0.82	-0.38	-0.77	-1.54	-6.33	-0.28	-0.38	-4.00
Impacts on human well-being	-0.22	-1.42	-2.43	-5.47	-0.37	-0.05	-0.07	-1.08	-9.13	-0.30	-0.42	-2.44
Impacts on resource productivity	-0.19	-1.51	-2.09	-5.03	-0.28	-0.04	-0.38	-1.21	-9.73	-0.28	-0.40	-2.35
All impacts aggregated	-0.24	-2.97	-1.67	-1.73	-0.70	-0.30	-0.61	-1.43	-7.08	-0.28	-0.39	-3.60

1. Catch crops on 10 million hectares. 143 % (=10/7) of the values in Table 4.21.
2. Improved growing practise and N intensification for cereal crops. Same values as in Table 4.21. No quantified synergies or dysergies.
3. Optimised protein feeding in pig and dairy farming. Subtracted half of the mutual dysergy between this improvement options and option 4 (liquid manure pH reduction).
4. Liquid manure pH reduction. Subtracted half of the mutual dysergy between this improvement options and option 3 (optimised protein feeding).
5. Tightening the regulation of manure application. With effects of simultaneous introduction of improvement options 3 and 4, which influence the soil N pool, and thus influence the amount that can be affected by tightening of the regulation.
6. Cu reduction in dairy cattle and pig diets. Same values as in Table 4.21. No quantified synergies or dysergies.
7. Methane-reducing diets for dairy cattle. Same values as in Table 4.21. No quantified synergies or dysergies.
8. Biogasification of 50 % of the manure from dairy cows and pigs. Same values as in Table 4.21. No quantified synergies or dysergies.
9. Home delivery of groceries. Double of the values in Table 4.21, as a synergy effect with improvement option 12 (household meal planning tools).
10. New cold appliances only A++. Same values as in Table 4.21. No quantified synergies or dysergies.
11. Power saving in farming, food industry, retail, and catering. In total nine TWh. Same values as in Table 4.21. No quantified synergies or dysergies.
12. Household meal planning tools. Double the values in Table 4.21, as a synergy effect with improvement option 9 (home delivery of groceries), subtracting the reduction in avoided environmental impacts resulting from the implementation of the agricultural improvements (options 1 to 8 as reported in Table 4.20).

The total improvements, if all the identified environmental improvement potentials are taken together, are shown in Table 4.23.

Table 4.23: Sum of all improvement potentials from Table 4.20, Table 4.21, Table 4.22, in % of the total impact from consumption of meat and dairy products in EU-27. Negative values signify an improvement.

	Without rebound effects	With rebound effects	With rebound effects, synergies and dysergies
Midpoint categories:			
Acidification	-21.55	-30.09	-31.02
Ecotoxicity, aquatic	-51.66	-66.90	-68.44
Ecotoxicity, terrestrial	-6.01	-15.06	-26.03
Eutrophication, aquatic	-29.34	-39.61	-43.11
Eutrophication, terrestrial	-23.00	-31.24	-30.84
Global warming	-11.73	-18.74	-25.30
Human toxicity, carcinogens	-1.45	-9.75	-19.43
Human toxicity, non-carcinogens	-9.94	-19.89	-29.14
Mineral extraction	-2.68	-11.78	-23.83
Nature occupation	-5.87	-12.18	-16.79
Non-renewable energy	-7.18	-18.12	-30.02
Ozone layer depletion	-1.49	-9.86	-20.49
Photochemical ozone, vegetation	-9.80	-16.82	-24.77
Respiratory inorganics	-12.05	-19.12	-23.35
Respiratory organics	-10.17	-16.99	-24.74
Endpoint (damage) categories:			
Impact on ecosystems	-8.49	-15.18	-20.20
Impacts on human well-being	-11.94	-19.03	-23.39
Impacts on resource productivity	-11.69	-18.77	-23.50
All impacts aggregated	-9.34	-16.13	-21.01

A separate analysis has been made of the importance of accounting for market constraints, as described in Chapter 8.5, by comparing with the results using the unadjusted (attributional) database described in Chapters 8.2 to 8.4. Market constraints imply that the changes in demand resulting from implementing an improvement do not affect the average suppliers, but rather affects specific suppliers with specific technologies that may be significantly different from the average technology. For most of the improvement options, the results are not noticeably affected by the modelled market constraints. However, as reported in Table 4.24, significant differences are found for improvement options 3, 8, 10 and 12:

- The differences for improvement option 3 (optimised protein feeding) are due to the adjustment for land use when using soy as the unconstrained protein source.
- The differences for improvement option 8 and 10 (liquid manure biogasification and Cold appliances regulation) are mainly due to the larger NO_X, CO_2 and CO emissions when using the unconstrained electricity supply.
- The differences for improvement option 12 (household meal planning) are mainly caused by smaller environmental impact of the alternative supply route for dairy products when dairy farming is constrained by quotas, partly counterweighted by the larger environmental impact of beef products when the beef production is modelled as coming exclusively from meat cattle. As mentioned in Chapter 8.5, the quota constraint on milk production may be eliminated in the long term, which would imply an increase in the improvement potential for improvement option 12.

Table 4.24: **Significant differences (differences of 10 % or more) in improvement potentials as reported in Tables Table 4.7, Table 4.12, Table 4.17, and Table 4.18, relative to the results using the unadjusted (attributional) data that do not take into account market constraints.**

Impact category	Optimised protein feeding (Table 4.7)	Liquid manure biogasification (Table 4.12)	Cold appliances regulation (Table 4.17)	Household meal planning (Table 4.18)
Midpoint categories:				
Acidification		113	119	83
Ecotoxicity, aquatic			68	62
Ecotoxicity, terrestrial				89
Eutrophication, aquatic		122	131	80
Eutrophication, terrestrial		264	263	83
Global warming		125	198	89
Human toxicity, carcinogens				
Human toxicity, non-carcinogens				85
Mineral extraction				89
Nature occupation	-42			111
Non-renewable energy				
Ozone layer depletion				88
Photochemical ozone, vegetation		116	194	
Respiratory inorganics		153	163	90
Respiratory organics		110	171	
Endpoint (damage) categories:				
Impact on ecosystems	-40	125	196	
Impacts on human well-being		148	165	90
Impacts on resource productivity		138	166	91
All impacts		128	187	

4.9 Accounting for autonomous developments

The magnitudes of the suggested improvement options have to be considered in relation to the autonomous development trends.

In Nowicki et al. (2007), the major future trends and driving factors as regards European agriculture and rural regions until the year 2020 are analysed. According to this forecast, the agricultural production will remain concentrated in the central regions of EU, though the rate of growth in crop production will be slightly larger in the new member countries than in the old EU-15. Use of fertiliser is expected to increase substantially in the new member countries following the intensification of the crop production here. The structural development within farming will continue, leading to a reduced number of farms and an increase in average size.

For cereals, the technical productivity is expected to increase and land occupied for cereals is expected to decline by 5 %. For animal husbandry, a concentration on dairy production, poultry meat and pork meat production is anticipated, whereas beef production is expected to decline, i.e. beef consumption will to a larger part rely on imports from outside the EU. The milk yield per cow will increase, leading to a shrinking of the cattle population.

Besides these structural developments, a reduction in the use of fertilizer and in nitrogen leaching is forecast in EU-15 countries. This can be ascribed to steadily more tight environmental regulation of agricultural activities following implementation of the Nitrate Directive (EEC 1991). This directive induces a gradual change in farming practises to reduce risk of nitrate leaching. So, in consequence the overall trend for EU-27 is a reduction in nitrate leaching to the aquatic environment (10 % reduction by 2020 according to EuroCARE 2004). The Water Framework Directive (EC 2000), which will be rolled out in coming years, will accentuate this development. This directive aims at maintaining and improving the aquatic environment with particular focus on rivers, lakes and groundwater basins. Based on setting of targets and monitoring of the ecological state of water resources, actions should be taken – coordinated for each river basin – to reduce impact of agricultural activities, if the ecological state is not satisfactory. The implementation of this framework is anticipated to have a substantial impact on nitrate leaching in agriculture intensive areas. The detailed efforts/initiatives regarding nitrate leaching can be foreseen to be local/regional based, since in some areas quite extensive measures will be needed in order to obtain a satisfactory ecological state of the water resources.

The suggested improvement options for reducing nitrogen leaching (catch crops and tightening of the manure regulation) are affected by the Nitrate Directive and the Water Framework Directive. The extent of this impact depends on how large a share of the total emissions take place in areas that can be expected to be affected by the Water Framework Directive. It is estimated that the major share of the spring cereal production resides in such designated areas, and that a requirement for the use of catch crops in spring cereal production will be an inherent part of measures taken in relation to implementation of the Water Framework Directive. Consequently, the benefits in terms of reduced leaching can, for the major part, be expected to be achieved from this current policy instrument, although not as a generalised measure. Tightening of the regulation for overall N utilisation in manure is also likely to be part of the measures resulting from the water framework directive, but it is estimated that a substantial share of the animal manure is still applied outside the affected designated areas. Since the impact from nitrate emissions are not restricted to the areas affected by the Water Framework Directive, the main part of the improvement potential can only be reached through a more general regulation.

The suggested improvement option of intensifying the cereal production (Chapter 4.2.2) is expected to reduce land use for cereal production by 9 %. This can be compared with the autonomous development scenario (Nowicki et al. 2007), forecasting a reduction of land use for cereals by 5 %. When taking this into account, the improvement potential of an intensifying regulation can thus be expected to be less than half of what is stipulated in Chapter 4.8.

The suggested improvement options for reducing the ammonia emissions in animal husbandry (Chapter 4.3.1) are not foreseen in existing forecasts, such as Capsim (EuroCARE 2004). However, the same elements are included in the most recent BAT recommendations (BREF 2003), and with the forecasted shift towards larger and more specialised production units that fall under the IPPC Directive it is likely that such measures, requiring farm-balances and inspection, will increasingly be used in local/regional regulation, at least for pig and poultry farms that fall under the IPPC Directive, but possibly also for larger dairy and cattle farms.

There are no autonomous developments that are expected to reduce Cu content in animal diets.

The suggested improvement options for reducing methane emissions from animal husbandry are not foreseen in existing forecasts, but the methane emission related to beef production is expected to decrease due to a reduced beef production within EU-27 (Nowicki et al. 2007). Since the consumption is expected to be fairly stable, this implies a larger import of beef. Therefore, the methane emission related to beef consumption will increasingly be dependent on the production systems in countries outside EU-27. The general intensification of dairy and pig farming facilitates the establishment of biogas plants due to size economy.

The calculation in this study of the changes in environmental impacts from an intensification of milk production, presented in Table 4.13 and Table 4.14, has its parallel in the forecast of Nowicki et al. (2007), where this development is the expected result of an agricultural 'liberalisation', of which a part will be abandonment of the milk quota system. As pointed out, the negative effects of intensification of dairy farming in general outweigh the benefits, implying that it would be possible to regard prevention of intensification as an improvement option.

It has not been possible to find any reliable data on autonomous developments in home deliveries of groceries or Internet shopping of groceries in Europe. In many of the new Member States it is probably almost non-existent today. In Denmark, the supermarket chain ISO made an attempt, which was cancelled after a few years. In the United Kingdom, Internet shopping of groceries seems to be a success. According to Online Business Toolkit (OBT 2006), 20 % of adults in the United Kingdom have bought food from the Internet within the last three months. The supermarket chain Tesco has 66 % of the country's online shopping market. In 2005-2006, Tesco's Internet sales of groceries grew by 32 %, now accounting for 3 % of Tesco's total sales in the United Kingdom. According to Sir Terry Leahy, chief executive of Tesco, Tesco is by far the largest in the world in the Internet groceries sector. That means that far less than 3 % of the total groceries sales in the United Kingdom are by Internet today. The conclusion here is that even with a growth of 32 %, there is a long way before Internet shopping is a common way of shopping for groceries. For Europe as such, 20 % by year 2020 is a probable target, if the development in the United Kingdom is seen as a model.

The suggested improvement option for cold appliances regulation will be affected by the general increase in the market for cold appliances, which is due to an increasing number of households (0.8 % annually, according to Mantzos & Capros 2006), an increase in number of appliances per household, and an increase in size of appliances (and in added functionalities like ice-machines, wine-coolers etc.). CECED (European Committee of Domestic Equipment Manufacturers) writes on their homepage www.ceced.be (accessed 2007-03-15): '*Our concern is that between 1995 and 2005 energy consumption dropped by only 20 % whilst energy efficiency has gone up by 45 %.*' This increases the need for energy efficient appliances and hence emphasizes the importance of the suggested improvement options.

The suggested improvement option of power saving in industry (targeted measures to increase electricity efficiency in farming, food industry, retail, and catering), will be reduced in proportion to the autonomous improvement in energy intensity in industry (1.2 % annually according to Mantzos & Capros 2006).

It has not been possible to find any reliable data on autonomous uptake of household meal planning tools.

5 SOCIOECONOMIC IMPACTS OF IMPROVEMENT OPTIONS

The purpose of the socioeconomic impact assessment is to allow an overall judgement on the desirability of implementing an environmental improvement option, including the trade-offs in changes of environmental impacts and the other socioeconomic impacts (economic costs, dietary health, etc.). The first step of the assessment is a qualitative screening for the relevant impacts. Then, impact indicator values are quantified for some of the most relevant impacts. Finally, the different types of environmental and socioeconomic costs and benefits, expressed in euro, are compared for each of the improvement options. The details of the impact assessment methodology are presented in Chapter 10.

5.1 Qualitative screening of improvement options

Initially, a qualitative screening was performed, identifying which of the socioeconomic indicators are likely to be affected by each improvement option, scored on a none-low-medium-high relevance scale. The resulting matrix of improvement options and socioeconomic indicators is shown in Table 5.1.

Table 5.1: Relevance of socioeconomic indicators per improvement option. O = no relevance; L = low relevance (insignificant); M = medium relevance; H = high relevance.

	Dietary health (a)	Food safety (a)	Supply security (a)	Well-being of animals (b)	Landscape main-tenance (c)	Employ-ment (d)	House-hold time usage (a)	Income re-distribu-tion (d)
1. Catch crops	O	O	O	O	M	O	O	O
2. Cereal intensification	O	O	L	O	L	L	O	M
3. Optimised protein feeding	O	O	O	L	M	O	O	L
4. Liquid manure pH reduction	O	O	O	L	M	O	O	L
5. Tightening of manure regulation	O	O	O	L	M	O	O	L
6. Copper reduction in animal diets	O	O	O	L	O	O	O	L
7. Methane-reducing animal diets	O	O	O	L	M	O	O	O
8. Liquid manure gasification	O	O	O	O	M	L	O	L
9. Home delivery of groceries	L	L	M	O	L	L	H	M
10. Cold appliances regulation	O	O	O	O	O	O	O	M
11. Power saving in industry	O	O	O	O	O	O	O	L
12. Household meal planning	H	L	M	O	L	O	M	M

a. The scores for dietary health, food safety, supply security, and household time usage are (except for cereal intensification) all related to increased (easy) availability of (healthy, safe) food, which is generally increased by the improvement options 'home delivery of groceries' and 'household meal planning'. For cereal intensification, availability / supply, security may in principle be affected, but this is assessed as insignificant.

b. Well-being of animals may be affected by diet changes, and by reduction in odour (ammonia).

c. The M scores relate to aesthetic (visual or odour) impacts from agricultural changes. The L scores relate to the reduction in the quantity of cultural heritage if arable land is abandoned as a result of intensification or reduction in food demand.

d. Improvement options that specifically affect geographical areas where employment options and/or income are low, or involve technologies or processes that are specifically affecting or addressing low-income and/or low-skilled persons.

5.2 Quantified socioeconomic indicator results

For combinations of indicators and improvement options assigned high or medium relevance, it has been attempted to quantify the indicator values for the relevant parts of the product systems, using available statistical and technical data sources. Quantification has not always been possible – see also the explanations given in Chapter 10. The following sections describe the impact assessment calculations made by each improvement option in turn (including data used, assumptions and results).

5.2.1 Catch crops

The introduction of catch crops on seven million hectares leads to a saving of artificial fertiliser of 25kg N/ha/year, as described in Chapter 4.2.1. At a fertiliser price of 0.51 EUR/kg N, the total annual saving of 175 Gg gives a reduction in production costs of 89 MEUR. The costs related to implementation of catch crops, based on Hansen (2003), cover costs for grass seeds (12 EUR/ha) and sowing (eight EUR/ha for work and machinery, ranging from 0 to 16 EUR/ha depending on whether seeds are sown together with the cereal or in a separate procedure), amounting to 20 EUR/ha/year or EUR 140 million in total. The net annual cost is thus EUR 51 million, of which 35 % or EUR 18 million is linked to the consumption of meat and dairy products in EU-27.

No attempts have been made to quantify the visual impact from 'greening' the winter landscape by catch crops, since the implementation of catch crops already shows a net benefit without accounting for the benefits from landscape maintenance.

5.2.2 Cereal intensification

Cereal intensification as described in Chapter 4.2.2 leads to a 9 % reduction in fuel costs or EUR 520 million annually for EU-27, of which 35 % or EUR 180 million per year is linked to the consumption of meat and dairy products in EU-27. The cereal intensification implies an increase in yield per ha, but with the same input of fertilisers and pesticides per kg of cereal produced. The reduction in area use obviously also means a reduction in land rent, but this is not included in the calculation of changes in production costs, since land rent is a transfer cost, i.e. it does not increase overall societal costs, but merely transfers some capital from one group to another; see also below.

The intensification of cereal growing increases supply security by increasing European production capacity and thus reducing the need for imports. However, the current supply security for cereals is already good, and the change is therefore not expected to influence the supply security significantly. No attempt at quantification has been made.

Cereal intensification leads to a reduction in the quantity of cultural heritage and in local low-skill jobs, when arable land is abandoned. The importance of this has been assessed as insignificant, and no attempt at quantification has been made.

Cereal intensification may increase income of farmers at the expense of owners of agricultural land. As European farmers often also own their own land, this income re-distribution will be limited, and no attempt at quantification has been made.

5.2.3 Optimised protein feeding

The cost of the suggested changes in feeding can be calculated from the data in Table 4.5 and Table 4.6. Using the prices of 0.109 EUR/kg cereals, 0.114 EUR/kg soy, 0.509 EUR/kg N in fertiliser, 2.1 EUR/kg amino acids and the production volumes 153 Tg milk and 299.2 million pigs, a value of EUR 1 380 million annually for EU-27 is obtained. Using 74 % of the values for pigs and 85 % of the values for dairy cattle, corresponding to the proportion of European production consumed in Europe, a value of EUR 1 050 million annually is obtained.

Animal welfare may be affected when diets are changed. The reduction in ammonia concentrations in stables may improve animal welfare. Overall, this has been assessed as insignificant.

Reduction in odour from ammonia emissions increases landscape accessibility and its recreational value. Since ammonia reduction already shows a net benefit without accounting for the benefits from landscape maintenance, no attempt at quantification has been made.

As the change in feeding involves a cost, this may imply that costs are shifted among income groups. This has been assessed as insignificant.

5.2.4 Liquid manure pH reduction

The pH reduction of the liquid manure through addition of sulfuric acid is most promising in fattening pig units where straw addition is very limited. Costs are estimated to EUR 1.3 per carcass produced (Pedersen 2007, personal communication) or EUR 1.24 per Mg manure.

As discussed in Chapter 4.3.1, a reduction of pH can also be achieved through changed feeding. By replacing 6 g of calcium in the form CaCO3 by calcium-benzoate in the feed mixture (per kg), ammonia was reduced by 60 % according to Aarnink et al. (2007). Estimated cost of this measure is 1 EUR/100 kg feed, i.e. more costly than treating the liquid manure directly. For dairy production there are less – probably no – practical possibilities to reduce the pH through dietary means.

For the total annual amount of manure in EU-27 from dairy farms (545 Tg) and pig farms (285 Tg), the cost of pH reduction through addition of sulfuric acid will be EUR 1 036 million. The proportion for consumption in Europe is 85 % and 74 % for dairy and pigs, respectively, or EUR 836 million per year in total. The pH reduction of the liquid manure increases the fertiliser value of the liquid manure slightly and thus reduces the need for artificial fertiliser, as described in Chapter 4.3.1. The saved artificial fertiliser amounts to EUR 85 million per year for the meat and dairy products consumed in Europe.

The net direct costs used in the impact assessment are EUR 750 million per year.

Impacts on animal welfare, landscape, and income distribution, are similar to those described for 'optimised protein feeding' (Chapter 5.2.3).

5.2.5 Tightening of manure regulation

The savings in artificial manure as shown in Table 4.9 amount to 1 140 Gg N per year, of which 83 % (950 Gg) relate to the meat and dairy products consumed in EU-27. With a price of 0.51 EUR/kg, this amounts to an annual cost saving of EUR 480 million .

Impacts on animal welfare, landscape, and income distributi are similar to those described for ‘optimised protein feeding’ (Chapter 5.2.3).

5.2.6 Copper reduction in animal diets

The additional costs for alternatives to copper in pig feed is estimated from average costs of probiotics in feed mixtures according to the list prices of DLG, the Danish cooperative feedstuff company, to 0.5 EUR/100 kg feed for piglets and 0.25 EUR/100 kg feed for fattening pigs. Of the total annual feed consumption of 91.2 Tg in the pig farming model for EU-27, 14 % is for piglets, 67 % for fattening pigs, and the rest for the sows. Thus, the average added price is 0.24 EUR/100 kg pig feed, or EUR 217 million annually for EU-27, of which 74 % or EUR 160 million are for the pork consumed in Europe.

Animal welfare may be affected when diets are changed. This has been assessed as insignificant.

As the change in feeding involves a cost, this may imply that costs are shifted among income groups. This has been assessed as insignificant.

5.2.7 Methane-reducing animal diets

In principle, introducing an additional restriction will reduce the flexibility in feed optimisation slightly. This is not expected to affect costs significantly.

Animal welfare may be affected when diets are changed. This has been assessed as insignificant.

The diet changes may slightly reduce odour. Since methane-reducing diets already show a net benefit without accounting for the benefits from odour reduction, no attempt at quantification has been made.

5.2.8 Liquid manure biogasification

The production via biogas of 52 kWh electricity per Mg manure, at a price of 0.054 EUR/kWh will provide an income from sale of electricity of 2.8 EUR/Mg manure. With capital costs of 1.8 EUR/Mg manure and operation costs of 4.4 EUR/Mg manure (based on Walla & Schneeberger 2003), the net costs are 3.4 EUR/Mg manure. With 415 Tg manure per year from 50 % of all dairy and pig farms in EU-27, this implies an annual cost of EUR 1 410 million, of which 81 % or EUR 1 150 million is linked to the consumption of meat and dairy products in EU-27.

Biogasification may reduce odour and thereby increase landscape accessibility and its recreational value. Since biogasification already shows a net benefit without accounting for the benefits from odour reduction, no attempt at quantification has been made.

Biogas production may involve low-skilled jobs in rural areas where employment and/or income are low. This has been assessed as insignificant.

5.2.9 Home delivery of groceries

The direct production costs for a delivery service is set at 3.99 EUR/delivery, based on a calculated costs of vehicles and fuel of EUR 0.07, wages of EUR 3.17 calculated from an hourly wage of EUR 9.5 (50 % of the average salary in EU-27 in year 2000) and 20 minutes per delivery, including packing of the goods, leaving EUR 0.75 for taxes and operating surplus. The corresponding saved costs for private transport amount to EUR 1.03 EUR per delivery. With an expected 23 deliveries per capita per year, this amounts to an additional cost of EUR 33 000 million per year of which 42 % or EUR14 000 million is linked to the consumption of meat and dairy products in EU-27.

The increased accessibility to food supply may have a beneficial effect on dietary health and food safety, especially when combined with meal planning tools. The effect has been assessed as insignificant.

A delivery service may increase supply security for the households, by reducing household-out-of-stock situations. It is estimated, based on data from Chandon & Wansink (2006), that 5 % of overall demand for meat and dairy products is unmet due to household-out-of-stock situations. However, as explained below (Chapter 10.6), the security of supply benefits have not been quantified in this study.

By possibly reducing food loss, home delivery of groceries may lead to a reduction in food demand and consequently abandonment of arable land and a reduction in the quantity of cultural heritage. This has been assessed as insignificant.

Home delivery of groceries may provide low-skilled jobs in areas where employment is low. This has been assessed as insignificant.

Household time usage for shopping for meat and dairy products is estimated to be 27 hours per capita per year (see Chapter 10.10 on the details). In Chapter 4.4, it was estimated that 25 % of the shopping trips could be replaced by a delivery service. The time spent for ordering groceries could be outweighed by the replaced trips being more time-consuming than an average shopping trip, so that also the overall time-saving could be estimated to 25 % of the time spent for food shopping. This would then liberate seven hours per capita or 3.3E+9 hours annually in EU-27. At EUR 9.5 per hour, this amounts to 65 EUR/capita/year or EUR 31 billion for EU-27. For the overall assessment of this improvement option, the amount of time saved and its monetary value are important factors and the dominating cause of uncertainty (estimated 95 % confidence interval: EUR 19 billion to EUR 43 billion).

As half the average wage has been applied to the home delivery service, the home delivery service will imply a significant income redistribution from high to low-income groups, provided that the participating households are average households with an average salary level. In practice, the participating households may even have above-average incomes implying larger redistribution. With the above wage of EUR 3.17 per delivery and 23 deliveries per capita per year, the transfer amounts to 73 EUR/capita/year or EUR 35 billion from the average-income group to the 50 %-income group.

With linear proportional weights, this would give a cost of EUR 35 billion distribution-weighted and a benefit of EUR 70 billion distribution-weighted, i.e. a net benefit of EUR 35 billion annually, of which 42 % or EUR 29 billion is linked to the consumption of meat and dairy products in EU-27. Proportional weighting is likely to underestimate the importance of income differences, and it is not recommended to add income re-distribution to the results for the other environmental and socioeconomic indicators, but rather to keep this information separate (see also Chapter 10.11).

5.2.10 Cold appliances regulation

In Chapter 4.5, the accumulated electricity savings from an 'A+/A++ appliance scheme', where consumers pay the price difference over the electricity bill year by year corresponding to the cost of the saved electricity, were calculated to 340 kWh per 100 litres, which is 11.3 kWh per 100 litres per year. This saving amounts to 6.7 % of the current annual electricity consumption of 168 kWh per 100 litres or a total of EUR 595 million annually for EU-27. Using the proportions from Mebane & Piccinno (2006) between electricity savings and equipment production costs, the additional annual equipment cost to achieve this electricity saving can be calculated to EUR 252 million, giving an undiscounted net annual saving from the 'A+/A++ appliance scheme' of EUR 343 million for EU-27, of which 42 % or EUR 144 million is ascribed to meat and dairy products.

Shifting production from the electricity industry to the manufacturing industry has little effect on income redistribution, since the wages are not significantly different between these two sectors.

5.2.11 Power saving in industry

The saving of nine TWh estimated in the last paragraph of Chapter 4.5, amounts to EUR 650 million annually, using the EU-27 average electricity price for industry of 72 EUR/MWh.

The change in electricity production involves a cost reduction, which may imply that costs are shifted among income groups. This has been assessed as insignificant.

5.2.12 Household meal planning

Saving 2.5 % of the purchased food amounts and associated waste treatment, as reported in Table 2.1, amounts to a cost saving of EUR 6 800 million annually, of which 28 % is from dairy products, 71 % is from meat and 1 % is from saved waste treatment.

Meal planning may have a beneficial effect on dietary health and food safety. A conservative assumption that the 9 % change in purchase behaviour in the trial of Huang et al. (2006) – see Chapter 10.4 – will be reflected in a 9 % decrease in dietary related morbidity and mortality, allows to combine the assumed 25 % uptake of dietary advice through Internet shopping with the estimate of Mathers et al. (2004) of the total damage of 28.6 E+6 QALY attributable to dietary health in Europe, to calculate a total annual potential health effect of meal planning tools to 0.64 E+6 QALY.

There is a lack of data to assess the consumer willingness to pay for the missing products during situations of undersupply and the benefits of increased security of supply have therefore not been quantified in this study (see also Chapter 10.6). It may be worthwhile to include this issue in further studies on household planning and shopping behaviour, since the implied welfare gains may indeed be significant.

The same recommendation is appropriate with respect to changes in household time usage with the introduction of household meal planning tools. It is currently not known whether these are time-saving or time-consuming and how they interrelate to other activities.

By reducing food loss, meal planning tools may lead to a reduction in food demand and consequently abandonment of arable land and a reduction in the quantity of cultural heritage. This has been assessed as insignificant.

The savings in food expenditures implies an income redistribution from agricultural and food producing industries to the average industries. This income redistribution has not been quantified.

5.3 Summary of socioeconomic impact assessment

The quantified socioeconomic impacts for all suggested improvement options are summarised in Table 5.2, along with the monetarised results from Table 4.20 and Table 4.21.

The socioeconomic impacts in Table 5.2 are those reported in Chapter 5.2, but adjusted for the same synergies and dysergies as reported for the environmental impacts in Table 4.22. Furthermore, the impacts have been adjusted for the same rebound effects as reported for the environmental impacts in Table 4.21, with the obvious exception that the price rebound does not apply to the direct costs.

Not all of the socioeconomic impacts discussed in this chapter have been quantified and it has therefore not been possible to include them in Table 5.2. In the case of 'household meal planning', a rough estimate pointed to potentially very large additional positive health effects, which would make this improvement option even more attractive. The estimate has been regarded as too uncertain to warrant inclusion in Table 5.2. Likewise, the quantified estimate of income redistribution for 'home delivery of groceries' has not been included.

The confidence intervals are calculated from the previously described uncertainty estimates on each of the components, mainly having a lognormal distribution, and under the assumption that the co-variance between socioeconomic and environmental impacts is negligible. This implies that the overall standard deviations have been calculated as the square root of the sum of the squared components. The confidence intervals may therefore be slightly overestimated.

Table 5.2: **Overall socioeconomic impact assessment with monetarised environmental impacts, including rebound effects and synergies/dysergies. The socioeconomic impacts are calculated after inclusion of rebound effects and synergies/dysergies. All values in MEUR per year. Negative values signify an improvement (= cost reduction). Sums are rounded**.

	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Quantified socioeconomic impacts:												
Direct costs	73	-515	1 400	940	-608	217	0	1 410	80 300	-343	-648	-15 400
Injuries	-4	15	-38	-32	19	-6	0	-49	-9 370	14	25	390
Household work (time usage)									-149 000			
Sum of quantified socioeconomic impacts	*70*	*-501*	*1 360*	*909*	*-590*	*211*	*0*	*1 360*	*-78 000*	*-329*	*-623*	*-15 000*
Monetarised environmental impacts:												
Impacts without rebound effects (from Table 8.21)	-140	-2 600	-3 200	-3 500	-1 630	-511	-1 280	-2 430	-912	-318	-1 080	-5 350
Rebound effects (cf. Table 8.22 – difference to Table 8.21)	-273	-4 690	-1 430	-1 260	-115	-220	-225	-1 090	-7 760	-371	127	640
Impacts of synergies/dysergies (cf. Table 8.23 – difference to Table 8.22)	-177	0	500	500	25	0	0	0	-8 680	0	0	-4 100
Sum of socioeconomic and environmental impacts	*-521*	*-7 790*	*-2 770*	*-3 350*	*-2 310*	*-520*	*-1 510*	*-2 160*	*-95 400*	*-1 020*	*-1 580*	*-23 900*
Low end of 95 % confidence interval	-33	-2700	780	690	-850	230	-520	310	-57 000	-480	-760	-14 000
High end of 95 % confidence interval	-1 100	-23 000	-13 000	-28 000	-6 700	-7 600	-4 400	-6 600	-140 000	-2 200	-3 400	-42 000

1. Catch crops on 10 million hectares.
2. Improved growing practise and N intensification for cereal crops.
3. Optimised protein feeding in pig and dairy farming.
4. Liquid manure pH reduction.
5. Tightening the regulation of manure application.
6. Cu reduction in dairy cattle and pig diets.
7. Methane-reducing diets for dairy cattle.
8. Biogasification of 50 % of the manure from dairy cows and pigs.
9. Home delivery of groceries.
10. New cold appliances only A++. Contrary to the other improvement options, this is a temporarily limited measure calculated over 30 years, and the annual impacts are therefore not directly comparable to that of other more 'permanent' improvement options.
11. Power saving in farming, food industry, retail, and catering.
12. Household meal planning tools.

It appears from Table 5.2 that all improvement options have an overall benefit (= negative cost) when adding the socioeconomic and environmental results. The results for the agricultural improvement options (options 1 to 8) and the power saving options (options 10 and 11) are largely dominated by the benefits from reduced environmental impacts. Although option 9 (home delivery of groceries) and option 12 (meal planning tools) also have large benefits from reduced environmental impacts, the main benefits comes from the socioeconomic impacts of reduced household work and reduced consumer expenditure, respectively.

For the four agricultural improvement options with the largest direct economic costs:

3. Optimised protein feeding
4. Liquid manure pH reduction
6. Copper reduction in dairy cattle and pig diets
8. Liquid manure biogasification.

the socioeconomic advantage is not certain at the level of 95 % confidence, due to the large uncertainties in the monetarisation of the environmental impacts, even though the reduction in environmental impact is very certain.

For home delivery of groceries, it should also be noted that a dominating share of the environmental impact improvement is due to the price rebound effect caused by unpaid household work being converted to a paid service. The same environmental impact reduction would be achieved by any other conversion of household work to paid services, and thus points to the general significance of binding the purchase power of the households in provision of services with a low environmental impact.

5.4 Effect of temporal discounting

The direct production costs (see Chapter 10.1), as well as most of the socioeconomic impacts, occur immediately from the time of implementation, while especially the most important environmental impact categories (such as nature occupation and global warming) have long-term impacts. The direct production costs include capital investments (unless explicitly excluded), but do not include costs of policy implementation and enforcement. For two of the improvement options (liquid manure gasification and cold appliances regulation), significant capital investments (with their own environmental impacts) are required initially, while cost savings and reductions in environmental impacts occur later.

A positive temporal discount rate implies that economic costs and benefits as well as the importance (cost) of environmental impacts occurring in future is reduced for each unit of time that the cost or benefit is removed from the present, and would therefore mainly affect the weight of environmental impacts relative to the socioeconomic impacts, i.e. it would emphasise the socioeconomic benefits of the household improvement options relative to the mainly environmental benefits of the agricultural improvement options and power savings. Temporal discounting would thus emphasise the general conclusion that the household improvements are the most important.

Furthermore, temporal discounting would reduce the importance of impact categories with long-term impacts (such as nature occupation and global warming) more than impact categories with more immediate impacts (such as toxicity and acidification).

Here, quantifying the effect of temporal discounting meets another practical problem, namely that the life cycle impact assessment methods do not explicitly specify the temporal aspects of the different impacts. A qualitative assessment, confirmed by the quantitative example in Table 5.4, shows that the reduced importance of long-term impacts would particularly affect the internal ranking of the agricultural improvement options, reducing the relative importance of the improvement options focused on reducing area use and methane emissions, and increase the relative importance of improvement options focused on, for example, reducing ammonia emissions.

The described effects of temporal discounting would increase with the size and uniformity of the discount rate.

As an example of the effects of temporal discounting, Table 5.3 presents the net present value for two of the improvement options, calculated at a 3 % constant annual discount rate, under the very simplified assumption that the reductions in impacts on global warming and nature occupation occur over a period of 100 years, while the other reductions in impacts occur over a period of 10 years, in both cases with an equal reduction in impact per year. Investments in capital equipment are placed at year one and therefore not discounted, while changes in operating costs and related emissions are distributed equally over the lifetime of the capital equipment and discounted accordingly.

Table 5.3: Examples of the effect of discounting at 3 % constant annual discount rate. All values in MEUR.

	Manure biogasification (improvement option 8)	New cold appliances only A++ (improvement option 10)
A. Average annual capital investment	750	250
B. Annual operating costs, undiscounted	660	-600
C. Annual operating costs, net present value	490	-390
D. Monetarised annual reduction in impact of global warming and nature occupation, undiscounted	2 810	500
E. Monetarised annual reduction in impact of global warming and nature occupation, net present value	690	110
F. Monetarised annual reduction in other impacts, undiscounted	760	170
G. Monetarised annual reduction in other impacts, net present value	510	100
Annual net benefit, undiscounted (-A-B+D+F)	2 160	1 020
Annual net benefit, net present value (-A-C+E+G)	-40	350

'Manure biogasification' is the improvement option for which temporal discounting has the largest possibility of reversing the result (i.e. resulting in an overall negative net present value, implying that the change is not an improvement), which happens if the constant annual discount rate above 2.8 % is applied. This discount rate (at which the improvement option will have zero net present value) is also known as the 'internal rate of return'. For 'new cold appliances only A++' the internal rate of return is 9.1 % under the conditions applied for the calculations in Table 5.3.

Since the above example of 'manure biogasification' is for the improvement option for which temporal discounting has the largest influence, it may be concluded that temporal discounting would not affect the overall conclusions of this study unless discounting future impacts above the equivalent of a 2.8 % constant annual discount rate.

Table 5.4 shows the net present value at 3 % constant annual discount rate and the internal rate of return for the remaining improvement options. The internal rate of return is undefined for options where both direct costs and environmental impacts show a benefit. It should be noted that the internal rate of return cannot be used to prioritise between improvement options. For this purpose, the net present value is the most appropriate.

Table 5.4: Undiscounted annual net benefits, net present value under 3 % constant annual discount rate, and the internal rate of return for the remaining improvement options not described in Table 5.3.

Improvement option	Annual net benefits, undiscounted	Annual net benefits, net present value	Internal rate of return
	MEUR	MEUR	%
1. Catch crops	521	224	47.0
2. Cereal intensification	7 790	4 760	undefined
3. Optimised protein feeding	2 770	970	12.1
4. Liquid manure pH reduction	3 350	2 630	76.6
5. Tightening of manure regulation	2 310	1 360	undefined
6. Copper reduction in animal diets	520	420	48.8
7. Methane-reducing animal diets	1 510	520	undefined
9. Home delivery of groceries	95 400	83 500	undefined
11. Power saving in industry	1 580	1 050	undefined
12. Household meal planning	23 900	18 300	undefined

6 FEASIBILITY AND POLICY ANALYSIS

6.1 Alignment with existing policies

6.1.1 Agricultural improvement options

The common agricultural policy includes agri-environmental measures that support specific farming practises protecting the environment and maintaining the countryside as a cultural heritage.

Two specific measures that are expected to influence farming practise and environmental issues further in the near future are the rural development plan and the single payment scheme.

The rural development Regulation (EC 2005b) aims at improving the competitiveness of the agricultural sector, improving the environment and the countryside, improving the quality of life in rural areas, and encouraging diversification. Member States are among other things encouraged to support and promote animal friendly farming practises, and to preserve the farmed landscape, ranging from wetlands to dry meadows and mountain pastures. This is also supposed to be part of promoting high value livestock products that can be marketed as regional products. It is common for such systems that grazing will be an inherent part of the system, and that it may sustain rather small enterprises. Although the suggested improvement options do not counteract these aims of farming, a number of the suggested improvement options are difficult to implement in such systems.

It should be noted that the land area maintained under extensive pasture is regarded as independent of the analysed meat and dairy systems, and therefore not included in this study.

The Regulation establishing common rules for direct support schemes under the Common Agricultural Policy (EC 2003b) placed increased focus on cross-compliance i.e. farmers must respect environmental standards (as well as standards on public, animal and plant health) in order to obtain direct payments. An important part of the cross-compliance scheme concerns keeping land in 'good agricultural and environmental condition', including the protection of permanent pasture:

- Member States shall ensure that land that was under permanent pasture in 2003 remains under permanent pasture.
- All agricultural land, especially land that is no longer used for production purposes, has to be maintained in good agricultural and environmental condition. Member States define minimum requirements for 'good agricultural and environmental condition' based on the general framework set up in Annex IV to the common Regulation. Typically, this implies that land must be kept covered with vegetation and be managed (through cutting or grazing) so that wood and bushes do not appear.

It should be noted that no improvement options have been suggested in relation to beef production, mainly because such improvements would require an intensification as in dairy farming, where the cattle is kept in stables for most of the time, being fed a more concentrate rich diet. Although such a beef production system is likely to have significant less emissions to the environment, it appears unrealistic to suggest such a measure, mainly because extensive grazing is financially supported in many countries and less favoured areas, as a means of preserving the agricultural heritage (the manufactured rural landscape and culture), which is also what is expressed in the rural development regulation and the above-mentioned measures in the single payment scheme. Fundamentally, there is a trade-off between having more nature or having more agricultural heritage, and the current policies have largely chosen in favour of the latter.

Other important policy instruments affecting the agriculture-environment interaction are the Nitrate Directive (EEC 1991) and the Water Framework Directive (EC 2000).

The Nitrate Directive aims at protecting waters from pollution caused by nitrates from agricultural sources. All improvement options suggested here are in line with this directive.

With one exception, all improvement options suggested here are in line with the aim of the Water Framework Directive (reduced emissions of nutrients and other harmful substances to the environment). The exception is the suggested intensification of the cereal production. While the intensification decreases emissions at the global level for a given amount of cereal produced, emissions may slightly increase at the local scale in some situations, and thus be counteracting the intentions of the Water Framework Directive at this local scale. However, the effect of the Water Framework Directive will rather be to direct the intensification to areas that are not sensitive to nitrogen leaching, implying that in practice there should be no conflict between the two incentives.

The suggested improvement option of liquid manure gasification is in line with the strategy paper for reducing methane emissions (EC 1996) and the Directive on the promotion of electricity produced from renewable energy sources (EC 2001a).

6.1.2 Other improvement options

The EU policies regarding cold appliances are also addressed in section 8.5. Cold domestic appliances are among the 14 prioritised product groups in the EU 'Priority Action 1' (EC 2006c). Accordingly, the suggested improvement option regarding cold appliances is in line with the EU policies. As the minimum energy performance standards for cold appliances are not yet ready (expected to be adopted by the European Commission by mid 2008) it is not clear whether the suggestion: 'all new appliances sold in Europe should be either A+ or A++' is more far-reaching than the upcoming minimum energy requirements by the EU Commission.

The improvement option of power savings in the industry is in accordance with the Action Plan for Energy Efficiency (EC2006c), which states that 'Even though many energy efficiency measures are fully cost effective with very short payback periods, many such measures are not undertaken due to financial barriers. This is not least the case in small and medium-sized enterprises.' The action plan includes a commitment to identify and seek to remove remaining legal barriers in national legislation to the use of companies supplying efficiency solutions (the so-called 'Energy Service Companies' or 'ESCOs') and to financing arrangements, as well as considering the use of tax credits as incentives. The action plan also includes the development of education and training plans and programmes for energy managers in industry, and an Environment Programme for SMEs (SME-Environment), including an energy efficiency toolkit.

According to the authors' knowledge, there are no EU policies with direct relevance for the improvement option of home delivery of groceries and household meal planning tools.

6.2 Feasibility analysis and assessment of policy instruments

6.2.1 Nitrogen management

Four of the suggested improvement options relate to nitrogen management; two with a focus on nitrate leaching (catch crops for spring cereals and tightening of manure regulation, involving minimum N utilisation rates for animal manure) and two with a focus on ammonia evaporation (optimised protein feeding and liquid manure pH reduction).

For these improvement options, policy instruments could build on a number of existing measures and ongoing developments:

- the existing instruments for detailed monitoring, in the form of the monitoring of land use for the agricultural support schemes, which could cover also the use of catch crops, and in the form of on-farm inspection which is widely applied in relation to cross-compliance measures (EC 2003b) and can be expanded to cover controls for the suggested improvement options;
- the existence of the cross-compliance instrument, which provides a strong incentive for farmers to respect existing legislation, and makes enforcement of standards more effective (Nitsch & Osterburg 2007);
- the autonomous intensification and specialisation of dairy and pig farming resulting in bigger enterprises, makes individual requirements and approval of the production practise more practicable.

In a number of countries a mandatory reporting of fertiliser application and production volume of livestock exists on farm level (such as Denmark, Germany and the United Kingdom). In countries/regions where this is not planned to be implemented, a possible alternative is to tax nitrogen in agriculture. More expensive N will stimulate a more efficient use, thus stimulate to improve the utilisation of manure N (and also catch crops). Although not implemented, this scenario was considered in Denmark as an alternative to direct management regulation (Jacobsen et al. 2004). While tax on N in fertilizer from an administrative and tax authority point of view was preferable, a tax on N balance (on farm) was found to be most efficient in terms of reducing N leaching. It should also be noted that economic instruments (subsidies, taxes, tradeable quotas) would have to rely on on-farm monitoring to be effective.

6.2.2 Cereal intensification

The reduction of land use for crop production through intensification of cereal production is largely considered an autonomous development. If this development should be furthered, it could be done through regulation or subsidies that stimulate the setting aside of arable land more permanently. Increasing arable land set-aside makes the costs of using land for cereal production more expensive and consequently stimulates intensification.

6.2.3 Requirements on feed contents

Feed regulation at the EU level (EC 2003a), already regulates the use of Cu in feed. As the instrument for enforcement of a reduced maximum limit is already in place, and technical alternatives exist to reduce the overall Cu emission from dairy and pig farming to one third of the current level, tightening the regulation to achieve this level would be a straightforward measure..

In addition to the overall nitrogen management regulation described in Chapter 6.2.1, it is also suggested that protein in feed could be targeted directly, to reduce ammonia emissions. While ‘easy’ to implement on the farms, it is less obvious how to secure implementation, as an on-farm monitoring of the actual ammonia release is not realistic. However, as already suggested in Chapter 6.2.1, larger agricultural enterprises can be asked to document key figures, like N in feed and manure handling, for a model calculation of ammonia emission potential.

The proposed option of methane-reducing diets of dairy cows in order to reduce the methane emission will – from a stimulation/regulation point of view – face the same challenges as for changing feeding to reduce ammonia emission. It appears that the most efficient regulation should be part of a ‘production approval’ approach, where larger dairy farms are required to document their feeding. As the required change in feed is largely cost-neutral for the farmer, informational or management tools that integrate regard for methane emission in the regular feed optimisation procedures, could possibly be sufficient.

6.2.4 Biogasification of liquid manure

Use of manure for biogas production is already subject to public subsidies in line with the Directive on the promotion of electricity produced from renewable energy sources (EC 2001a). The level of support varies quite dramatically from country to country (EC 2005a), up to more than 100 EUR/MWh in Austria and the United Kingdom, which is close to or exceeds the production costs. An alternative instrument would be to require the establishment and use of biogas plants for farms over a certain size, e.g. as a cross-compliance measure. This would place the costs on the producer, and eventually on the consumer of meat and dairy products, in line with the ‘polluter-pays’ principle. Obviously, market access for the electricity produced would have to be ensured by the national regulation.

6.2.5 Power saving

Saving of electricity is in general also cost-saving, at least for the individual owner of the electricity-using equipment. This is also true for new energy-saving household cold appliances, which have a pay-back time of less than seven years, except for forced early replacement, as calculated in Chapter 4.5.

Since the equipment cost is an investment, while the saving is an operational expense, the main barrier for implementation of power savings is financial. This barrier is addressed by the Action Plan for Energy Efficiency (EC2006c), which includes a commitment to further appropriate financing arrangements, as well as considering the use of tax credits as incentives. The action plan also aims to remove legal barriers in national legislation to the use of ‘Energy Service Companies’ and commits the Commission to develop education and training plans and programmes for energy managers in industry. Further legislative initiatives towards the industry should not be necessary.

For the households, it appears that a financing mechanism via the consumers’ electricity payment could provide a cost-neutral incentive, as it would make the energy-saving equipment appear cheaper at the point of sale. It may even be possible that it would be sufficient to add to the energy-labelling requirements that the lifetime costs of appliances shall be presented alongside the appliance price in the same letter size. This would of course require the establishment of a standard procedure for calculating lifetime costs.

A realistic target is that all new appliances sold in Europe be either A+ or A++, as soon as the European industry can supply these. If informative and economic incentives are not enough to ensure the conversion, legislative demands banning the sale of the least efficient appliances could be used.

EU policies already focus on the electricity consumption of cold appliances, with household cold appliances being among the 14 prioritised product groups in the EU ‘Priority Action 1’ (EC 2006c). The suggested improvement option aiming at all new applicances sold being A+/A++ could well be implemented within the scope of Priority Action 1.

6.2.6 Meal planning and home delivery

The suggested improvement options of household meal planning tools and home deliveries of groceries are closely interdependent, as outlined in Chapter 4.7.2. Meal planning tools cannot be prescribed, but need to provide direct benefits and maybe even entertainment value to the user. A way to ensure development of high-quality meal planning tools could be to link their implementation to a quality-scheme for delivery services, possibly compulsory for supermarkets over a certain size. In this way, all supermarket customers would contribute to the cost of development.

A tool that could look like the start of a Household Meal Planner can already be seen at www.tesco.com. The customers can browse more than 250 recipes by course (breakfast, starters, main course, desserts & cakes) or by ingredient (dairy, fish, fruit, meat, vegetables, pasta, and grains & pulses), and it is possible to select lifestyles, such as ‘vegetarian’. By a click, it is possible to select all ingredients for the recipe (and furthermore, it is easy to de-select items you already have at home). When selecting, Guideline Daily Amounts (GDAs) are a given. The GDA is a guide to the total amount of calories, sugar, fat, saturated fat, and salt, that a typical adult should be consuming in a day. A similar ‘lunchbox selection tool’ is also available, where the customer can combine items for the lunchbox. GDAs are also shown for the lunchbox tool.

In spite of being potentially the most important of all the suggested improvement options, it is striking that household meal planning tools are not mentioned in any of the reviewed policy documents. This mirrors the very scarce research on household food losses, and on the environmental impact of household behaviour in general. The actual size of the potential improvement, as well as the necessary instruments to realise it, are therefore largely unknown, and the estimates in this study vastly uncertain. It is therefore advisable that any policy instrument in this area is first tested in a number of smaller pilot areas. Such a pilot implementation could also be used as a study ground to determine more precisely the actual food loss and the causes of this. This may point to further improvement options than the ones proposed here. Furthermore, pilot studies should investigate the time rebound effect (how meal planning and home delivery of groceries affect the household behaviour and time allocation) and the size of the welfare loss from out-of-stock situations.

Pilot studies should be performed in several cities in Europe, under different conditions, and preferably be comparative. Some of the pilot studies should be in the United Kingdom, where Internet shopping for groceries is relatively successful, to investigate the drivers for this success as well as the barriers that are encountered at the next level of market penetration. Questions to be investigated are: Why is far less than 3 % of the grocery shopping in the United Kingdom by Internet, when 20 % of the adults in the country have tried to order their groceries by Internet? What are the negative experiences? How can they be overcome? Pilot studies should include practical attempts to meet the customers' demands, not only theoretical studies on consumer opinions. Some of the pilot studies should include the use of household meal planning tools and the synergy between this and Internet shopping.

The environmental issue that led to suggesting home deliveries of groceries as an environmental advantage was its potential for reducing private car driving. However, as pointed out in Chapter 4.7.2, a large part of this saving may be lost due to the space rebound effect: that additional road space released will largely be filled by generated traffic. Nevertheless, due to its synergy with household meal planning, home delivery of groceries may still have an important role to play. In any case, pilot studies of combined home delivery and meal planning tools should also include detailed calculations of the logistics of shopping and food distribution, to determine the actual potential savings, including the traffic rebound effect and the options for preventing this.

7 INTERPRETATION, CONCLUSIONS AND RECOMMENDATIONS

7.1 Environmental impacts of meat and dairy products

Chapter 3 estimated and compared the environmental impact potentials of meat and dairy products consumed in EU-27, taking into account the entire value chain (life cycle) of these products. The methodology applied was a hybrid life cycle assessment method, which implies a system model that combines the completeness of 'top-down' input-output matrices, based on national accounting statistics combined with national emission statistics (known as NAMEA matrices), with the detailed modelling of 'bottom-up' processes from process-based life cycle assessments. For the environmental impact assessment, a combination of characterisation models was selected from two of the most recent impact assessment methods and an endpoint model that allowed linking the resulting midpoint indicators to a single value expressed in Quality Adjusted Life Years (QALYs) or monetary units (euro).

From the results in Chapter 3, it is particularly interesting to note that:

- The consumption of meat and dairy products constitutes only 6 % of the economic value of the total final consumption in EU-27, while contributing on average 24 % of the environmental impacts (with a large variation between impact categories, e.g. from 6 % for terrestrial ecotoxicity to more than 35 % for eutrophication, nature occupation and aquatic ecotoxicity).
- The monetarised environmental impacts (externalities) are of considerable size compared to the private costs of the products (from 34 % of the private costs for pork to 112 % of the private costs for beef). The large uncertainty on the monetarisation implies that this proportion can be an order of magnitude smaller or larger.
- The aggregated (monetarised) result is dominated by three impact categories: nature occupation (49 %), respiratory inorganics (23.5 %) and global warming (22.5 %), thus leaving only 5 % for all other impact categories. Using as an alternative the weights from Ecoindicator99 does not alter this picture, although slightly shifting the relative importance between the three large impact categories.
- Several sources of uncertainty influence the overall size of the aggregated result, notably the emission estimates, uncertainty on the monetary value of ecosystem impacts and on the midpoint characterisation factors for respiratory inorganics. Since these uncertainties also apply to the normalisation reference, the conclusions on the relative importance of meat and dairy products are not affected.
- The four main product groups (dairy, beef, pork and poultry products) contribute respectively 33-41 %, 16-39 %, 19-44 %, and 5-10 % to the impact of meat and dairy products consumption in EU-27 on the different environmental impact categories.
- Per kg slaughtered weight, there is a clear difference between the three types of meat, with beef having four to eight times larger environmental impacts than poultry and up to five times larger than pork. These differences are less pronounced when comparing the environmental impact intensity (impact per euro spent) of the three types of meat, where pork generally has the lowest impact intensity (down to 40 % of the impact of poultry and 23 % of the impact of beef), with the exception of aquatic ecotoxicity, where pork production contributes to high copper emissions.

7.2 Environmental improvement options, potentials and socioeconomic impacts

Chapter 4 analysed the most promising environmental improvement options for the processes that per environmental midpoint impact category contribute with more than 10 % of the total impact of meat and dairy products in EU-27. An assessment of the rebound effects and interrelationships of the improvement options (Chapter 4.7) was included, as well as their relations to autonomous developments (Chapter 4.9). Furthermore, the socioeconomic impacts of the improvement options (Chapter 5) were analysed, their relations to current policies (Chapter 6.1), and their feasibility of implementation (Chapter 6.2).

To conclude, the different improvement options are now grouped according to feasibility of implementation and ranked according to their overall environmental improvement potential and socioeconomic impacts.

From a feasibility perspective, three groups of improvement options can be discerned, in order of increasing challenges for implementation:

- Power savings, which can largely be covered by existing policy initiatives.
- Agricultural improvements, which can be achieved through different well-established and tested policy measures.
- Household improvements, mainly focusing on behavioural changes, supported by appropriate tools and regulations. This group of improvement options is the most difficult to realise as it is not covered by any current policies, would require new types of regulation, and is very scarcely researched.

Power savings have both economic and environmental benefits, the latter nearly exclusively due to reductions in global warming potential and respiratory inorganics. Even with a discount rate of 9 %, these improvement options show a net benefit.

The agricultural improvements show a more diverse picture:

- Without negative effects, and even with direct economic benefits, aquatic eutrophication may be reduced by 30 % by tightening the regulation of manure application, area use for meat and dairy production may be reduced by nearly 5 % through cereal intensification, and respiratory inorganics and global warming impacts from meat and dairy production may be reduced by 4 % and 2.5 % respectively, through methane-reducing animal diets. All other agricultural improvement options involve a trade-off between economic costs and environmental benefits.
- Although involving direct economic costs, catch crops may reduce aquatic eutrophication from meat and dairy production by nearly 4 % and show an undiscounted net benefit of this at 95 % confidence level.
- Acidification and terrestrial eutrophication from meat and dairy production may be reduced by 8-9 % through optimised protein feeding and by 14-16 % through liquid manure pH reduction. Copper reduction in dairy cattle and pig diets may reduce aquatic toxicity from meat and dairy production by two thirds. Liquid manure biogasification may reduce the global warming potential from meat and dairy production by nearly 5 %. Large uncertainties apply to the monetarisation of these impacts, so although these improvement options show a net benefit on average, this cannot be shown at 95 % confidence level.

The household improvements have a broad effect on all impact categories. Home delivery of groceries may give reductions in impacts from meat and dairy products from 3 % (for aquatic ecotoxicity) to 25 % (for non-renewable energy and mineral extraction). The effect of household meal planning tools is more modest (1-5 % for the different impact categories). An important part of the effect of 'home delivery of groceries' is the price rebound effect, i.e. that household income is tied up in payment for a service and therefore cannot be used for other spending with larger environmental impacts. The main uncertainties for these improvement options are the extent to which household behaviour can actually be affected, i.e. the assumption that the joint introduction of these measures would be accepted by 50 % of the households, and the uncertainty on the socioeconomic impacts, i.e. large, but very uncertain savings in household time usage, and possible impacts on dietary health.

In general, the socioeconomic and environmental impacts of the three groups of improvement options are roughly reverse proportional to their feasibility of implementation: power savings have a mean net benefit of EUR 2 600 millon annually, including an environmental improvement potential of approximately 0.6 % of the total environmental impact of meat and dairy products in EU-27, while agricultural and household improvements both have environmental improvement potentials of approximately 10 % when including rebound effects and synergies/dysergies, and mean annual net benefits of EUR 21 000 million and EUR 119 000 million, respectively.

When all the identified environmental improvement potentials are taken together (Table 4.23), the total improvement amounts to a reduction of 17 % for nature occupation, around 25 % for global warming and respiratory inorganics, 31 % for acidification and terrestrial eutrophication, 43 % for aquatic eutrophication, and 68 % for aquatic ecotoxicity (when rebound effects and synergies have been accounted for). Since the first three impact categories make up 95 % of the aggregated (monetarised) environmental impact, the aggregated improvement potential amounts only to 21 % of the total environmental impact of meat and dairy products in EU-27 (and significantly less if rebound effects are not accounted for). Noting that the aggregated impact from meat and dairy products amount to 24 % of the overall impact of EU-27 total final consumption, this implies that after all improvement options have been successfully implemented, the impact from meat and dairy products would still amount to 19 % of the aggregated impact of EU-27 total final consumption. This seems to suggest that large reductions in the overall impacts from meat and dairy products cannot be obtained from the identified improvement options alone, but will require targeting the level and mode of consumption as such. One of the proposed improvement options may be applicable also for this purpose, namely household meal planning tools.

In the following, each of the three groups of improvement options is described in more detail and specific recommendations made.

7.2.1 Power savings

As was already concluded in Chapter 6.2.5, the initiatives foreseen in the Action Plan for Energy Efficiency (EC2006c) are expected to be adequate to ensure the 10 % power savings in farming, food industry, retail, and catering.

For the household cold appliances, a realistic policy target could be that all new appliances sold in Europe be either A+ or A++, as soon as the European industry can supply these. Possible incentives include:

- The establishment of a standard procedure for calculating lifetime costs of appliances, and adding to the energy-labelling requirements that the lifetime costs of appliances shall be presented alongside the appliance price in same letter size.
- A European-wide scheme where the consumers can buy A++ appliances at approximately the same price as an average appliance in exchange for returning an old appliance.
- As an alternative, the energy requirements for new appliances could also be enforced by direct regulation.

7.2.2 Agricultural improvements

The following policy targets are implicit in the calculated improvement potentials:

- To reduce nitrate leaching from animal manure to an average 0.64 kg N per pig and 6.3 kg N per Mg milk produced, and from cereal production to an average 6.5 kg N per Mg cereal (53 %, 39 % and 90 % of current levels, respectively).
- To increase cereal yields to an average 4 500 kg per hectare, thus reducing the area requirement.
- To reduce ammonia emissions from pig and dairy farming to an average 0.72 kg per pig and 5 kg per Mg milk produced (43 % and 69 % of current levels).
- To reduce Cu emissions to soil from pig and dairy farming to an average 2.3 g per pig and 10 g per Mg milk produced (21 % and 44 % of current levels).
- To reduce methane emissions from dairy cows and animal manure to an average 5.3kg per pig and 18 kg per Mg milk produced, partly by ensuring that 50 % of all manure from pig and dairy farms is utilised for biogas production.

The policy targets have been expressed relative to the quantities produced, implying that the corresponding absolute emission targets will change with changes in production volume. In this way, the policy targets do not have to be revised if the production volume changes.

Instruments for achieving such targets include informational or management tools for dairy farms, integrating regard for methane emissions in the regular feed optimisation procedures. A further option is placing ‘license-to-operate’ requirements on agricultural enterprises above a certain size (in terms of acreage and animal units).

Furthermore, there is a need for investigating the options for encouraging:

- Catch-crops for spring cereals;
- Stimulating intensification of arable production where yields are comparatively low today;
- Standard farm accounting procedures for nitrogen in inputs and outputs, for copper and fat in animal diets, and for documenting manure handling procedures, to allow model calculations of emission potentials for nitrate, ammonia, and methane;
- Minimum manure N utilisation rates relative to N in artificial fertiliser of 70 % and 75 % for dairy and pig manure, respectively;

- Maximum rates of copper in dairy and pig feed 25 % above the dietary minimum requirement (noting that the cost of this recommendation may exceed the benefits, see Chapter 7.1);
- A maximum calculated methane emission potential of 24 kg per Mg milk produced;
- Establishment and use of biogas plants for liquid manure treatment.

It should be noted that while the study concludes in favour of an intensification of arable production, it does not suggest an intensification of cattle and dairy farming. This is due to the larger area requirement for feed that this would entail, and for the intensified dairy production also due to the reduced meat output which would require a compensating dedicated meat production which has relatively larger environmental impacts; see also the discussion in Chapter 4.3.4. Furthermore, an implicit premise in the study is that extensive grazing lands are expected to be maintained as such, independently of the demand for meat and dairy products. An intensification of cattle farming would therefore not yield any benefits in terms of reduced nature occupation. As the negative effects of intensification of dairy farming in general outweigh the benefits, it could be argued that it would be environmentally beneficial to restrict further specialisation in dairy farming, or at least to remove any existing incentives for such specialisation.

7.2.3 Household improvements

Household decisions have a major influence on the environmental impact from meat and dairy products. First, the very decision of buying these products instead of other foods is the driver for the entire chain of processes. Second, shopping and food preparation are important household activities with their own environmental impacts, largely dependent on household decisions. Third, the amount of food loss in households is significant and largely avoidable, as can be seen from the small losses in full-service restaurants.

At the same time, household decision-making and its environmental implications are largely unresearched areas, which implies that it is problematic to put up definite policy targets.

There is a need for comprehensive research in this area, covering at least:

- The household decision-making processes with respect to diet choices, meal planning, food shopping, meal preparation and food waste; the actual behaviour, the rationales applied, attitudes and conceptual understanding, and the environmental and dietary health implications. The research should cover the relationships to different lifestyles, socioeconomic characteristics, and geographical differences.
- Logistics of shopping and food distribution, to determine the actual potential savings, including household time saving, the traffic rebound effect and the options for preventing this, as well as the extent to which additional shopping is induced by out-of-stock situations in the household.
- The options for improving household decision-making processes and/or their environmental impacts, e.g. through information campaigns, meal planning tools, and increased household services, and what characteristics are essential for their acceptance.

- Rebound effects of changes in household decision-making, e.g. how meal planning and home delivery of groceries affect household behaviour and time allocation, and what value the households assign to the different activities.

To avoid postponing the initiation of these potentially important improvements in expectation of further research results, an important part of the above research may be carried out in connection to pilot schemes seeking to implement some of the potential improvements in smaller geographical areas.

In particular, pilot schemes could serve to study:

- The effect of different home delivery schemes, and their essential quality aspects.
- Meal planning tools and their ability to change household behaviour with respect to diet choices, meal planning, food shopping, meal preparation, and food waste.
- Different ways to stimulate development of and acceptance of meal planning tools and home delivery services, e.g. 'license-to-operate' requirements to supermarkets over a certain size.
- The relationship between meal planning tools and home delivery services.

Information campaigns should ensure the dissemination of successful experiences from the pilot projects. Consumer information may also be an important tool to prevent food being discarded because of misconceptions about freshness, colour, texture, and food safety issues. To support this, it may be useful to perform a review of national legislations to identify possible technical barriers that increase food loss or hamper the implementation of technologies that improve shelf life, such as too tight requirements on 'preferably consumed before' dates, perverse measuring standards, and demands for what may be labelled 'fresh'.

7.3 Limitations and uncertainties

The main limitations and uncertainties that may influence the conclusions of this study are analysed and described in detail in Chapters 8.6 (inventory uncertainty) and 9.4 (impact assessment uncertainty) and in the chapters describing each improvement option.

Summarising, the following uncertainties dominate the assessment of the different improvement options:

- For the majority of the improvement options, the overall uncertainty on the environmental improvement is dominated by the assumption of the degree to which the improvement option can be implemented, i.e. the area for which catch crops can be implemented, the actual cereal yields that can be achieved, the level of reduction in emissions, the extent of the power saving, and the extent that household behaviour can be affected. For the aggregated impacts, the uncertainty on the characterisation factors (see Table 9.4) are dominating.
- For improvement options involving large changes in direct production costs, the uncertainty on the cost estimates may contribute significantly to the overall uncertainty. This is particularly the case for cold appliances regulation and for biogasification of liquid manure; see also Chapter 5.4 on discounting.
- For some improvement options, the uncertainty on the socioeconomic impacts dominates the overall uncertainty. This is particularly the case for home delivery

of groceries (large, but very uncertain savings in household time usage) and meal planning tools (possibly large, but very uncertain impacts on dietary health).

The different uncertainties have been combined to yield a 95 % confidence interval on the overall assessment for each improvement option in Table 5.2. The overall confidence intervals have been calculated under the assumption that the uncertainties are independent and with normal or lognormal distribution.

Most improvement options show a net benefit at the 95 % confidence level, but due to the large uncertainties in the characterisation factors, this is not the case for the four agricultural improvement options with the largest direct economic costs: optimised protein feeding, liquid manure pH reduction, copper reduction in dairy cattle and pig diets, and liquid manure biogasification. This also makes these improvement options more sensitive to temporal discounting; see Chapter 5.4. Particularly the benefit of copper reduction is uncertain, since it depends on the impact potential of metal emissions, which may be overestimated in current characterisation models.

A number of impacts have been entirely omitted from the study (impacts from occupation of extensive grazing lands, disruption of archaeological heritage, antibiotic resistance, species dispersal, noise, pesticides transmitted through treated food, depletion of phosphate mineral resources), some have been modelled only very coarsely (all area uses treated equally, despite large differences in biological value), and some have been only qualitatively touched upon (erosion and water balance). Likewise, a number of rebound effects, synergies/dysergies, and socioeconomic impacts have not been quantified, but only described qualitatively in Chapters 4.7 and 5. It is likely that these shortcomings mainly bias the study results towards a smaller overall impact and smaller overall improvement potentials relative to the result if these impacts had been quantified. It is not expected that inclusion of these impacts would change the overall conclusions of this study.

PART II: METHODOLOGY

8 SYSTEM MODEL OF THE PRODUCTION AND CONSUMPTION OF MEAT AND DAIRY PRODUCTS (LIFE CYCLE INVENTORY)

8.1 Introduction

This chapter describes the life cycle inventory analysis methodology according to the ISO standard on life cycle assessment (LCA).

ISO 14040/14044 rules are applied, as well as – when relevant – the supplementary assumptions and procedures outlined in the Danish LCA inventory guidelines (Weidema 2003), which are the result of an extensive scientific consensus process.

All requirements of the ISO 14040-series have been adhered to, with the exception of formal third-party critical review that was not performed.

The life cycle inventory is produced by a hybrid method, which implies a system model that combines the completeness of 'top-down' input-output matrices of the EU-27 economy, based on national accounting statistics together with national emission statistics (known as NAMEA matrices), with the detailed modelling of specific 'bottom-up' processes from traditional process-based life cycle assessments. Both elements are combined so that they fit official statistics on the production and consumption of meat and dairy products in EU-27.

The different elements and how they are combined are described in the following sub-chapters. Chapter 8.2 specifies the data used for the production and consumption of meat and dairy products in EU-27. Chapter 8.3 describes the input-output matrices for EU-27 and how they were set up. Chapter 8.4 describes in detail the data sources and models for the specific processes and how these processes are integrated in the input-output framework. Chapter 8.5 describes additional data used for the modelling of the environmental improvement options. Chapter 8.6 discusses the uncertainties in the life cycle inventory data and Chapter 8.7 describes the additional data used when analysing autonomous developments.

8.2 Production and consumption of meat and dairy products in EU-27

Data on both production and consumption of meat and dairy products have been used from Faostat (2006). The Faostat data are complete and available for each individual country in EU-27. They are aggregated at the level of animal types (e.g. cattle meat, pig meat, chicken meat, all given in slaughtered weight, and raw milk-equivalents).

Data on consumption would also be available at a more detailed commodity level from the Eurostat household budget surveys (Eurostat 2004) and similarly production data are available from the Prodcom database (Eurostat 2006). Unfortunately, the most recent household budget survey is from 1999, and the Prodcom data is incomplete due to the suppression of confidential information. This makes it very difficult to use the Eurostat data to arrive at complete and updated values for all meat and dairy products.

Since this study looks at the entire consumption of meat and dairy products in EU-27, and since all the different meat products of a specific type (cattle, pig, chicken) are produced from the same slaughtered animals (and all dairy products from the same raw milk) with the same environmental impacts, it appears unnecessary to subdivide the consumption data further than has been done in the Faostat data.

The Faostat data for 2004 are reported in Table 8.1. The difference between production and consumption is accounted for as net export.

Table 8.1: Production and consumption of meat and dairy products in EU-27 in 2004.

Product group	Bovine meat	Pork	Poultry meat[1]	Dairy products
Unit	Gg slaughtered weight[2]	Gg slaughtered weight[2]	Gg slaughtered weight[2]	Gg raw milk equivalents
Production in EU-27	8 250	22 000	10 600	153 000
Consumption in EU-27	6 420	15 300	9 260	114 000
Consumption in % of production	78	69.5	87	75

1. Includes meat from rabbits and other minor unspecified animals.
2. Slaughtered weight does not include edible offal, which is also a consumed commodity included in this study. Edible offal constitutes approximately 5 % of the total meat consumption.

8.3 Environmentally extended input-output tables for the EU-27 (EU-27 NAMEA)

The input-output table for EU-27, distinguishing 60 sectors, was constructed by aggregating the available 20 national input-output tables, assuming that the remaining 2 % of the output can be represented by the average of these 20 countries. (See Table 8.3 for an overview of the countries for which data were available.) Input-output tables in national currencies were converted to euro, using the mid-year exchange rate as provided by Eurostat. For four countries (United Kingdom, Spain, Slovakia, Malta), the IO table has been constructed from the national supply-use tables, distributing transport and trade margins over the relevant industries, and for the United Kingdom and Spain re-allocating the use table applying the industry-technology assumption. In two cases, the original table was for 1997 (Estonia) or 1998 (Greece), and the input coefficients from these years have been used, but the total table multiplied up to the national output of 2000. For four countries (Ireland, Austria, Poland, and Sweden), where the original IO table contains some industries more aggregated than the 60-industry level, the aggregated industries were disaggregated in proportion to the average output from the remaining 16 countries.

Table 8.2: Overall economic output and availability of NAMEA and input-output data for EU-27 countries. Unless otherwise specified, data on economic output taken from the input-output (IO) tables, and IO tables taken from Eurostat (2006).

Country	Economic output year 2000 (MEUR)	% of EU-27 output	Accumulated %	Data availability
Germany	3,650,460	21.9%	22%	NAMEA
United Kingdom	2,908,456	17.4%	39%	NAMEA[3]
Italy	2,200,709	13.2%	53%	NAMEA
Spain	1,120,716	6.7%	59%	NAMEA[4]
Netherlands	759,501	4.6%	64%	NAMEA
Sweden	489,361	2.9%	67%	NAMEA
Denmark	289,967	1.7%	68%	NAMEA
Hungary	111,995	0.7%	69%	NAMEA
France	2,544,398	15.3%	84%	IO only
Belgium	540,420	3.2%	88%	IO only
Austria	362,790	2.2%	90%	IO only
Poland	353,444	2.1%	92%	IO only
Finland	250,018	1.5%	93%	IO only
Portugal[1]	230,803	1.4%	95%	IO only
Ireland	211,936	1.3%	96%	IO only[5]
Greece[1]	188,525	1.1%	97%	IO only[6]
Slovakia	51,724	0.3%	98%	IO only[7]
Slovenia	42,142	0.3%	98%	IO only
Estonia[1]	13,321	0.1%	98%	IO only[6]
Malta	8,055	0.05%	98%	IO only[7]
Czech Republic[1]	149,330	0.9%	99%	neither
Romania[2]	74,173	0.4%	99%	neither
Luxembourg[2]	39,868	0.2%	99.5%	neither
Bulgaria[1]	27,937	0.2%	99.7%	neither
Lithuania[1]	20,702	0.1%	99.8%	neither
Cyprus[2]	18,191	0.1%	99.9%	neither
Latvia[1]	16,193	0.1%	100.0%	neither
Sum for EU-27	16,675,136			

1. Data on output from Eurostat (Updated 5 September 2006)
2. Data on output estimated as GDP*1.81, which is the average ratio between output and GDP of the EU-15 countries for which both GDP and output data are available from Eurostat.
3. Input-output table calculated using the industry-technology model assumption with basis in a supply table constructed applying RAS methodology to the 76x76 supply tables estimated by Cambridge Econometrics and further processed by Stockholm Environment Institute, and the use tables obtained 8 September 2006 from: http://www.statistics.gov.uk/about/methodology_by_theme/inputoutput/latestdata.asp.
4. Input-output table not available, but were constructed from the available symmetrical Supply-Use tables using the industry-technology model assumption.
5. Input-output table obtained from http://www.cso.ie/releasespublications/2000_input_output_table.htm; accessed 8 September 2006.
6. The most recent available table is for year 1997 (Estonia) and 1998 (Greece).
7. Input-output table not available, but an unallocated proxy was constructed from the available use table. Due to the relatively small economic output of this country, it was decided not to apply the more time-consuming allocation procedures involving technology model assumptions

To this EU-27 input-output table, environmental coefficients (e.g. kg CO_2/EUR) were added, calculated per industry as the output-weighted environmental coefficients from those countries for which the specific environmental data were available. For some countries and industries, not all environmental data were available at the same level of aggregation as the input-output data (60 industries), but only at a more aggregated level. In these instances, the aggregated environmental data were distributed over the underlying industries in proportion to the environmental coefficients calculated for those countries where disaggregated data were available. Table 8.3 provides an overview of the original country-wise availability of environmental data. These data were supplemented with data from EIPRO (Tukker et al. 2006). This has been particularly relevant in order to include emissions of pesticides and heavy metals to soil.

To account separately for imports to EU-27, the part of the imports in each national IO table that were reported as Extra-EU imports was isolated and linked to the US data provided by Suh (2003). It was judged that of the available non-EU data, the US data were superior due to the large number of emissions for which data were included and the relatively large completeness of the US economy in terms of industries covered (due to its size, practically all kinds of industries are found within the country). A rough correspondence matrix was applied to link the imports classified at the level of 60 industries to the 480 industries of the US table, assuming that within each of the 60 industries, the distribution of the import is the same as the distribution of the output of the corresponding US industries. Data for aluminium extraction, injuries, and value added were added to the US data, to obtain the same completeness in inventory parameters as in the European data. Emissions to industrial soil in the US data were eliminated, as these were dominated by heavy metal emissions from mining, an emission type that is not covered by the impact assessment method; see Chapter 9.1 for a more detailed discussion of this issue.

Table 8.3: Availability and data sources of environmental data for the EU-27 NAMEA.

Environmental compartment	Inventory item	Coverage of original data	Data source
Mineral resources	Aluminium, copper and iron	EU-27	USGS Minerals Yearbook 2004, Vol. I
Energy resources	Hard and brown coal, natural gas and oil	EU-27	Eurostat (2006): Energy: yearly statistics
Air	CO_2, N_2O, CH_4, NOx, SO_2	DE, DK, ES, GB, HU, IT, NL, SE	ETC/RWM (2005) & SE[4]
Air	NH_3	DE, DK[1], ES[1], GB, IT, NL[1], SE[1]	ETC/RWM (2005) & SE[4]
Air	NMVOC	DE, DK, ES[2], GB[2], HU[2], IT, SE[2]	ETC/RWM (2005) & SE[4]
Air	CO	DE, DK, ES, GB, IT, SE	ETC/RWM (2005) & SE[4]
Air	PM10	DE, GB, IT, SE[1]	ETC/RWM (2005) & SE[4]
Air	Pb	GB, IT	ETC/RWM (2005)
Air	As, Cd, Cr, Cu, Hg, Ni, Se, Va, Zn	GB	ETC/RWM (2005)
Air	PAH	DK[3]	Weidema et al. (2005a)
Water	N-tot, P-tot	DK[3], NL	Weidema et al. (2005a)
Soil	Heavy metals and pesticides	EU-15	Tukker et al. (2006)
Social	Injuries, road or work	EU-15	Eurostat and CARE road database
Economic	Value added	as for IO-tables except IT	IO-tables

1. Not reported for all 60 industries. Average coefficient of other countries used as proxy.
2. Data for forestry (and in case of Spain also for agriculture) ignored, as they appear to include emissions from vegetation.
3. Data for 1999.
4. SE: http://www.mirdata.scb.se, accessed 2 August 2007.

The resulting EU-27 database has been checked using an EU-27 input-output table that was constructed by the Institute of Prospective Technology Studies concurrently with this work. Only minor discrepancies were found. In particular, the database of this study allocates approximately 2 % more of the total intermediate inputs to imports from non-EU countries. Such differences can be attributed to estimates and modelling assumptions that are necessary when the original statistics are incomplete. The database of this study provides results that are slightly larger (on average 9 % more environmental impacts), mainly due to the larger import.

For comparison, a version of the database was also made with the standard assumption that foreign industries have the same input requirements and environmental impacts as the corresponding European industry. Since the European industries generally have lower emissions than the foreign, the resulting database provided results with substantially less environmental impacts, on average 70 % of the results from the database with specifically modelled imports. This demonstrates the importance of the imports, and also stresses the importance of the uncertainty associated with the modelling of imports.

Annex I provides a list of all industries in the EU-27 input-output table, indicating also which of these that have been disaggregated with specific process models (see Chapter 8.4).

8.4 Data sources and models of specific processes

This section describes in detail how the specific processes were modelled, and into which the sectors of the original 60 sector NAMEA were subdivided, the data used, and how the processes were integrated in the input-output framework.

For the input-output tables, each industry is generally assumed to produce one homogeneous product, i.e. all products have the same environmental impact per euro. This is equivalent to an economic allocation over the co-products.

While this assumption may be reasonable in general, it may lead to misallocation of environmental impacts for industries that are in reality very inhomogeneous. For example, the emission coefficients for the meat industry are very different from those of the sugar industry; yet, in the 60-industry input-output tables, they are both subsumed under ‘food products and beverages’.

The most important inhomogeneous industries (‘agriculture, hunting and related services’, ‘food products and beverages’ and ‘hotels and restaurant services’) have therefore been subdivided. Furthermore, waste management processes and household processes have been introduced. This has resulted in a LCA/IO hybrid model with 110 processes/sectors.

Regarding the specific livestock production processes, a range of production systems were modelled based on well-documented biological input-output relations, such as nutrient balances. These production systems have then been scaled to the level of EU-27, to fit the production volume, area, and number of livestock given by Faostat. It was found that it was appropriate to represent the dairy production by five systems (Chapter 8.4.1), the beef production by two suckler systems and two systems for specialised fattening units (Chapter 8.4.2), and the pig production by three systems (Chapter 8.4.3).

In addition, an average broiler production system and a cereal production system are defined (Chapter 8.4.4) ([1]).

For the food industry, nitrogen fertiliser industry, restaurants and catering, specific models have been derived by sub-dividing the original data from the EU-27 NAMEA using more detailed NAMEAs from certain countries and LCA databases (Chapters 8.4.5 to 8.4.7).

Specific estimates have been made for own-account transport performed within each industry (Chapter 8.4.8), consumer transport (Chapter 8.4.9), household processes (Chapter 8.4.10), end-use losses (Chapter 8.4.11) and treatment of food and packaging waste (Chapter 8.4.12).

8.4.1 Dairy farming

Of the total milk production in EU-25, 85 % is produced within EU-15 countries. Five countries – Germany, France, United Kingdom, Netherlands and Italy – produce more than 60 % of the EU-25 milk (EC 2006b). Outside EU-15 Poland is the main producer with 8 % of EU-25 production. Among and within these countries, production conditions differ.

The dairy production systems in Germany, France, United Kingdom and the Netherlands are described in more detail in Bos et al. (2005), through the activities of a trans-European working group. Based on this information, four typologies of dairy farming systems were developed taking into account the farm and herd structure reported (number of dairy cows and beef fatteners, land use, concentrate use and use of fertiliser). The country labels have been maintained for ease of identification but the typologies should not be seen as averages for the said countries. For each typology a coherent input-output relation was established based on assumed yield of fodder crops, assumed utilisation of the N circulating within the farm and using well accepted biological efficiencies in feed (and protein) use for production of milk and beef.

It was assumed that these four typologies could represent the dairy production in the western and the central-western parts of Europe and the Scandinavian countries (AT, BE, CZ, DE, DK, ES, FI, FR, GB, IE, NL, PT, and SE). These countries account for 74 % of the milk produced in EU-27 and 67 % of the total number of dairy cows in 2003 according to Faostat (2006). The number of cows and volume of milk not included in the aforementioned systems is used to calculate the average milk production per cow for the remaining milk production in EU-27. This is then regarded as one typology, assumed to represent the milk production in Eastern and South-Eastern Europe, while noting that many different production systems are present in these countries. However, since the major part of milk production is represented by the aforementioned well-defined systems, it was found to be an acceptable approximation to represent the total EU-27 milk production in this way.

([1]) While these systems are all compatible with the Farm Accounting Data Network (FADN), in the sense that each system fits into one of the officially defined systems, there is no direct correspondence with the data appearing in the FADN database, and the way these data sum up to the total EU agricultural production. The FADN data were surveyed for this purpose, but were found inadequate to model the bio-physical turn-over to the degree needed to establish the environmental impact of the primary agricultural production. The FADN has a very complex composition of agricultural activities and lacks the necessary bio-physical data, besides the interpretation of classifications appearing to be inconsistently applied between countries.

Some important base line characteristics are given in Table 8.4. Biological N fixation was assumed to amount to 10 kg/ha after van Egmond et al. (2002). Of the model-estimated meat production from the dairy farming systems, 90 % is estimated to be sold as beef, the 10 % accounting for deaths of calves and non-accepted carcasses at the slaughterhouse.

The emissions for the dairy farms were estimated based on the type-specific feed intake, manure handling, use of fertiliser and land use, as specified in Chapter 8.4.4.

Table 8.4: Typologies of European dairy farming systems (developed after Bos et al. 2005). The typologies do not represent averages of any specific country.

	Central EU (model DE)	Western EU (model FR)	UK-type (model UK)	Lowland (model NL)	Eastern & Southern EU
Total annual production volume in EU-27 (Tg):	42	34	23	14	40
Land use, ha per dairy cow:					
- Grass, rotation	0.46	0.30	0.09	-	0.40
- Grass, permanent	0.39	0.30	0.84	0.47	0.40
- Maize (silage)	0.26	0.53	-	0.12	0.26
- Cereals (mature)	0.20	0.37	-	-	0.26
Herd:					
- Dairy cows (no.)	38	36	104	64	36
- Replacement (no.) (2)	38	36	104	56	36
- Fatteners (no.)	14	8	15	0	0
Milk yield, kg/cow	6 585	6 133	6 665	7 415	4 600
Beef, kg/cow	458	403	362	285	320
Cereals sold, kg/cow	279	2500	-	-	-
Feed, kg DEM/cow/year:					
- Grass grazed	2 654	990	2 854	1 330	1 600
- Grass silage	1 872	1 710	3 693	2 470	1 000
- Maize, silage	2 368	4 725	-	1 306	2 133
- Cereals/Concentrate	1 500	1 150	1 500	2 150	1 742
Yield/ha; kg DEM/year:					
- Maize, silage	9 000	9 000	-	11 000	8 000
- Grass rotation	7 000	6 500	9 000	-	5 000
- Grass permanent	3 300	2 500	6 800	8 000	1 500
- Cereals	5 000	7 000	-	-	4 000
N-application (1) kg N/ha/year:					
- Maize	240	240	240	240	160
- Grass, in rotation	220	220	220	220	180
- Grass, permanent	150	150	250	350	70
- Cereals	130	130	130	130	110
N-balance kg N/ha/year:	143	142	173	265	83

1. Including utilisable N from farm produced manure (assumed N utilisation: 60 % for collected manure and 20 % for manure deposited on grassland) and including an estimated biological N fixation of in average 10 kg N/ha.
2. This includes female calves and heifers from 0 to 27 months of age.

8.4.2 Beef/veal production

More than 90 % of the total EU-25 consumption of beef/veal is produced in EU-15 countries. An increase is expected in the import (EC 2006a). The major suppliers (in descending order) are France, Germany, Italy, United Kingdom, Spain, and Ireland, representing together 75 % of total EU-25 production. The beef systems in these countries therefore largely represent the total systems for beef production.

Beef production systems differ concerning the age and weight at which animals are slaughtered, the method of feeding, and the type of housing. Two main categories exist, depending on whether the animals come from dairy farms or from suckler herds.

The typical European beef production systems are described in quite some detail in EC (2001b). Although the description was meant for evaluating animal health and welfare issues, the description also serves the present purpose.

The main suckler systems can be described as (1) a quite intensive system where the offspring are fed a cereal based diet and slaughtered at about 12 months of age, or (2) an extensive system based on grassland and modest amounts of concentrates, and where the offspring are slaughtered about 16 months of age. It is difficult to estimate the respective share of animals in intensive and extensive systems. Ireland and the United Kingdom, with approximately one million beef cattle each, can be characterised as extensive systems. Spain (with 1.7 million beef cattle) and Italy (with 0.4 million beef cattle) can be characterised as intensive systems. Germany (with 0.7 million beef cattle) and France (with 4.1 million beef cattle) can be considered intermediate. Based on these descriptions – and taking into account that these countries are the major beef producers – it is estimated that 50 % of suckler herds in EU are reared intensively and 50 % are reared extensively.

A large proportion of the male offspring of the 26 million European dairy cows (Holstein/Friesian and dual purpose cows) are destined for beef fattening units. These offspring are separated from their mothers at one or two days of age and artificially reared on milk or milk replacers plus solid food for a six to nine week period. These animals subsequently enter beef fattening systems, where they are fed on a diet of solid food, i.e. forage (hay, straw, grass, silage) or forage plus concentrates.

Because of the effects of trade between EU countries, the number of fattening bulls as a proportion of the number of cows varies between EU countries. For example, France has a small proportion of fattening cattle relative to its cow population. In contrast, Italy has a large proportion of fattening cattle relative to its cow population because of the movement of young animals between the two countries. Therefore, it is difficult to establish typologies for beef production based on the information from individual countries.

For the present purpose, the detailed description of beef rearing systems in EC (2001b) was used. The major proportion of the dairy bull calves and surplus heifers are reared on a mixed diet aiming at a slaughter age of approximately 16 months. A smaller proportion of the dairy calves are reared more intensively, based on cereal feeding, and aiming for slaughter at 12 months. Particularly in the United Kingdom and Ireland, the main production system is steers fattened on grassland. These include steers from dairy herds as well as from suckler herds.

Each of the aforementioned two suckler herd systems (intensively and extensively reared offspring) and each of two other fattening systems (offspring reared on a mixed diet and slaughtered at 16 months of age and offspring reared as steers) were described in more detail as for the dairy systems. It was assumed that fatteners already accounted for in the dairy typologies represented intensively reared bull calves slaughtered at an age of 12 months. Key figures of the four other systems are given in Table 8.5.

Land area has been included to the extent necessary to support the roughage needed and the land available for manure application. Thus, the amount of land area included is only that intensively used, either for fodder crops or intensive grazing. Out of the total European pasture area of more than 75 million ha, only 21.6 million ha pasture is included in the livestock model. This implies that at least 54 million ha of extensive grazing lands are not included in this study, i.e. land regarded as being maintained as extensive grazing area independently of the demand for meat and dairy products.

Table 8.5: Key figures for beef production systems

	Beef fattening; 16 months at slaughter		**Steers, grassland based; 24 months at slaughter**		**Suckler herds: Cereal based fattening; 12 months at slaughter**		**Suckler herds: Roughage based fattening; 16 months at slaughter**	
Total annual production volume in EU-27 (Gg slaughtered weight)	764[1]		395[2]		1 332[3]		1 434[3]	
Intake (kg DEM):	*per animal and year*				*per suckler cow and year*			
Grass grazed	-		1 000		2 155		2 477	
Grass, silage	300		1 000		1 202		1 495	
Maize, silage	1 125		-		-		6 60	
Cereal	550		300		1 368		619	
Soy meal	150		0		56		3	
Output (kg):								
Slaughter-weight	268		316		217 290 357	female male cow	239 316 357	female male cow
Effective carcass produced4 per year	191		158		239		222	
Yield per ha, DEM (ha used):								
Grass, grazed	-		6 800	(0.14)	3 000	(0.87)	3 000	(1.02)
Grass, silage	8 000	(0.04)	8 000	(0.13)	8 000		8 000	
Maize	11 000	(0.10)		-	-		11 000	(0.06)
Cereal	6 000	(0.09)	6 000	(0.05)	6 000	(0.23)	6 000	(0.10)
N-balance kg N/ha:	68		76		107		113	

1. Calculated from the dairy farming production of 11.5 million calves, minus the five million that are included in the dairy systems, minus 2.5 million steers.
2. 2.5 million steers (EC 2001b).
3. EC (2001b), assuming an equal amount of six million calves in each system.
4. The effective carcass weight is corrected for assumed deaths of calves and non-accepted carcasses at the slaughterhouse due to illness (5 % for beef fatteners and 10 % for suckler herds).

The total meat production in each system was estimated from the total number of suckler cows and dairy cows, respectively, and from the slaughter-weights of different categories compared with the overall production of carcass in EU-27. This is shown in Table 8.5.

8.4.3 Pig farming

Major players in pork production within EU-27 are, in descending order, Germany, Spain, France, Poland and Denmark. Combined, they account for more than 60 % of EU-27 production (FAO 2006). There is practically no import and a considerable export of pork from EU-25 (EC 2006b).

Pig production in Europe takes place mainly in large specialised units following the same system, although differences may occur in feed composition and manure handling. Differences may also occur in the overall system; for example, the United Kingdom has a tradition of outdoor sow keeping. Guy & Edwards (2006) estimate that 30 % of the sows in the United Kingdom are kept on free range. However, the progeny are almost entirely reared under indoor conditions and since the United Kingdom represents only 3 % of total EU-27 pig production (FAO, 2006), it is not necessary to consider this system in detail for the present purpose. The same is true for the traditionally extensive production systems in Mediterranean Countries.

Dourmad et al. (1999) evaluated the N efficiency in pig production at animal level in France, Netherlands and Denmark and observed no important difference in g N lost per kg pig produced. Weidema et al. (2005b) compared pig production in Denmark, Netherlands and Spain. They observed the same technical efficiency in Denmark and Netherlands but a lower efficiency in Spain in terms of piglets born per sow and a slightly larger feed consumption per kg gain.

The other important aspect in relation to environmental impact is the manure handling techniques. Weidema et al. (2005b) estimated a considerably larger emission of ammonia from pig houses and the manure storing facilities under Spanish conditions than under conditions in the Netherlands and Denmark.

For the present purpose it is found appropriate to represent the bulk of the European pig production by three systems: one system with optimised feed and a high efficiency in manure handling – assumed to cover the situation in Germany, France and Denmark and other North-Western European countries; one system with sub-optimal manure handling efficiency – assumed to cover roughly the situation in Spain and Southern European Countries; and one system with reduced feed efficiency and sub-optimal manure handling – assumed to cover roughly the situation in Poland and other Eastern European countries. The systems are described in more detail in Table 8.6.

Table 8.6 Key assumptions for pig production systems.

System	'Intensive' (NW Europe)	'High ammonia' (Southern Europe)	'Low efficiency' (Central Europe)
Total annual production volume in EU-27 (Tg slaughtered weight)	15	4	3
No. of pigs (millions)	196	58	45
Feed use[1]/kg pork live weight	3.03	3.03	3.35
N-surplus ex animal, per 100 kg pork live weight produced, kg	5.03	5.66	5.74
NH_3-losses (stables, storage, spreading)	25 %	35 %	25 %
Manure N utilisation in crops (% replacement of fertiliser N)	60 %	40 %	40 %

1. Including consumption by sows and piglets.

For simplicity is was assumed that the pork production took place in very specialised units where all feed was bought in to the farm and all manure was exported out of the farm to be used in cereal production on arable farms. Assessments based on data in the Danish LCA-FOOD database (Nielsen et al. 2003), which include pig systems differing in land use (and manure export), shows that this does not change the outcome of the assessment. The differences in ammonia losses are related to temperature and manure handling technique. The utilisation of the produced manure N in cereal production on arable farms is affected by assumed differences in environmental regulations in different regions in Europe (limits of manure use per ha) and the applied technology for manure spreading.

For a long time, copper has been included in pig's diet at a rate of 150-250 ppm because of its growth promoting effects (Dourmad & Jondreville 2006). This has resulted in large concentrations of copper in the manure, since less than 1 % is retained in the carcass. Since 2003, EU regulation limits the rate of copper incorporation in the diets (EC 2003a). Dourmad & Jondreville (2006) estimated that the implementation of the EU regulation no 1334/2003 (EC 2003a) would result in a decrease in manure copper concentration from approximately 900 ppm in manure DEM to 400 ppm. Spanish data (Naves & Torres 1999) showed a concentration of 430-624 ppm in DEM from pig farms. Recent Danish data from pig farms showed a concentration of 460 ppm in DEM (Landskontoret for Planteavl 2006). These data indicate that it is reasonable to assume that use of copper and concentration in manure to a large extent follows the maximum limits set by the recent EU regulation.

8.4.4 Agriculture in general

In addition to the five dairy farming systems, four beef production systems, and three pig farming systems, a poultry system and a cereal production system were defined.

The technical coefficients for the poultry system are documented in the Danish LCA Food Database (Nielsen et al. 2003).

Cereal is an important constituent of feed and the major part is produced within EU. In addition, for simplicity, complementary feed to roughage can in most cases be composed by cereal and soy bean products; and even if other feedstuffs may be important as complementary feed, the cereal-soy bean mix can be considered as the marginal livestock feed – meaning that an increase or decrease in livestock production mainly influences the demand for cereal and soy bean products. For the present purpose, it was found justified to consider only an average cereal crop, since the input of fertiliser (which accounts for a major part of the environmental impact from cereal production) can be considered proportional to the cereal yield, and thus the main environmental impacts expressed per kg of cereal produced does not necessarily differ much across crops (see Chapter 4.2.2). An average yield of 4 100 kg/ha with an input of 100 kg N/ha in fertiliser and manure was estimated according to Faostat. It was assumed that biological N fixation amounted to on average 10 kg/ha for all land use, following Egmond et al (2002).

As for the systems described earlier, the internal consistency of the poultry and cereal systems is ensured, in terms of nutrient balances. The systems are documented in Table 8.7.

Finally, a residual was calculated, accounting for the remaining agricultural production, including horticulture and permanent crops. The residual accounts for the remaining use of land area and fertiliser consumption up to EU-27 totals as provided by Faostat and Eurostat, respectively. The residual, named 'agricultural products n.e.c.', is documented in Table 8.7. The plausibility and physical consistency of the residual, in terms of nutrient balance, has been checked.

Table 8.7: Poultry and cereal production models for EU-27 and calculated residual for remaining agricultural products.

	Unit	Poultry farms	Cereal production	Agricultural products n.e.c.
Area, in rotation	km^2		536 000	222 000
Poultry population	No (at 2 kg live weight)	7 510 million		
Products per year:				
Cereal	Gg		219 760	
Poultry	Gg, slaugthered weight	10 514		
Manure	Gg N	139		
Inputs:				
Cereal	Gg	16 920		
Soy meal	Gg	7 352		
Vegetable oil	Gg	1 337		
Manure[a]	Gg N		644	267
Fertiliser N	Gg N		4 717	2 333[b]
Emissions to air per year:				
Methane	Gg CH_4	4.5		418[c]
Ammonia	Gg NH_3	201	497	558[d]
N_2O	Gg N_2O	6.4	221	107
Emissions to water:				
Nitrate	Gg NO_3	644	7 075	3 308
Phosphate	Gg PO_4	14	78	32

a. Surplus from all livestock systems distributed relative to the area in rotation.
b. Calculated as a residual compared to EU-27 total from Eurostat.
c. Calculated as a residual compared to EU-27 total from the national inventories. The amount represents sheep/goats and other small ruminants.
d. Calculated as a residual compared to EU-27 total from the national inventories. The amount represents emissions from egg production, sheep/goats and other small ruminants.

The emissions for both livestock and arable farming systems have been calculated in the following way:

- Methane: Based on IPCC (1997). For cattle, the amount is a combination of a fraction of energy intake ((DEM * 18.45 MJ * 0.06)/55.65) and DEM in manure deposited in stable (0.007 kg CH_4/kg) and on grass (0.0015 kg CH_4/kg). For pigs, a fixed amount of 6.5 kg CH_4 per produced pig is used (after Weidema et al. 2005b)
- Ammonia: Based on Andersen et al. (2001) as a fraction of N deposited in stables and on grassland, respectively, according to livestock species and geographical differences, plus a fixed amount per ha crop use (grassland 3 kg N/ha; other 5 kg/ha) and a fraction (3 %) of fertiliser used.
- N_2O: Based on IPCC (2000) as a fraction of N deposited in stables depending on livestock group, a fraction (0.02) of N deposited on grassland, a fraction (0.0125) of N in manure storage minus ammonia loss, a fraction (0.0125) of N in fertiliser used, a fixed amount (0.7 kg) per ha used, representing crop residues, a fraction (0.025) of NO_3 leached, and a fraction (0.01) of NH_3 lost.
- NO_3 leaching: Calculated as a difference from the total N balance, i.e. N input minus N in farm outputs (products and manure), minus NH_3–N, minus de-nitrification (fixed 10 kg per ha for livestock farms and 6 kg per ha for arable farms), and minus the build-up in soil. On livestock farms the latter was estimated

as a fixed 5 kg/ha/year + 0.8 kg per percentage of grassland in total land use. At specialised cereal farms, a negative build-up in soil of 5 kg/ha/year was assumed. For landless production (pig and poultry), the contribution to leaching (when manure is used on arable farms) is calculated as the difference between total surplus N (less ammonia emission) and the substituted fertiliser-N.

- PO_4 leached: 7.25 % of the P-balance. This figure takes into account the relatively large emissions in lowland areas such as the Netherlands.
- Cu to soil: Recent data from Landskontoret for Planteavl (2006) on copper in pig and dairy liquid manure. For beef production systems calculated from the nutritional requirements of the livestock. For arable farming, an average of 30 g per ha, as an additive in artificial fertilisers.

Changes in the soil carbon pool have not been accounted for, although this may be influenced by different farming practises, because the available models are not yet well-established. Where included in national reporting, often only particular soil types are included, like organic soils or wetlands. Danish experiences (Petersen et al. 2006) are that 'livestock-crop rotations', including the area for feed-cereals, in total will have a zero net impact on soil carbon pools (grass based and manure rich crop rotations increase C-pool, whereas cereal production decreases). This means that for the present purpose, where the functional unit is the entire meat and dairy production in Europe, the net result is expected to be very modest.

The above-described farm models were embedded into the EU-27 NAMEA as a subdivision of the industry 'products of agriculture, hunting and related services', applying the average price of the agricultural products and allocation parameters for inputs and environmental exchanges, as shown in Table 8.8.

Table 8.8: Prices of agricultural products and allocation parameters applied to embed the farm models into the NAMEA.

Products	**Prices (EUR/kg)**
Beef, live weight	1.375
Poultry, live weight	0.714
Pigs, live weight	0.966
Milk	0.272
Cereal	0.1091
Inputs	**Allocation parameter**
Seeds for sowing (10 % of internal agricultural trade), inputs from forestry, inputs of energy carriers	Agricultural area, not including permanent grass
Inputs from chemical industry	Fertiliser use, N
Inputs from food industry (fodder) and wood industry	Value of animal output
Cereal input (37 % of internal agricultural trade)	Specified by livestock model
3 % of internal trade in agriculture (horticultural products), inputs from mining	Exclusively to 'agricultural products n.e.c.'
Agricultural services (50 % of internal agricultural trade) and all inputs not specified above	Total output value
Emissions	**Allocation parameter**
Cereal related herbicides	Cereal area (incl. for silage)
Emissions related to fuel combustion (heavy metals, NOx, NMVOC, PAH, PM10, SO_2), insecticides	Agricultural area, not including permanent grass
NH_3, N_2O, CH_4, N and P to water, Cu to soil	Specified in livestock model
All emissions not specified above	Total output value

8.4.5 Food industry

The industry 'food products and beverages' in the 60-industry input-output tables has been subdivided to provide specific modelling of the meat and dairy industries. The subdivision has been made by applying, for each input and emission coefficient, the same proportions between the new sub-industries as found in the more detailed Danish NAMEA (Weidema et al. 2005a). The resulting new sub-industries have been embedded into the overall EU-27 input-output table, using the proportional input coefficients for each of the sub-industries as found in the Danish NAMEA. See Annex I for a list of all processes.

When using data from the input-output tables, by-products from the food industry are automatically included as inputs to agriculture, thus reflecting the average benefit of recovering these by-products, which otherwise would have been classified as wastes.

8.4.6 Chemical industry

In the 60-industry input-output tables, nitrogen fertiliser production is included in the industry 'chemical products and man-made fibres'. This industry has been subdivided into 'N-fertiliser' and a residual 'chemicals and man-made fibres n.e.c.' to provide a more specific modelling of Nitrogen fertiliser production. The subdivision has been made by identifying the primary energy requirement and the most important process-specific emissions (ammonia, arsenic, CO_2, copper, N_2O, nickel, NOx, and particulates) for the fertiliser industry as described in the Ecoinvent data (Althaus et al. 2004). The primary energy requirement and emissions have been scaled to the level of the EU fertiliser production in the year 2000 (a total of 11.8 Tg with a distribution of 3 % ammonium sulphate, 30 % urea, 46 % ammonium nitrate, and 21 % calcium ammonium nitrate) and subtracted from the totals of the industry ‘chemical products and man-made fibres’. The resulting new sub-industries have been embedded into the overall EU-27 input-output table. For three emissions (ammonia, N_2O and nickel), the totals for the fertiliser industry calculated from the Ecoinvent data exceeded the amount reported for the industry ‘chemical products and man-made fibres’ in the EU-27 NAMEA, and the entire amount was therefore allocated to the fertiliser industry. Although this suggests an underestimate of the emissions of the chemical industry in the EU-27 NAMEA, the original total emissions have been maintained, because the residual chemical industry does not contribute significantly to the environmental impacts of meat and dairy products.

8.4.7 Restaurants and other catering

In the 60-industry input-output tables, catering is included in the industry ‘hotels and restaurant services’. This industry has been subdivided into ‘hotels and other lodging places’ and ‘restaurants and other catering’, using for each input and emission coefficient the same proportions between the two sub-industries as found in the more detailed Danish NAMEA (Weidema et al. 2005a). The resulting two industries have been embedded into the overall EU-27 input-output table, using the proportional input coefficients for each of the two sub-industries as found in the Danish NAMEA.

Since the input of food to ‘restaurants and other catering’ has already been included in the consumption values reported in Chapter 8.2, a special version of the catering industry has been made, named ‘restaurants and other catering, not incl. food’, in which the inputs of food products and the corresponding wholesale and retail-processes have been omitted and the overall output adjusted. In this way, a process is obtained that provides the catering service only, while keeping the input of food separate.

Out of the process ‘restaurants and other catering, not incl. food’, only a part should be ascribed to the preparation of meat and dairy products. Failing to find any physical causality, the same share of the process has been applied as the economic share of meat and dairy products in the food inputs to ‘restaurants and other catering’ from the Danish NAMEA. This share is 18 %, and with a total economic output of the service process ‘restaurants and other catering, not incl. food’ of EUR 340 billion, the 18 % share for meat and dairy products becomes EUR 61 billion.

It should be noted that some meals are prepared within industries other than ‘restaurants and other catering’, e.g. hospitals and social institutions, but the preparation of these meals have not been included in this study. This implies that while the meat and dairy products consumed for these meals are included in the study, the preparation of these meals are ascribed to the primary service provided by these industries. Since these meals represent less than 3 % of all meals, this omission is regarded as negligible.

8.4.8 Transport processes

The transport processes involved in the life cycle of meat and dairy products are generally included through the inputs from the three transport industries: 'transport by road; pipelines', 'transport by ship' and 'air transport'.

However, additional transport is performed within each industry, e.g. the food industry, by trucks and vans owned and/or operated by the industry itself. For those 15 countries (EU-15, minus Greece, plus Czech Republic) that report annual road freight transport work by type of operation, and by type of goods, to Eurostat, the transport of agricultural and food products that take place in trucks and vans owned and/or operated by the industry itself adds 29 % to the transport work performed by the industry 'transport by road; pipelines'. Since transport by road makes up approximately 80 % of all freight transport-work from the transport industry 'transport by road; pipelines', an estimate of the overall amount of transport related to meat and dairy products can be obtained by adding 23 % to the share of 'transport by road; pipelines' in the overall results.

8.4.9 Consumer transport for food purchase

Not all countries report the transport work split out on purpose. However, based on the transport statistics for the United Kingdom (DfT 2005a) and Denmark (Danmarks Statistik 2005) it is estimated that out of all private transport, the vehicle-km spent for shopping constitute 18 % of all private car-km and 11 % of all vehicle-km by public transport.

According to the transport statistics for United Kingdom (DfT 2005b), see Table 8.9, approximately half of the shopping trips are for food. These trips, however, are typically shorter than non-food shopping ones. From Table 8.9 it can be calculated that shopping trips for food constitute 37 % of the shopping trip distance by car and 24 % of the shopping trip distance by public transport.

Combining the above values results in 6.7 % of all private car-km and 2.6 % of all vehicle-km by public transport are spent for food shopping. Out of these, only a proportion should be ascribed to shopping for meat and dairy products. Considering that the frequency of food shopping is mainly determined by the short shelf-life of food products, and to a lesser extent the refrigerator space, it appears reasonable to allocate the food shopping trips exclusively to those food products that have a shelf-life below one week (bread, dairy products, meat, and vegetables). According to the Danish NAMEA (Weidema et al. 2005a), meat and dairy products constitute 42 % of the economic value of these short-lived food products. With this allocation parameter, the vehicle-km spent for shopping for meat and dairy products are finally calculated as 2.8 % of all private car-km and 1.1 % of all vehicle-km by public transport.

Table 8.9: Shopping trips by main mode (Reproduced from DfT 2005b; Table 3.8)

GB 2002/3	Trips per person per year			Average trip length (miles)		
	Food shopping	Non-food shopping	All shopping	Food shopping	Non-food shopping	All shopping
Walk	23	27	51	0.5	0.6	0.6
Car driver	40	42	82	3.9	6.5	5.2
Car passenger	18	23	42	4.5	8.6	6.8
Other private	1	1	2	1.7	3.4	2.6
Local bus	7	10	17	2.9	4.6	3.9
Other public	1	2	4	3.4	11.2	8.0
All modes	91	106	197	2.3	5.6	3.7

Table 8.10: Private car-km in EU-27 (estimate for 2000)

Country	Vehicle-km per person per day	Total annual vehicle-Tm	Source
AT	15.8	46	1)
BE	20.2	76	2)
CZ	8.4	31	2)
DE	16.1	482	1)
DK	16.6	32	6)
EE	10.3	5	2)
ES	15.0	221	1)
FI	21.1	40	2)
FR	21.1	466	1)
GB	20.7	444	2)
GR	12.9	51	1)
HU	8.3	31	1)
IE	19.8	28	3)
IT	21.3	443	1)
LT	7.8	10	7)
LU	21.3	3	1)
LV	6.6	6	2)
NL	16.0	93	4)
PT	15.5	58	1)
SE	18.3	59	2)
SI	11.2	8	2)
BG, CY, MT, PL, RO, SK	8.8	240	5)
EU-27	16.3	2875	

1. Calculated from Eurostat year 2000 passenger-km assuming 1.5 passengers per vehicle.
2. Eurostat 'Motor vehicle movements on national territory'; data for 2000.
3. Ditto; data for 2001.
4. Ditto; data for 1997.
5. Estimated as 8.8 vehicle-km per capita per day, which is the average for the group of countries (CZ, EE, HU, LV, LT, SI) that have a low per capita car use.
6. Statistics Denmark.
7. Nikolaou (2006); data for 2003.

The total distance driven in private cars in EU-27 is estimated in Table 8.10 to 2.9 vehicle-Pm annually or 16 vehicle-km per capita per day. Thus, the distance for shopping for meat and dairy products can finally be calculated to 0.46 km per capita per day or 80.4 vehicle-Tm in total for EU-27.

Private car driving is modelled with the final use process ‘car purchase and driving’ from the Danish NAMEA (Weidema et al. 2005a), which is embedded into the EU-27 NAMEA by linking to the corresponding supplying industries aggregated at the 60-industry level. The emissions per km of the Danish car fleet is validated against the Ecoinvent database process ‘operation, passenger car/RER U’ (Spielmann et al. 2004), which refers to average transport conditions in EU-15, and found to give representative results.

Vehicle-km by public transport is modelled within the process ‘transport by road; pipelines’ in the EU-27 NAMEA. As a rough estimate, 40 % of this process is regarded to be passenger transport and thus 0.5 % (equal to an output of EUR 1.9 billion) to be passenger transport for shopping of meat and dairy products.

8.4.10 Household processes

The handling of meat and dairy products in the household (in the processes ‘food storage’, ‘cooking’, and ‘dishwashing’) and the complementary purchase of ‘glass, tableware and household utensils’, have been modelled by the per capita consumption of these four household processes from the Danish NAMEA (Weidema et al. 2005a), multiplied with an adjustment factor of 0.77, based on the relative household electricity use per capita in EU-27 compared to Denmark, as the most important environmental impacts from these processes are related to electricity use. This implies an assumption that the generally lower electricity use is evenly distributed over all electricity-using equipment in the household and that this also reflects a lower consumption of the other items involved in the four household processes (purchase of household machinery, utensils, water, etc.). The Danish data do not include the use of gas for residential cooking, which is common in countries like Spain and Italy. Thus, the use of natural gas and LPG (liquefied petroleum gas) for cooking has been added, using the data for EU-15 for year 2000 in the Odyssee database (see link in next paragraph) amounting to 340 MJ natural gas and 260 MJ LPG, per capita per year. Emission data for gas stoves have been taken from the mean scenario of Jungbluth et al. (1997).

The adjusted Danish NAMEA data have been verified using data from the Odyssee and the NMC databases (http://www.odyssee-indicators.org/Databases/databases.php) for year 2000. These databases are made in cooperation between ADEME, the EIE programme of the European Commission/DGTREN and national energy efficiency agencies. The verification showed that the electricity consumption for refrigerators, freezers, and dishwashers in the adjusted NAMEA data used in this report for storage of food in households was at the same level as the Odyssee data for EU-15 countries in combination with the (rather sparse) information from the NMC database for the new Member States, when including information on the ‘equipment rate’, i.e. the percentage of households having refrigerators, freezers and dishwashers. For cooking in households, the adjusted NAMEA data used in this report is in accordance with the Odyssee database when looking at the electricity consumption alone. However, gas for cooking was missing in the Danish data, and was therefore added, as reported in the previous paragraph.

The resulting annual expenditure per capita and in total for EU-27 appears in Table 8.11.

Table 8.11: Estimated expenditures for food related household processes in EU-27.

Process	EUR per capita per year	Total in EU-27 (EUR/year)	% of process allocated to meat and dairy products	Total allocated to meat and dairy products in EU-27 (EUR/year)
Storage of food in household	44	2.12E+10	42[1]	8.91E+09
Cooking in household[2]	46	2.19E+10	28[3]	6.14E+09
Dishwashing in household	34	1.62E+10	28[3]	4.54E+09
Glass, tableware and household utensils	66	3.20E+10	28[3]	8.95E+09

1. Household expenditures for meat and dairy products relative to household expenditures for all food products that have a shelf-life below one week (bread, dairy products, meat, and vegetables) according to the Danish NAMEA (Weidema et al. 2005a).
2. Composed of EUR 38 per capita for the original NAMEA process and an additional EUR 8 for 600 MJ gas, using fuel prices of 0.009 EUR/MJ for natural gas and 0.018 EUR/MJ for LPG.
3. Household expenditures for meat and dairy products relative to household expenditures for all food products.

The modified processes from the Danish NAMEA have been embedded into the EU-27 NAMEA by linking to the corresponding supplying industries aggregated at the 60-industry level.

It could be argued that also the dwelling space occupied by kitchens and dining rooms should be allocated to the food consumption, since these rooms would not be necessary if all meals were prepared and/or eaten outside the dwellings. Both in terms of construction and space heating, this would amount to a significant environmental impact. However, kitchens and dining rooms often serve other purposes than food preparation and consumption, and are usually included in the general consumption domain of ‘housing’. For this reason, the building structure and space heating for kitchens and dining rooms have not been included in this study.

8.4.11 End-use losses

While the losses in the industries along the product chain are implicitly included when applying the EU-27 NAMEA data, data on food losses during end-use storage, meal preparation and final plate-waste have to be added separately.

Data on end-use losses are scarce. Sonesson et al. (2005) determine household waste of meat and dairy products, but the obtained values cannot be regarded as representative, because the sample size is small (the data are based on dairies kept by 35 households during a two-week period) and changes in stocks were not accounted for. For their calculations of 1995 per capita food availability, the Economic Research Service of the US Department of Agriculture used gross factors of foodservice and household losses of 30 % for dairy products and 15 % for meat products. These factors are derived from a literature search of previous studies. A more recent USDA Food Loss Project (Jones 2005), using hand-sorted refuse data and measures of food purchased and used, arrived at 1.5, 12, 13 and 19 % loss of meat products in the full-service restaurants, fast food restaurants, households and convenience stores, respectively.

The detailed Danish 1999 supply statistics (Danmarks Statistik 2003) were compared with the Danish consumption statistics, which are based on detailed consumer surveys (Groth & Fagt 1997). This showed discrepancies between supply and consumption of 23 % for meat and cheese and of 11 % for other dairy products. These data are indicative only, as they do not measure food losses directly.

Considering the lack of precise and representative data, a rough value of 20 % for both meat and dairy products is used.

8.4.12 Treatment of food and packaging waste

Input-output tables include waste treatment services at a very general level, i.e. reflecting input and emission coefficients per EUR paid for the waste treatment service, without distinction of different waste treatment technologies and the specific kinds of wastes treated. Specific waste treatment processes have therefore been included, representing three technologies (composting, landfilling and incineration) for household food wastes and the latter two technologies for the average composition of packaging for meat and dairy products. The technologies are those used in a recent study for the European Commission (Weidema et al. 2006). As carbon dioxide uptake in fodder is not included in the NAMEA data, carbon dioxide emissions from waste handling are also eliminated.

The amount of food waste is calculated from the total supply of meat and dairy products from Chapter 8.2 and the waste percentage from Chapter 8.4.11, assuming an average carcass cutting yield of 63 % and dry matter contents of 25 %, 12 % and 40 % in meat, milk, and food wastes, respectively. This gives an annual amount of post-consumer food waste of 2.5 Tg of meat and meat products and 6.9 Tg of dairy products.

The average composition of the packaging waste is derived from the 2004 background data to the Danish packaging statistics (Jakobsen 2005) and is shown in Table 8.12.

The food waste is distributed with 15 % to composting, and the remaining waste with 78.6 % to landfill and 21.4 % to incineration, which is the year 2000 proportion of these technologies in the treatment of municipal solid waste in EU-27, according to the Eurostat table ‘recovery and disposal of municipal waste’ (14 June 2006).

Table 8.12: Composition of packaging for meat and dairy products (not including fats and butter). Calculated from the 2004 background data to the Danish packaging statistics (Jakobsen 2005) and related to the 2004 Danish consumption of meat and dairy products according to Faostat (2006).

	kg per Mg meat (slaughtered weight)	kg per Mg dairy product (in raw milk-eqivalents)
Aluminium	2.7	0.2
Unspecified plastics (calculated as PP)	3.4	2.2
Paper	10.0	3.7
Corrugated board	35.0	3.7
PS	7.8	0.3
Glass	12.6	0.4
PE	10.5	1.2
Tins	29.4	2.0
Laminated cartons (calculated as paper)	0.3	14.9
PET	3.4	0.1
PP	-	1.4
Total weight of packaging	115.0	30.0

Statistics that deal specifically with recycling of packaging from meat and dairy products is scarce. ACE (2007) provide statistics for recycling of laminated cartons showing a 20 % recycling rate in year 2000, growing to 30 % in year 2004, i.e. well below the average European recycling rate for paper of 50 % in year 2000. When applying NAMEA data for the production, the average recycling rate is applied to all products, thus implying a slight underestimate of the environmental impacts from packaging for meat and dairy products. Including waste treatment of all packaging wastes, i.e. ignoring that some of this is in fact recycled, slightly counterbalances this underestimate. This coarse treatment of recycling is unlikely to affect the results in any noticeable way, since the impact from packaging is anyway negligible.

8.5 Additional data sources for modelling environmental improvement options

Additional specific data used when modelling the environmental improvement options, are reported for each improvement option in Chapter 4.

In addition to these specific data, a slightly adjusted version of the database described in Chapters 8.2 to 8.4 has been used when modelling the environmental improvement options. The adjustments have been made to account for the market constraints that are likely to be encountered when modelling improvements that involve substitutions or changes in efficiency of use of current inputs or increased utilisation of co-products and wastes. Such market constraints imply that the changes in demand resulting from implementing an improvement do not affect the average suppliers, as modelled by the above-described database, but rather affects specific suppliers with specific technologies that may be significantly different from the average technology. The adjustments have only been made to account for the most significant market constraints, based on experience from previous analyses (Weidema 2005a), implying that for all other processes, it is assumed that changes in demand affect the average suppliers. The adjustments made are:

- Changes in milk consumption have been modelled as resulting in a reduction in milk supply for dried milk and butter, rather than an increase in output from dairy farms, reflecting that total milk output from dairy farms is constrained by milk quotas. As current discussion indicates that milk quotas may be increased or eliminated because of increases in demand, the current constraint may not be valid for modelling the long term; see also the discussion in relation to Table 4.24.
- Changes in beef and pork consumption have been modelled as coming from meat cattle and intensive pig farming, respectively, reflecting that these types of suppliers are the most sensitive to changes in demand.
- For protein inputs to agriculture, yields and area requirement has been modelled by the specific data for soy crops from Dalgaard et al. (2007), reflecting that changes in demand for fodder protein are expected to affect the supply of soy protein rather than the average protein supply. In accordance with this, protein-rich fodder by-products from the meat industry are also substituted by supply from the resulting process 'vegetable and animal oils and fats, EU-27, land use corrected', reflecting that the by-products are not affected by changes in demand for fodder. Likewise, carbohydrate-rich fodder by-products from the food industry are substituted by supply from 'grain crops, EU-27'. The substitution implies also that the part of the emissions of the affected food industries that were previously allocated to the fodder by-products are now allocated to the food output.
- For production of electricity, emissions of NOx, CO_2 and CO have been increased to the level typical for natural gas combustion in electricity generation (0.81 kg CO_2, 0.21 g CO and 1.6 g NOx per kWh), reflecting that changes in electricity consumption are expected to affect fossil fuel based electricity rather than the average electricity supply.
- The electricity produced by the specific waste treatment processes described in Chapter 8.4.11 has been modelled as substituting the above described natural gas based electricity.

The consequences of these adjustments for the results of the improvement options, as opposed to the results without modifications are discussed at the end of Chapter 4.8.

8.6 Uncertainties in inventory data

While the statistical data that underlie the analysis show large variation between countries and individual producers, the models applied provide a relatively accurate estimate of the average behaviour at the level of EU-27. The most important data on flows of products and environmental stressors have been determined with greater accuracy than data with lower influence on the result. The main contributor to the overall uncertainty is the data on the share of imports to EU-27 and the data that have been used to model this import. As mentioned in Chapter 8.3, using the typical assumption that imports have the same input and emission coefficients as the corresponding European industries would have reduced the overall environmental impacts by 30 %. This can be taken as an expression of the confidence interval related to the import assumptions, since it is possible that the model used either under or over-estimates foreign emissions relative to European conditions. Since this uncertainty applies to all products, it does not affect the conclusions on the relative importance of meat and dairy products, nor the conclusions with respect to the improvement options. However, it does affect the overall level of environmental impacts, and thus the importance of the environmental impacts relative to the socioeconomic impacts.

The uncertainty on the inventory data differs widely between the environmental impact categories. Especially the toxicity categories have large data gaps in terms of substances included and large uncertainties on the substances included. Most of the other impact categories are area or fuels related and have been determined with greater accuracy. The environmental stressors with greatest influence on the aggregated results for meat and dairy products in EU-27 (see Chapter 1) are area use (51 %), carbon dioxide (11 %), particles (10 %), ammonia (9 %), methane (7 %), NOx and N_2O (5 % each), with 3 % left for all other stressors. The geatest uncertainties on the stressors are found for N_2O, particles and NOx, with estimated coefficients of variance of 1.5, 1.2 and 0.2, respectively. The uncertainty on area use and carbon dioxide is low (estimated coefficient of variation 0.02), with ammonia and methane at an intermediate position with an estimated coefficient of variation of 0.1.

8.7 Modelling autonomous developments

The data used to model autonomous developments up to year 2020 (Chapter 4.9) have been taken from the reference run of Capsim (EuroCARE 2004), and the baseline scenarios of Primes (Mantzos & Capros 2006) and CAFE (Amann et al. 2005). When data for 2020 have not been available, linear interpolation from the data of the closest years has been applied.

For meat consumption, the CAPSIM reference run forecasts an EU-average 3.6 % increase from 2001 to 2020 composed of a 2.6 % increase for pork, 14.3 % for poultry and a reduction of 6.9 % for beef and veal. The baseline scenario of the Scenar2020 study (Nowicki et al. 2007) forecasts a slightly larger increase of 4.5 % per capita from 2005 to 2020, but with less variation between meat types: 6 % for pork and 6.4 % for poultry and less for beef.

For dairy products, seen as a whole, the forecasted changes in demand are negligible in the Capsim reference run.

For non-agricultural emissions in 2020, data for CO_2 are taken from the baseline scenario of Primes and data on SO_2, NOx, VOC, NH_3 and PM2.5 from the CAFE baseline scenario. Other emissions (notably heavy metals) are assumed to follow the trend of the CO_2 emissions.

9 ENVIRONMENTAL IMPACT ASSESSMENT METHODS

Recent reviews of the state-of-the-art of life cycle impact assessment can be found in Udo de Haes et al. (2002) and Jolliet et al. (2004). The common approach is that for each impact category, a category indicator is chosen and a characterisation model is applied to convert the relevant inventory results (e.g. emissions of different substances) to a common unit, i.e. the unit of the category indicator. Among the different existing impact assessment methods, there is a reasonable similarity in the impact categories included. The differences between the methods are rather in the models applied to characterise each impact category, and in the extent to which the mid-point results (for individual impact categories) are modelled further in the impact chain towards a single end-point.

In this study the impacts are calculated at three levels: 15 midpoint impact categories, three endpoint impact categories, and a single overall impact value. The impact categories at the different levels and the characterisation models used to calculate the impact indicators for the different categories are described in Chapter 9.1.

Chapter 9.2 treats the normalisation reference used in this study, i.e. the environmental impacts of the total final consumption in the EU. (The normalisation reference allows expressing the environmental impacts of specific products in questions, such as meat and dairy products, as a percentage of total environmental impacts.) Chapter 9.3 addresses additional aspects of weighting environmental impacts, and Chapter 9.4 assesses the uncertainties in the impact assessment.

9.1 Impact categories and characterisation models

To calculate the contribution of individual environmental exchanges (e.g. emissions of a certain substance) to each midpoint impact category, a combination of characterisation models from two of the most recent impact assessment methods has been selected, the IMPACT2002+ v. 2.1 and the EDIP2003 methods (Jolliet et al. 2003, Humbert et al. 2005, Hauschild & Potting 2005, Potting & Hauschild 2005). Both methods are second-generation methods, building on previous work (Ecoindicator1999 and EDIP1997, respectively). The main criteria for choosing a specific characterisation model is completeness in coverage, both in terms of how much of the impact chain is covered by the model, and in terms of substances included (especially relevant for toxicity). The specific combination of characterisation models chosen are described in detail in Weidema et al. (2007), which is reproduced as Annex II to this report ([1]).

([1]) Weidema et al. (2007) describes a comprehensive impact assessment method, named Stepwise2006 v. 1.2. It builds on Weidema et al. (2006), with a few minor modifications.

The following midpoint impact categories are applied in this study:

Midpoint impact category	Category indicator	Source of characterisation model
Acidification	m^2 unprotected ecosystem (i.e. the ecosystem area that is brought to exceed the critical load for acidification)	EDIP2003
Ecotoxicity, aquatic	kg-equivalents triethylene glycol into water	IMPACT2002
Ecotoxicity, terrestrial	kg-equivalents triethylene glycol into soil	IMPACT2002
Eutrophication, aquatic	kg nitrate equivalents	new; specifically developed for this study
Eutrophication, terrestrial	m^2 unprotected ecosystem (i.e. the ecosystem area that is brought to exceed the critical load for terrestrial eutrophication)	EDIP2003
Global warming	kg CO_2 equivalents (100 years time horizon)	IPPC 2001 (also used in EDIP2003)
Human toxicity	kg-equivalents of chloroethylene emitted into air	IMPACT2002
Mineral extraction	MJ additional energy (the difference between the current energy requirement for extraction and an estimated future energy requirement for extraction from lower grade ores)	IMPACT2002
Nature occupation	m^2-equivalents arable land (representing the impact on biodiversity from the occupation of one m^2 of arable land during one year)	modified from IMPACT2002
Non-renewable energy	MJ total primary non-renewable energy	IMPACT2002
Ozone layer depletion	kg-equivalents of CFC-11 into air	IMPACT2002 (taken from the US EPA ozone depletion potential list)
Photochemical ozone impacts on vegetation	m^2*ppm*hours (i.e. the product of the area of vegetation exposed to the 40 ppb threshold of chronic effects, the annual duration of exposure over the threshold, and the accumulated hourly mean ozone concentration over the threshold during daylight hours in the vegetation period)	EDIP2003
Respiratory inorganics	kg-equivalents of $PM_{2.5}$ into air (i.e. particulate matter < 2.5 μm)	IMPACT2002
Respiratory organics (photochemical ozone impacts on human health)	person*ppm*hours (i.e. the product of the number of people exposed above the 60 ppb threshold, the annual duration of the exposure above the threshold, and the accumulated hourly mean ozone concentration over the threshold)	EDIP2003

For aquatic eutrophication, a damage model has until now been missing. Specifically for this project, Dr Michael Hauschild of the Technical University of Denmark reviewed the currently available evidence and suggested new damage factors. The findings of Dr Hauschild are presented in Annex III.

For emissions of metals, such as Cu and Zn, it should be noted that current characterisation models do not take into account all relevant metal-specific properties and processes, and the impact potential of metal emissions may therefore be overestimated (Heijungs et al. 2004). This proviso is particularly relevant for the emissions of Cu to soil in this study and its dominating role in the impact category 'aquatic ecotoxicity'.

In relation to the application for agricultural products, the following issues should be noted:

- *Pesticides:* Pesticides contribute to both ecotoxicity and human toxicity. The human toxicity impact included is only that caused by the diffuse emission of pesticides in the environment and not that caused by transmission through treated food. The damage caused by pesticide transmitted through treated food is in general much larger than the diffuse emissions, but at present, this is not quantified (Humbert et al. 2005).
- *Phosphate mineral extraction:* The impact category 'mineral extraction', which is taken from IMPACT2002+ (Humbert et al. 2005) mainly based on Ecoindicator99 (Goedkoop & Spriensma 2001), only have characterisation factors for metals. This implies that the additional future energy requirement for phosphate is not included, although it can be expected that future production of phosphate will have to be obtained from less concentrated raw materials than today, or alternatively that more effective technologies must be found to mobilise the current reservoirs of phosphate available in agricultural soils.

In the first test run of the impact assessment, it was noticed that the assessment results for human toxicity were dominated by arsenic emissions to soil, mainly stemming from mining and first processing of metals. While recognising that these emissions may indeed be important, it was noted that the human toxicity characterisation factors for emissions to soil in the IMPACT2002 impact assessment method are intended for a uniform emission to soil, where there is a uniform possibility for humans to come into contact with the emission, for example through working with agricultural soils. As the emissions to soil from mining and metal processing do not fulfil this condition of uniform distribution, it has been decided to eliminate these emissions from the database applied for this project. This decision also affects the normalisation reference for aquatic ecotoxicity.

Note that the 15 midpoint impact categories included do not cover all biophysical impacts. Notably, the following issues are covered under social impacts in Chapters 5 and 10:

- Injuries
- Dietary health
- Food contamination
- Well-being of animals in human care
- Maintenance of agricultural landscape heritage (the manufactured rural landscape)

and the following biophysical impacts are altogether excluded from the assessment:

- Disruption of archaeological heritage
- Antibiotic resistance
- Species dispersal
- Noise.

To aggregate the different categories of environmental impacts further, the midpoint results are linked to a single value expressed in monetary units (euro) via clearly described impact pathways and indicators (see Annex II and IV). The first part of the impact pathways is modelled in analogy with the procedures of the IMPACT2002+ method, arriving at separate scores for the three end-point damage categories human health (measured in Quality Adjusted Life Years — QALY), ecosystem quality (measured in species-weighted area-time), and productivity (measured in monetary units). To further model QALYs and species-weighted area-time in monetary units two conversion factors are applied:

- A conversion factor of 74 000 EUR/QALY, based on the overall budget constraint (see Annexes II and IV for details). The derived monetary value of a QALY has a low uncertainty range (62 000-84 000 EUR/QALY) and is of the same size as the undiscounted value of a life year of 74 627 EUR recommended in the recent update of the ExternE methodology (Markandya et al. 2004), based on willingness-to-pay studies.
- A conversion factor of 0.14 EUR/species-weighted m^2*year, reflecting that in developed countries 1-2 % of the GDP is reserved for environmental protection expenditures, the upper value of which is taken as a proxy for the marginal willingness to pay for preservation of ecosystems at the current level of damage (approximately 50 % of the global land area or 1.3 E+14 species-weighted m^2*years left for protection). The value has been validated against Japanese estimates (Itsubo et al. 2003, 2004) of the willingness to pay to avoid species-extinctions, which is the only choice modelling study known to the authors that seeks to compare impacts on human well-being and nature in generic terms. Expressed as a relative weighting between protecting the global population (6.2E+9 people = 6.2 E+9 QALY) and the global terrestrial ecosystem area (1.3 E+14 m^2), the conversion factor can be seen as a weight of 25:1 to human well-being vs. ecosystem. In Weidema et al. (2006) the weighting 10:1 was proposed, based on the 10 % protection target of the UN Convention on Biological Diversity, which are now rather seen as an upper value (corresponding to an assessment of the current damage to 5 % of GDP). The value of 0.14 EUR/species-weighted m^2*year (= 1 400 EUR/species-weighted ha*year) is still substantially larger than previous estimates, such as the ExternE range of 63-350 EUR/ha of ecosystem protected from acidification and eutrophication (Markandya et al. 2004), derived from revealed preferences from political negotiations. The ExternE values may be seen as a low-end estimate, and together with the value based on the 10 % protection target, this gives us an uncertainty range of 0.035-0.35 EUR/species-weighted m^2*year around the central estimate of 0.14 EUR/species-weighted m^2*year.

The characterisation factors per impact category are provided in Table 9.1, both for each damage category and for the overall impact. Uncertainties on the characterisation factors are reported in Chapter 9.4.

Table 9.1: Summary of the damage (endpoint) characterisation factors, and aggregation of all impacts into a single-score indicator measured in EUR.

Impact category	Unit of characterised values at midpoint	Impact on ecosystems		Impacts on human well-being		Impacts on resource productivity	All impacts aggregated
		Species-weighted m^2*years / characterised unit at midpoint [1]	EUR / characterised unit at midpoint [2]	QALY / characterised unit at midpoint [3]	EUR / characterised unit at midpoint [4]	EUR / characterised unit at midpoint [5]	EUR / characterised unit at midpoint [6]
Acidification	m^2 UES	5.5E-02	7.7E-03				7.7E-03
Ecotoxicity, aquatic	kg-eq. TEG wat.	5.0E-05	7.1E-06				7.1E-06
Ecotoxicity, terrest.	kg-eq. TEG soil	7.9E-03	1.1E-03				1.1E-03
Eutrophication, aq.	kg NO_3-eq.	0.72	0.10				0.10
Eutrophication, terr.	m^2 UES	8.9E-02	1.3E-02				1.3E-2
Global warming	kg CO_2-eq.	0.58	8.2E-2	2.1E-08	1.6E-03	-3.7E-04	8.3E-2
Human toxicity	kg C_2H_3Cl-eq.			2.8E-06	0.21	6.4E-02	0.27
Mineral extraction	MJ extra					4.0E-03	4.0E-03
Nature occupation	m^2 arable land	0.88	0.12				0.12
Ozone layer deplet.	kg CFC-11-eq.			1.1E-03	78	24	100
Ph.chem. ozone – veg	m^2*ppm*hours	6.6E-04	9.3E-05			2.8E-04	3.7E-04
Respirat. inorganics	kg PM2.5-eq.			7.0E-04	52	16	68
Respiratory organics	pers*ppm*hours			2.6E-06	0.20	6.1E-02	0.26

[2] Values from column [1] multiplied by 0.14 EUR / species-weighted m^2*year.
[4] Values from column [3] multiplied by 74000 EUR / QALY.
[6] Sum of values from column [2], [4] and [5].

9.2 Normalisation reference

Both the 'Stepwise' method as described in Weidema et al. (2006) and the EIPRO study rely on the 1995 normalisation values for Western Europe by RIZA-CML (Huijbregts et al. 2001, van Oers et al. 2001).

The construction of a NAMEA for EU-27 allows us to apply a more recent and more relevant normalisation reference, namely the emissions related to the total final consumption in EU-27. Table 9.2 shows EU-27 emission references compared to important environmental exchanges (stressors) in the EIPRO normalisation reference (Tukker et al. 2006) and Table 9.3 shows the normalisation data for EU-27 at the level of impact categories compared to the RIZA-CML-based normalisation reference.

The difference between the year 2000 normalisation reference for EU-27 and the RIZA-CML based normalisation references have several causes:

- When using consumption as normalisation reference, the emissions outside the EU are also included, while the RIZA-CML normalisation data refer to emissions on the European territory. This is particularly relevant for emissions related to primary production (agriculture and mining). Agricultural imports to the EU account for a rather large land use, and consequent nature occupation, outside the EU. Mining takes place mainly outside Europe, which led to the complete exclusion of mineral extraction from the EIPRO dataset, and explains the larger values for non-renewable energy carriers. It is also the emissions from metal mining and processing that cause the larger normalisation reference for terrestrial ecotoxicity.

- The difference in years can explain lower emissions, especially of ozone depleting substances, lead and sulphur dioxide. The latter also partly explains the smaller normalisation reference for acidification.
- Extrapolation has been applied to different emissions in the RIZA-CML normalisation data, as compared to the EU-27 dataset. The more recent estimates in national data may be more precise.

Table 9.2: Some important environmental exchanges (stressors) related to the total final consumption in EU-27 compared to the same emissions in the EIPRO dataset.

Environmental exchange	Unit	EU-27 Consumption 2000	EU-25 Production 1995 (EIPRO)	Ratio EU-27/EIPRO
Extraction of energy carriers:				
Coal, hard, in ground	kg	5.69E+11	6.26E+10	9.10
Gas, natural, in ground	m3	1.51E+12	3.03E+11	4.98
Oil, crude, in ground	kg	1.58E+12	3.36E+11	4.69
Air emissions:				
Ammonia	kg	4.36E+09	3.68E+09	1.18
Arsenic	kg	2.46E+05	2.00E+05	1.23
Carbon dioxide	kg	3.56E+12	3.52E+12	1.01
Carbon monoxide	kg	3.23E+10	4.45E+10	0.73
Dinitrogen monoxide	kg	1.36E+09	1.35E+09	1.01
Lead	kg	2.39E+06	1.30E+07	0.18
Methane	kg	3.02E+10	2.11E+10	1.43
Nitrogen dioxide	kg	1.16E+10	1.47E+10	0.79
NMVOC	kg	6.19E+09	3.22E+10	0.19
Particulates, < 10 um	kg	3.46E+09	1.40E+09	2.47
Sulphur dioxide	kg	7.31E+09	2.49E+10	0.29
Water emissions:				
Phosphorus	kg	3.56E+08	2.37E+09	0.15
Soil emissions:				
Arsenic	kg	2.98E+03	2.63E+04	0.11
Copper	kg	1.34E+07	2.16E+08	0.06

Table 9.3: The normalisation reference for EU-27 at the level of midpoint and endpoint (damage) impact categories compared to the RIZA-CML-based normalisation reference for Europe.

Impact category	Unit	EU-27 Per EUR household consumption 2000	EU-27 Total final consumption 2000 (this study)	Europe Production 1995 (RIZA-CML)	Ratio EU-27/ RIZA-CML
Midpoint categories:					
Acidification	m^2 UES	6.61E-02	3.81E+11	1.06E+12	0.36
Ecotoxicity, aquatic	kg-eq. TEG w.	53.74	3.08E+14	6.55E+14	0.47
Ecotoxicity, terrestrial	kg-eq. TEG soil	1.53	9.27E+12	1.14E+12	8.16
Eutrophication, aquatic	kg NO_3-eq.	5.25E-03	3.01E+10	3.65E+10	0.83
Eutrophication, terrestrial	m^2 UES	0.18	9.94E+11	1.01E+12	0.98
Global warming	kg CO_2-eq.	0.81	4.72E+12	5.13E+12	0.92
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	3.00E-03	1.73E+10	2.19E+10	0.79
Human toxicity, non-carc.	kg C_2H_3Cl-eq.	2.65E-03	1.70E+10	8.36E+10	0.20
Mineral extraction	MJ extra	1.48E-02	9.12E+10	1.41E+11	0.65
Nature occupation	m^2 arable land	0.50	2.73E+12	1.41E+12	1.94
Non-renewable energy	MJ primary	23.76	1.40E+14	7.33E+13	1.91
Ozone layer depletion	kg CFC-11-eq.	4.81E-07	3.00E+06	9.83E+07	0.03
Photochemical ozone, veg.	m^2*ppm*hours	9.33	5.37E+13	6.76E+13	0.79
Respiratory inorganics	kg PM2.5-eq.	8.41E-04	4.80E+09	4.23E+09	1.14
Respiratory organics	person*ppm*hours	9.80E-04	5.63E+09	4.83E+09	1.17
Endpoint (damage) categories:					
Impact on ecosystems	Species-weighted m^2*years	0.953	5.40E+12	4.47E+12	1.21
Impacts on human well-being	QALY	6.25E-07	3.58E+06	3.48E+06	1.03
Impacts on resource productivity	EUR	0.016	9.36E+10	9.52E+10	0.98

9.3 Weighting

The Stepwise2006 endpoint impact assessment method (see Annex II and IV) avoids weighting in its strict sense, by allowing extension of the characterisation beyond the traditional midpoint impact categories, arriving at one single impact category, measured either in Quality Adjusted Life Years (QALY) or in monetary units. All environmental impacts are subsumed under this category, using appropriate characterisation factors, including the necessary value choices in the characterisation models.

In addition to the 'Stepwise' endpoint impact assessment method, an alternative weighting method is applied. The average weights of the Ecoindicator 99 (Goedkoop & Spriensma 2001) have been applied for this purpose, due to their widespread use. This weighting (40 % to impacts on ecosystems, 40 % to impacts on human well-being, 20 % to impacts on resources) is to be applied on the normalised endpoint results. The 20 % weighting on resources have been interpreted to be applicable for all impacts on resource productivity.

9.4 Uncertainties in impact assessment

The uncertainties in the characterisation models have been derived by combining the uncertainty information from Hauschild & Potting (2005), Goedkoop & Spriensma (2001), Chapter 9.1 and Annex II. The resulting coefficients of variance are reported in Table 9.4. The coefficient of variation on the combined results have been calculated as the square root of the sum of the squared components, assuming all component distributions to be lognormal and independent. This assumption may slightly overestimate the uncertainty.

Table 9.4: Uncertainties of the damage (endpoint) characterisation factors.

Impact category	Coefficient of variation of characterised data at midpoint	Coefficient of variation of characterised data at endpoint (species-weighted m^2*years, QALYs)	Coefficient of variation of monetarised values
Acidification	0.9	1.6	2.4
Ecotoxicity	1.8	3.2	3.6
Eutrophication, aq.	0.2	1.4	2.3
Eutrophication, terr.	1.2	1.8	2.6
Global warming	0.02	[a]	1.8 [a]
Human toxicity	1.8	3.2	3.3
Mineral extraction	1.6	1.9	1.9
Nature occupation	1.1	1.1	2.1
Ozone layer deplet.	1.4	2.0	2.3
Ph.chem. ozone – veg	1.6	2.4	3.0
Respirat. inorganics [b]	1.8	3.2	3.3
Respiratory organics	2.2	2.6	2.9

[a] Since the applied endpoint characterisation factor for global warming is a low estimate, a coefficient of variation is not an appropriate way to describe the uncertainty. The value of 1.8 on the monetarised value expresses the uncertainty on the monetarisation alone.
[b] As for human toxicity

Using other impact assessment methods, especially for the endpoint modelling (the aggregation of impacts on humans, nature and resources) can have large importance for individual impact categories. This is shown in Table 3.3, comparing to the results with the Ecoindicator99 weights, but the comparison also shows that the overall result is only little affected, since the importance is simply shifted between the impact categories.

10 SOCIOECONOMIC IMPACT ASSESSMENT METHODOLOGY

This chapter introduces first the different parameters considered in the socioeconomic impact assessment of the improvement options and gives and overview of the general approach to impact assessment (Chapter 10.1). Then it describes each of the socioeconomic impact indicators in more detail, explains the extent to which they were quantified, the data sources and the models used, and any limitations encountered (Chapters 10.2-10.11).

10.1 Overview

The socioeconomic impact assessment considers the following indicators ([1]):

- Direct production costs
- Injuries
- Dietary health (human health related to diet and nutrition)
- Food contamination (food safety)
- Supply security
- Well-being of animals in human care
- Landscape maintenance
- Employment
- Household work (time usage)
- Income distribution (between different regional, social and economic groups)

Initially, a qualitative screening is performed, identifying which of the socioeconomic indicators are likely to be affected by each improvement option, resulting in a matrix of improvement options and socioeconomic indicators with scores of none-low-medium-high relevance (Chapter 5.1).

For combinations of indicators and improvement options assigned high or medium relevance, the indicator values are quantified for the relevant parts of the product systems, using available statistical and technical data sources. The quantifications use the system model and boundaries described in Chapters 2 and 8, just as for the environmental quantifications, and the socioeconomic impact assessment methods are analogous to the environmental impact assessment methods described in Chapter 9. The following sections describe in more detail the different indicators, the assumptions made and the models used, in particular the characterisation models linking the socioeconomic indicators to midpoint and endpoint impact categories.

10.2 Direct production costs

Production costs can be measured as the value added for each process in the analysed system, added up over the entire product life cycle.

([1]) These socioeconomic indicators are not sharply delimited from the environmental indicators outlined in Chapter 9 and the rebound effects outlined in Chapter 4.7.1. Formally, some aspects of supply security and landscape maintenance can be regarded as rebound effects, as noted in Chapter 4.7.1, while injuries, dietary health, food contamination, and well-being of animals in human care, can all be regarded as biophysical environmental indicators. However, for practical reasons and for the purpose of this report, all the above indicators are denoted as socioeconomic.

Most required data come directly from the EU-27 input-output table that was also used for the environmental assessment. The measurement unit is EUR_{2000}. Additional data for specific processes are given in the footnotes to Table 10.1, as far as possible collected from literature.

Due to the straightforward availability of data, the indicator result has been calculated for all improvement options, disregarding their relative importance (see Table 10.1). Thus, the indicator 'production costs' has not been included in the qualitative screening in Chapter 5.1.

No further processing of the data is necessary, since the data are already expressed in the unit used for the aggregated environmental impact.

Table 10.1: Changes in annual direct production costs for the different improvement options. Only that part of the improvement which can be ascribed to the consumption in EU-27 has been included (in parallel to the data in Table 4.20). Negative values signify an improvement (a cost reduction).

	Direct cost of improvement option [MEUR/year]	Cost per capita [EUR/year]	In % of total cost for meat and dairy products
1. Catch crops	18	0.04	0.004
2. Cereal intensification	-180	-0.37	-0.04
3. Optimised protein feeding	1 050	2.17	0.24
4. Liquid manure pH reduction	750	1.55	0.17
5. Tightening of manure regulation	-480	-1.00	-0.11
6. Copper reduction in animal diets	160	0.33	0.04
7. Methane-reducing animal diets	~ 0	~ 0.00	~ 0.00
8. Liquid manure biogasification	1 150	2.37	0.26
9. Home delivery of groceries	14 000	28	3.1
10. Cold appliances regulation	-144	-0.30	-0.03
11. Power saving in industry	-650	-1.34	-0.15
12. Household meal planning	-6 800	-14	-1.6

1. Seeds, sowing, etc., minus saved artificial fertiliser.
2. Saved fuel costs. Saved land rent not included since this is a transfer cost.
3. Changes in feed costs plus increased costs for artificial fertiliser.
4. 1.24 EUR/Mg manure, 674 Tg manure, minus saved artificial fertiliser.
5. Saved artificial fertiliser; 950 Gg N at 0.509 EUR/kg.
6. 0.24 EUR additional costs per 100 kg feed in pig farming.
7. Restricts the flexibility in feed optimisation slightly, but is not expected to affect costs significantly.
8. Income from electricity production minus capital costs (see Chapter 4.3.4) and minus operation costs set at 4.4 EUR/Mg manure (based on Walla & Schneeberger 2003).
9. Saved costs for private transport, plus costs for delivery service set at 3.99 EUR/delivery (calculated from costs of vehicle and fuel 0.07 EUR, wages 3.17 EUR, and 0.75 EUR operating surplus).
10. Saved electricity costs minus additional costs of new appliances.
11. Saved costs of electricity. No consulting or investment costs have been included.
12. Saved costs for food purchase and waste treatment. Not including consulting or investment costs.

Cost data generally vary over time and are typically more uncertain than data on physical flows. However, it is only for a few of the improvement options that direct production costs play a major role for the overall assessment, as can be seen in Table 5.2. When assessing the overall net benefit of the improvement options, an uncertainty of +/- 40 % is applied on the cost data. The improvement option with the largest costs is 'home delivery of groceries' (improvement option 9) and the uncertainties on the costs play an important role for the overall uncertainty of this improvement option. However, the improvement option that is most sensitive to changes in costs is 'liquid manure biogasification' (improvement option 8); see also the discussion on discounting in Chapter 5.4.

10.3 Injuries

Injuries at work or from road accidents are recorded in the NAMEA database applied for this study. The data includes fatal and non-fatal injuries from road traffic and work (occupational injuries). The category indicator is 'fatal-injury-equivalents'.

For assessing the impacts of injuries further, the characterisation factors are derived from the overall proportion of YLL (years of life lost) to YLD (years-of-life-equivalents lost due to disability) for these causes in the Global Burden of Disease study (Mathers et al. 2004, using the values without discounting and age-weighting), compared to the proportion of reported cases from Eurostat and the CARE Road Accident Database (http://europa.eu.int/comm/transport/care/). The midpoint characterisation factors calculated from these relationships are 133 non-fatal road injuries / fatal injury (death), 1 300 non-fatal work injuries/fatal injury. The damage factor is 43 QALY/ fatal injury.

For the total annual consumption of meat and dairy products in EU-27 (the functional unit of the study) the impact is 2 040 fatal injuries-equivalents or 3.4 % of the total amount of road or work related injuries caused by EU-27 total final consumption. Using the damage factor of 43 QALY/fatal injury, the total impact is 87 500 QALYs or EUR 6.5 billion.

Shopping by car contributes 46 % of all work or road related injuries in the life cycle of meat and dairy products, while truck transport is responsible for 18 % of the impact.

Due to the straightforward availability of data, the indicator result has been calculated for all improvement options, disregarding their relative importance (see Table 10.2). Thus, the indicator 'injuries' has not been included in the qualitative screening in Chapter 5.1.

Table 10.2: **Changes in injuries potential for the different improvement options. Only that part of the improvement which can be ascribed to the consumption in EU-27 has been included (in parallel to the data in Table 4.20). Negative values signify an improvement (a reduction in injuries potential).**

	Injuries [QALY/year]	Aggregated impact per capita [EUR/year]	In % of total injuries for meat and dairy products
1. Catch crops	-	-	-
2. Cereal intensification	-22	0.00	-0.02
3. Optimised protein feeding	62	0.01	0.07
4. Liquid manure pH reduction	-	-	-
5. Tightening of manure regulation	-35	-0.01	-0.04
6. Copper reduction in animal diets	-	-	-
7. Methane-reducing animal diets	-	-	-
8. Liquid manure gasification	-16	0.00	-0.02
9. Home delivery of groceries	-7 500	-1.50	-8.5
10. Cold appliances regulation	-	-	-
11. Power saving in industry	-45	-0.01	-0.05
12. Household meal planning	-770	-0.15	-0.88

10.4 Dietary health (human health related to diet and nutrition)

Dietary health is measured in terms of dietary related diseases, quantifiable in the same way as other impacts on human health, with the unit Quality Adjusted Life Year (QALY).

This impact indicator is quantified only for one of the improvement options, namely meal planning tools. (For others the impact is found of low or no relevance.) The uncertainties involved in the quantification are particularly high. Therefore it was decided that the calculated value should not be added to the other indicator values for the overall impact assessment.

10.5 Food contamination (food safety)

As indicated in Table 5.1, there is none of the suggested improvement options that have significant impacts on food contamination, and therefore no attempts have been made at quantifying these impacts.

10.6 Supply security

Supply security is the ability to control the temporal and geographical availability of goods or services in demand. Supply security is desirable to avoid unnecessary periods of undersupply, which increase costs and/or reduce productivity (in industry) or needs satisfaction (in households).

Impacts on supply security are mainly related to processes that have temporal or geographical variation in output and products that have a limited ability for storage. Furthermore, deficiencies in planning may lead to shortages in any type of product.

Supply security is normally increased by increasing storage capacity, when this is possible, or by maintaining capacity of alternative production routes in case of supply failure. The costs of maintaining excess storage or production capacity in industry is normally factored into the normal production costs and is therefore covered by the indicator 'direct production costs' (Chapter 10.2). The benefits of supply security are equal to the avoided costs of undersupply or – in the case of households – the avoided reduction in needs satisfaction. In both cases, this may be measured as the willingness to pay for the missing product during situations of undersupply, multiplied by the expected extent of undersupply.

However, it is not trivial to assess the consumer willingness to pay for the missing products during situations of undersupply. The premium price paid in convenience stores, as well as the value of the additional time spent for shopping for out-of-stock items may be used as indications of the willingness to pay, but quantification also requires data on how often an out-of-stock situation actually leads to corrective behaviour.

As indicated in Table 5.1, changes in supply security are identified mainly for the households, and to a lesser extent for cereals.

Because of the difficulties described above, it was decided not to quantify the supply security impacts in this study. The issue is deemed worthwhile enough to be included in further studies on household planning and shopping behaviour, since the implied welfare loss may indeed be significant.

10.7 Well-being of animals in human care

Out of the different improvement options in Table 5.1, the reduction of ammonia emissions and the dietary adjustments may influence animal well-being. The significance of these impacts is not found adequate to warrant a quantification. As the suggested improvement options already show a net benefit, the addition of a value for well-being of animals would only further increase this benefit.

10.8 Landscape maintenance

In contrast to the environmental indicator 'nature occupation', which addresses natural biodiversity, the indicator for landscape maintenance addresses the cultural landscape and its aesthetic and heritage values.

In the qualitative screening reported in Chapter 5.1, the parameter 'landscape maintenance' has been used to report such disparate impacts as:

- The impact of 'greening' the winter landscape by catch crops, which may be seen as an aesthetic improvement.
- The reduction in odour by changes in animal diets and manure treatment. This impact should possibly have been covered by a separate indicator.
- The reduction in the quantity of cultural heritage if arable land is abandoned as a result of intensification or reduction in food demand.

Aesthetic value of landscapes is typically measured by willingness-to-pay studies or by the travel cost method, i.e. calculating the money that visitors are willing to spend to see them. No studies have been found that address issues such as those listed above and it is out of the scope of this study to carry out such quantification.

However, in the case of ‘greening’ winter crops and reduction of odour, the suggested improvement options show a net benefit, and the addition of a value for landscape maintenance would only further increase this benefit.

10.9 Employment

Changes in employment rates are generally not related to specific production activities, but rather to the general level of economic activity. The general effect of introducing specific technologies that are either more labour-extensive or more labour-intensive is therefore simply to shift employment to or from other productive activities, through the labour market.

Two exceptions to this general situation may be found:

- When the new technologies or processes are specifically affecting or addressing low-skilled persons that have difficulty in finding other jobs. In such cases, overall employment rates can be affected directly. An example could be home delivery of groceries, which may be perceived as a low-skill job.
- When the new technologies or processes are specifically affecting employment in geographical areas where unemployment is large. For example, the area released because of cereal intensification in Europe may be used to produce other crops, displacing imports from less developed countries, where unemployment may be larger.

As indicated in Table 5.1, none of these situations are assessed as having a significant impact on employment, and therefore no attempts have been made at quantifying these impacts.

10.10 Household work (time usage)

The time spent for different household tasks are reported by national time surveys, which are unfortunately not conducted on a regular basis. In Europe, the only country that has established a household satellite account is Finland (Varjonen & Aalto 2006). Time for shopping is rather low in Finland, compared to other countries (Chadeau 1992), so the mean of the studies reported by Chadeau (1992) was used, which is 15 % of household work, or approximately three hours/week per adult or 2.5 hours/week per capita. Using the same allocation to food and meat and dairy products as in Chapter 8.4.8 (half of shopping is for food, 42 % of this is for meat and dairy products), an annual average time usage of 27 hours per capita for shopping for meat and dairy products is obtained.

To assess the importance of household work, the hours can be monetarised. Typically, the value of household work is set to its market cost, i.e. the cost of hiring someone to perform it. For the present study, this rate is set to 9.5 EUR per hour, which is 50 % of the average salary in EU-27 in year 2000.

Household work has been quantified for the home delivery improvement option.

10.11 Income distribution (between different regional, social and economic groups)

The value of an additional Euro of income is not the same for persons with high and low incomes. It is therefore of interest to assess whether the costs and benefits of an improvement option is distributed evenly over all persons, or if there are particular regional, social, gender or economic groups that are particularly affected.

The category indicator may most appropriately be named 'income redistribution', as its purpose is to indicate the transfer of money between different groups. Ideally, the indicator should capture disproportional distributions over societal groups of the costs reported by the indicator 'direct production costs' as well as disproportional distributions of the income generated from the costed activities.

To assess income re-distribution, it is necessary to identify the groups particularly affected. These groups may be identified in terms of their geographical, social, gender or economic position. The gender differentiation may be relevant in relation to changes in household work, but has not been quantified in this report, which only differentiates over countries and income groups. Data on the average wages for different industries and job types are obtained from national wage statistics and converted to purchase power standards.

The same principle may be applied to the results of impact indicators other than 'direct production costs', to capture the distribution of these impacts over the different societal groups. Thus, the aggregated result from the different environmental and socioeconomic indicators may be sub-divided according to the societal groups affected. Often, environmental impacts affect low-income groups more than high-income groups.

Income redistribution is not a single indicator to be added to the other environmental and socioeconomic indicators, but rather a sub-division of the impacts recorded by the other indicators. The result, specified per income group, may be re-aggregated to a single monetary value, if specific distributional weights are assigned to different income groups. In principle, the distributional weights should be assigned according to the marginal utility function of each income group. A more straightforward approach is to use linear proportional weights, where the value of a EUR is weighted according to the average income divided by the income of each income group. This proportional weighting is likely to underestimate the importance of income differences, but it still gives a less biased result than not applying any distributional weighting on the aggregated result.

The situations where a significant redistribution among income groups can be expected are parallel to those mentioned under employment (Chapter 10.9):

- When new technologies or processes are specifically affecting or addressing persons with low wages. An example is home delivery of groceries, which involves a transfer of income from the average household (and their expenditure on car and fuel purchase) to service workers.

- When new technologies or processes are specifically affecting employment in geographical areas where wages are low. Cereal intensification in Europe may increase income of European farmers at the expense of owners of European agricultural land and/or producers in less developed countries, depending on whether the released land will be abandoned or used for other crops.

For the methodological provisos made above, it was decided that the calculated values should not be added to the other indicator values for the overall impact assessment.

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ANNEXES

12 ANNEX I. PROCESSES INCLUDED IN THE PROJECT DATABASE

Process (in EU-27)	**Unit**	**Comment**
Meat cattle, extensive	EUR2000	Disaggregated from 'products of agriculture, hunting and related services' using farm model
Meat cattle, intensive	EUR2000	ditto
Calves, 16 months	EUR2000	ditto
Calves, 24 months	EUR2000	ditto
Dairy farming, central	EUR2000	ditto
Dairy farming, west,	EUR2000	ditto
Dairy farming, UK type	EUR2000	ditto
Dairy farming, lowland	EUR2000	ditto
Dairy farming, south	EUR2000	ditto
Poultry farms	EUR2000	ditto
Pigs, intensive	EUR2000	ditto
Pigs, low efficiency	EUR2000	ditto
Pigs, high ammonia	EUR2000	ditto
Grain crops	EUR2000	ditto
Agricultural products n.e.c.	EUR2000	ditto
Forest products	EUR2000	
Fish	EUR2000	
Coal, lignite, peat	EUR2000	
Crude petroleum and natural gas	EUR2000	
Metal ores	EUR2000	
Mining and quarrying products n.e.c.	EUR2000	
Pork and pork products	EUR2000	Disaggregated from 'food products and beverages' using coefficients from Danish NAMEA
Beef and beef products	EUR2000	ditto
Chicken meat products	EUR2000	ditto
Fish products	EUR2000	ditto
Processed fruits and vegetables	EUR2000	ditto
Vegetable and animal oils and fats	EUR2000	ditto
Dairy products	EUR2000	Disaggregated from 'food products and beverages' using coefficients from Danish NAMEA
Dog and cat food	EUR2000	ditto
Animal feeds	EUR2000	ditto
Cocoa products	EUR2000	ditto
Candy and other confectionery products	EUR2000	ditto
Flavouring extracts and syrups	EUR2000	ditto
Roasted coffee	EUR2000	ditto
Food preparations, n.e.c.	EUR2000	ditto
Flour	EUR2000	ditto
Oatflakes	EUR2000	ditto
Bread, cakes and biscuits	EUR2000	ditto
Bakers' shops	EUR2000	ditto
Sugar	EUR2000	ditto
Beverages	EUR2000	ditto
Tobacco products	EUR2000	
Textiles	EUR2000	
Wearing apparel and furs	EUR2000	
Leather products, footwear	EUR2000	

Wood products, except furniture	EUR2000	
Pulp, paper and paper products	EUR2000	
Printed matter and recorded media	EUR2000	
Refined petroleum products and fuels	EUR2000	
N-fertiliser	EUR2000	Disaggregated from ‘chemical products and man-made fibres’ using Ecoinvent data for emissions and primary energy for fertiliser production
Chemicals and man-made fibres n.e.c.	EUR2000	ditto
Rubber and plastic products	EUR2000	
Mineral products n.e.c.	EUR2000	
Basic metals	EUR2000	
Fabricated metal products, except machinery	EUR2000	
Machinery and equipment n.e.c.	EUR2000	
Office machinery and computers	EUR2000	
Electrical machinery n.e.c.	EUR2000	
Radio, television and communication equipment	EUR2000	
Instruments, medical, precision, optical, clocks	EUR2000	
Motor vehicles and trailers	EUR2000	
Transport equipment n.e.c.	EUR2000	
Furniture; other manufactured goods n.e.c.	EUR2000	
Secondary raw materials	EUR2000	
Electricity, gas, steam and hot water	EUR2000	
Water, fresh	EUR2000	
Construction	EUR2000	
Trade and repair of motor vehicles; service stations	EUR2000	
Wholesale trade	EUR2000	
Retail trade and repair services	EUR2000	
Hotels and other lodging places	EUR2000	Disaggregated from ‘hotels and restaurants services’ using coefficients from Danish NAMEA
Restaurants and other catering	EUR2000	ditto
Restaurants and other catering, not incl. food	EUR2000	The above process without input of food ingredients and the corresponding trade margins
Transport by road; pipelines	EUR2000	
Transport by ship	EUR2000	
Air transport	EUR2000	
Cargo handling, harbours; travel agencies	EUR2000	
Post and telecommunication	EUR2000	
Financial intermediation	EUR2000	
Insurance and pension funding	EUR2000	
Services auxiliary to financial intermediation	EUR2000	
Real estate services	EUR2000	
Renting of machinery and equipment etc.	EUR2000	
Computer and related services	EUR2000	
Research and development	EUR2000	
Business services n.e.c.	EUR2000	
Public service and security	EUR2000	
Education services	EUR2000	
Health and social work	EUR2000	
Sanitation, sewage and refuse disposal	EUR2000	
Membership organisations	EUR2000	
Recreational and cultural services	EUR2000	

Services n.e.c.	EUR2000	
Meat and dairy consumption	year	Summary process
Dairy products purchase	EUR2000	Total consumption cf. definition of reference flows
Beef and beef products purchase	EUR2000	ditto
Poultry and poultry products purchase	EUR2000	ditto
Pork and pork products purchase	EUR2000	ditto
Car purchase and driving	vehicle-km	Household consumption cf. definition of reference flows
Storage of food in household	EUR2000	ditto
Cooking in household	EUR2000	ditto
Dishwashing in household	EUR2000	ditto
Glass, tableware and household utensils	EUR2000	ditto
MSW Incineration – Average meat packaging (115 kg per Mg meat)	Mg meat	Waste treatment cf. definition of reference flows
MSW Incineration – Average dairy packaging (30 kg per Mg raw milk-eqv.)	Mg raw milk-eqv.	ditto
Landfill – Average meat packaging (115 kg per Mg meat)	Mg meat	ditto
Landfill – Average dairy packaging (30 kg per Mg raw milk-eqv.)	Mg raw milk-eqv.	ditto
MSW Incineration – Food waste	Mg	ditto
Landfill – Food waste	Mg	ditto
Composting – Food waste	Mg	ditto

13 ANNEX II. PREPARING CHARACTERISATION METHODS FOR ENDPOINT IMPACT ASSESSMENT ([1])

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13.1 Abstract

Background, Aim and Scope: Consistent endpoint impact assessment, i.e. the modelling of all midpoint impacts to a single endpoint score, requires consistent midpoint modelling. At the same time, it provides a way to ensure this consistency. This paper reports on the characterisation methods chosen or developed for the Stepwise Impact Assessment – a new impact assessment method with an optional choice between QALY and monetary units as endpoint.

Materials and Methods: To ensure overall consistency and broader coverage, several modifications have been made to the characterisation methods from EDIP2003 and IMPACT2002+. Also new impact categories, such as injuries, have been added, and the midpoint categories have been re-defined to allow for integration of social and economic impacts.

Results: The midpoint impact categories of two recent LCIA methods have been prepared for further modelling to a single endpoint, thus providing a new LCIA method. New European normalisation values are provided.

Discussion and conclusions: From our application of the new method to different case studies, we particularly note the importance of quantifying the global warming damage to ecosystems, and the more modest role of resource inputs when these are placed in the context of other impacts on resource productivity.

Recommendations and Perspectives: An outlook is provided for issues that need further elaboration.

13.2 Introduction

Most life cycle impact assessment methods (see the reviews of Udo de Haes et al. 2002, Pennington et al. 2004, and Jolliet et al. 2004) express their results at the level of midpoint or damage categories, and either refrain from further aggregation or rely on weighting factors based on value-choices, and the possible aggregation of these weighted category indicator results.

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For weighting, in the strict sense of applying numerical weighting factors based on value choices, two approaches have been attempted in previous LCIA methods:

- Weighting by a panel procedure: To derive weights from expert or lay panels, many different approaches exist, from simple questionnaires over Delphi-approaches to advanced computer-supported multi-criteria decision-making techniques. An applied example of the panel approach is the weighting between human health, ecosystem quality and resources in the Ecoindicator99 method (Goedkoop & Spriensma 2001). An inherent problem in using panel procedures is that the resulting weighting factors for the different categories often turn out to be very close to each other, so that the weighted results do not add much information beyond what is provided by the normalised results. The reason for this is the psychological mechanisms known as cognitive biases, e.g. ‘anchoring’ (Tversky & Kahneman 1974), which implies that panel participants will typically relate to all of the presented impact categories as being of some importance, and depending on the scale presented, they will seek to accommodate all categories on this scale, in not too extreme positions. This means that the results depend very much on what impact categories are included, and at what level of aggregation, and also on the scale and accompanying information (Mettier & Hofstetter 2004).
- Weighting by distance-to-target: The idea is to compare the current level of impacts for each impact category to political targets, derived from the implied or stated preferences of politicians, e.g. through an analysis of current conventions or legislation. An example of this approach is the weighting of the EDIP97 method (Stranddorf et al. 2003). A similar problem applies to the distance-to-target approach as that described for the panel approach, and is even more pronounced: The weighting factors for the different categories turn out to be very close to each other (see for example Stranddorf et al. 2003). This reflects that politicians will typically wish to do something about all problems (to avoid criticism for being inactive) but not too much about any (to avoid criticism about overspending). In consequence, the politically determined targets do not necessarily reflect the importance of the different impact categories.

Even when refraining from an explicit weighting, i.e. when presenting normalised results for each impact category separately, it is difficult to avoid an unconscious 1:1 weighting across the normalised indicator results.

Since both distance-to-target and panel approaches tend to arrive at weights very close to each other and any difference is likely to be arbitrary and unlikely to reflect the true differences in importance between the impact categories, we propose not to weight at midpoint level but to first convert midpoint results into damages.

Ideally, an endpoint impact assessment should reflect the absolute prevalence, duration and severity of the impact described by each impact category.

One way to arrive at such an assessment is to convert the indicator results to units of a physical single score, such as the ‘ecological footprint’ (Wackernagel et al. 1999) that in aggregate impacts on nature and natural resources in area units. Such approaches are seldom comprehensive; for example, the ‘ecological footprint’ does not capture ecotoxic impacts or impacts on human health.

Monetarisation is another way to aggregate indicator results across impact categories. However, monetarisation requires that the impacts can be expressed in terms to which meaningful preferences can be attached, which often implies that the impacts must be modelled further towards the endpoints than is the case in the above described characterisation. An example of this approach is the EcoSense model of ExternE (Bickel & Friedrich 2005). Monetarisation methods are also often limited in scope, for example, the EcoSense model covers only the impacts from 13 pollutants. In addition to the completeness problem, monetarisation methods often have internal consistency problems due to the very different ways that the monetarised values are derived.

Both physical single-score methods and monetarisation methods, as described above, are not weighting methods in the strict sense, but rather attempts at extending the characterisation models beyond the traditional impact categories. However, the results from such modelling may of course be applied as weighting sets.

Weidema (2007, reproduced as Annex IV in this report) presents a procedure for endpoint impact assessment that seeks to improve the completeness and consistency of physical single score modelling and monetarisation. This procedure takes its starting point in the physical indicator results for the three safeguard subjects humans, ecosystems, and resources, as provided by the LCIA method 'EcoIndicator99' (Goedkoop & Spriensma 2001). To prepare for their aggregation into an encompassing physical single-score, the three physical indicators are slightly re-defined:

- For ecosystems, the category indicator is defined as 'Biodiversity Adjusted Hectare Year (BAHY)'. This measurement unit is identical to the PDF*m^2*years used by Goedkoop & Spriensma (2001), where PDF is an abbreviation of Potentially Disappeared Fraction of species, except for the more convenient size of the unit (1 hectare = 10 000 m^2), a reversal of signs (BAHY measures a positive state, while PDF*m^2*years measure damage, i.e. 1 BAHY = -10 000 PDF*m^2*years), and that the damage is specified relative to the number of endemic species under natural conditions. Damage to ecosystems is thus measured as a loss of BAHYs.
- For human well-being, the category indicator is defined as 'Quality Adjusted Life Years (QALY)', calculated as the number of human life-years, multiplied by a quality adjustment (severity score) between 0 and 1, where 0 is equal to death and 1 is equal to perfect well-being. This measurement unit is identical to the Disability Adjusted Life Year (DALY) used by Goedkoop & Spriensma (2001), except for a reversal of signs (QALY measures a positive state, while DALY measures damage, i.e. 1 QALY = -1 DALY) and that while the disability adjustment is limited to health issues, the quality adjustment may also apply to social aspects, such as infringements on autonomy and equal opportunities (Weidema 2006). Damage to human well-being is thus measured as a loss of QALYs.
- For resource productivity, the category indicator is defined in monetary units (more specifically as 'EUR_{2003}', i.e. the currency unit euro at its average value in 2003), and calculated as the future economic output derived from application of the resource. Damage to resource productivity is thus measured as a loss of future economic output caused by the current damage to the resource.

The procedure of Weidema (2007, reproduced as Annex IV in this report) aggregates these three physical impact categories, by expressing ecosystem impacts in terms of either human well-being (0.019 QALY/BAHY) or monetary units (1 400 EUR/BAHY), and by introducing a conversion factor between human well-being and monetary units (74 000 EUR/QALY), thus allowing aggregation of all three endpoint indicators in a single impact category ‘human production and consumption efficiency’, measured either in QALYs or in the monetary unit EUR_{2003} to determine the economic externalities of an activity or product system.

This procedure may be applied to any combination of characterisation methods that have humans, ecosystems or resources as category indicators. In Table 13.1, two previously published impact assessment methods are compared to the new Stepwise method. The three methods all use comparable units for impacts on humans, ecosystems and resources. Besides the obvious differences in characterisation models, it appears from this comparison that:

- There are large differences in the completeness of the methods, in terms of impact categories covered.
- When comparing across impact categories, some resulting values of the previously published methods appear to be ‘out of range’, which can be traced back to deficiencies in the characterisation models. Since the data in Table 13.1 refer to the total consumption in EU-27, this also provides the opportunity for a ‘sanity check’ of the characterisation factors: do the results appear realistic in absolute terms? It was by applying such a check that we discovered a significant overlap between the midpoint impact categories ‘ecotoxicity, terrestrial’ and ‘nature occupation’, which led us to omit localised impacts of emissions to soil in the category ‘ecotoxicity, terrestrial’; see Table 13.2.

The development of the Stepwise method, described in the following, provides an example of how the preparation for further modelling to a single endpoint in itself reflects back on the midpoint modelling, in terms of increased demands for consistency and broader coverage of impact categories. The development involved the following steps:

1. Choosing for each impact category, the most developed of recent midpoint characterisation models, adding or modifying for completeness or consistency.
2. Adding missing damage models (from midpoints to BAHY, QALY and/or EUR).

Table 13.1: Comparing the impacts from one year of final consumption in the EU-27, using three different impact assessment methods. The inventory data are identical, which means that the differences are purely caused by the applied characterisation models.

Impact category	Unit	Eco-indicator 99 (H)	IMPACT 2002+	Stepwise 2006 [k]
Impacts on human well-being:				
Global warming / Climate change [a]	DALY / QALY	9.85E+05	[a]	9.95E+04
Human toxicity, carcinogens [b]	DALY / QALY	2.58E+04	4.84E+04	4.84E+04
Human toxicity, non-carcinogens	DALY / QALY		4.76E+04	4.76E+04
Injuries, road or work	DALY / QALY			2.60E+06
Ozone layer depletion [c]	DALY / QALY	4.01E+03	3.15E+03	3.15E+03
Photochemical ozone / Respiratory organics [d]	DALY / QALY	1.15E+04	1.15E+04	1.49E+04
Respiratory inorganics [e]	DALY / QALY	3.33E+06	3.36E+06	3.36E+06
Sum of impacts on human well-being	*DALY / QALY*	*4.36E+06*	*3.47E+06 [a]*	*6.18E+06*
Impacts on ecosystems:				
Global warming / Climate change	BAHY			2.74E+8
Acidification & Eutrophication, terrestrial	BAHY	1.42E+7	1.42E+7	
- Acidification	BAHY			2.08E+6
- Eutrophication, terrestrial	BAHY			8.80E+6
Eutrophication, aquatic	BAHY			2.17E+6
Ecotoxicity	BAHY	9.57E+6		
- Ecotoxicity, aquatic	BAHY		1.54E+6	1.54E+6
- Ecotoxicity, terrestrial [f]	BAHY		1.11E+8	7.33E+6
Nature occupation / Land occupation / Land use [g]	BAHY	3.14E+8	3.14E+8	2.40E+8
Photochemical ozone	BAHY			3.54E+6
Sum of impacts on nature	*BAHY*	*3.37E+8*	*4.41E+8*	*5.40E+8*
Impacts on resource productivity:				
Global warming / Climate change	EUR			-1.72E+09
Human toxicity, carcinogens	EUR			1.11E+09
Human toxicity, non-carcinogens	EUR			1.09E+09
Injuries, road or work	EUR			5.99E+10
Mineral extraction	EUR	3.65E+08 [h]	3.65E+08 [h]	3.65E+08
Non-renewable energy / Fossil fuels	EUR	6.99E+10 [i]	5.59E+11 [i]	0.00E+00
Ozone layer depletion	EUR			7.20E+07
Photochemical ozone, crops	EUR			1.50E+10
Photochemical ozone, human productivity	EUR			3.42E+08
Respiratory inorganics	EUR			7.73E+10
Sum of impacts on resource productivity	*EUR*	*7.03E+10*	*5.59E+11*	*1.53E+11*

a. See text for details. IMPACT 2002+ does not transform the CO_2-equivalents to DALY.
b. Very different characterisation models (EUSES vs. IMPACT 2002).
c. Ecoindicator uses a different source for characterisation factors.
d. Different characterisation models (Hoffstetter 1998 vs. EDIP2003).
e. Ecoindicator does not include a characterisation factor for carbon monoxide.
f. To avoid double-counting with the impact category 'nature occupation', the more localised impacts of emissions to soil are excluded in the Stepwise method.
g. See text for details.
h. MJ surplus energy is converted to EUR using 0.004 EUR/MJ.
i. MJ surplus energy (Ecoindicator) and MJ primary energy (IMPACT 2002+) are converted to EUR using 0.004 EUR/MJ.
j. For impact categories that have no correspondence in Ecoindicator99 or IMPACT 2002, see the text and tables below for more details.

13.3 Choice of impact categories, category indicators and characterisation models

Among the different existing impact assessment methods (see the reviews of Udo de Haes et al. 2002, Pennington et al. 2004, and Jolliet et al. 2004), there is a reasonable similarity in the impact categories included. The difference between the methods lies rather in the characterisation models applied.

For each impact category, a category indicator is chosen and a characterisation model is applied to model the impact of the inventory results on the chosen midpoint, thereby expressing them in a common unit, i.e. the unit of the category indicator.

We have selected a combination of characterisation models from two of the most recent impact assessment methods, the IMPACT 2002+ v. 2.1 method (Jolliet et al. 2003, Humbert et al. 2005) and the year 2010 characterisation factors of the EDIP2003 method (Hauschild & Potting 2005, Potting & Hauschild 2005). Both methods are second-generation methods building on previous work (Ecoindicator99 and EDIP1997, respectively). For some impact categories, we have made minor modifications, as described below.

The main criterion for choosing a specific characterisation model has been completeness in coverage, both in terms of substances included (the characterisation method with the largest number of included substances has been chosen, which is especially relevant for toxicity), and in terms of how much of the impact chain is covered by the model (the characterisation model which covers the largest part of the impact chain and/or provides the best options for site-dependent characterisation has been chosen).

Because of its overall importance to human health, we have added the impact category ‘injuries’ (see below), to complement the impact categories from IMPACT 2002+ and EDIP2003. With this addition, we believe that our midpoint impact assessment method covers all potentially important environmental (biophysical) impact categories, with the exception of noise and invasive species dispersal, both impact categories primarily associated with transport activities.

In general, the methods from IMPACT 2002+ and EDIP2003 do not treat emissions via groundwater separately, i.e. characterisation factors for water emissions are all related to direct emissions to surface waters, and characterisation factors for emissions to soil assume a diffuse emission rather than a point source. No LCA characterisation model is currently available that takes into account the binding of pollutants to soil particles after the release to groundwater and the significant reduction in concentration of emissions compared to the pulse emissions that are normally assumed for the characterisation factors applied to surface water emissions. With a special view to the emissions from landfills, we have therefore introduced specific characterisation factors for groundwater emissions, where the original characterisation factors for surface water are reduced. The reduction factors have been calculated to represent the reduced concentrations of a groundwater emission over 100 years and over 60 000 years (the periods used in the Ecoinvent database inventory; Doka 2003) relative to the equilibrium concentration of a pulse emission, which is the basis for the surface water characterisation factors. Assuming that an equilibrium concentration of a pulse emission is reached after two weeks, the reduction factors for groundwater becomes 2 weeks/100 years = 2/(52*100)

= 4E-4 and 2 weeks/60 000 years = 2/(52*60000) = 6E-7, for emissions before and after 100 years, respectively.

The midpoint impact categories are listed in Table 13.2, indicating the original source and the modifications we have applied.

Table 13.2: Midpoint characterisation models for Stepwise2006, their sources and modifications.

Midpoint impact category	Original source		Comments / modifications
	EDIP 2003	IMPACT 2002+	
Acidification	x		
Ecotoxicity, aquatic		x	Added factors for 'aluminium, ion' emissions
Ecotoxicity, terrestrial		x	To avoid double-counting with the impact category 'nature occupation', the more localised impacts of emissions to soil are excluded here
Eutrophication	x		
Global warming (100 years)	x		
Human toxicity		x	Added factors for 'aluminium, ion' emissions
Injuries, work and traffic			New, see text for description
Ionizing radiation		x	
Mineral extraction		x	
Nature occupation		x	Modified, see text for description
Non-renewable energy		x	
Ozone layer depletion		x	
Photochemical ozone impacts on vegetation	x		
Respiratory inorganics		x	
Respiratory organics	x		Impact of photochemical ozone on humans

13.3.1 New impact category: injuries

The impact category 'injuries' addresses injuries from road traffic and work-related injuries (occupational injuries), i.e. the LCI indicators 'fatal injuries', 'non-fatal road injuries' and 'non-fatal work injuries'. The category indicator is 'fatal-injury-equivalents'.

Hofstetter & Norris (2003) suggest a procedure for including work-related injuries in life cycle assessments. We estimate characterisation factors for both occupational and road traffic injuries from the overall proportion of YLL (Years of life lost) to YLD

(Years-of-life-equivalents lost due to disability) for these causes in the Global Burden of Disease study (Mathers et al. 2004, using the values without discounting and age-weighting), compared to the proportion of reported cases from Eurostat and the CARE road accident database (http://europa.eu.int/comm/transport/care/). These data sources provide us with the values 43 YLL/injury-related death, 0.323YLD/non-fatal road injury, and 0.0333 YLD/non-fatal work injury, from which we derive the characterisation factors 43/0.323 = 133 non-fatal road injuries/fatal injury (death), and 43/0.0333 = 1 300 non-fatal work injuries/fatal injury.

13.3.2 Nature occupation

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

In the IMPACT 2002+ method, a similar impact category exists under the name of 'land occupation', taken directly from Ecoindicator99 (Goedkoop & Spriensma 2001), where the impact is assessed on the basis of the duration of area occupied (m^2*years) multiplied with a severity score, representing the potentially disappeared fraction (PDF) of species on that area during the specified time.

Compared to this method, we have made the following modifications:

- We have applied an estimated severity of 0.8 for the direct impact of urban and intensive agricultural land use (see Millennium Ecosystem Assessment 2005), which is intended to be representative of all species affected, while Goedkoop & Spriensma (2001) arrive at a larger severity, mainly because their value also includes an estimate of the regional effect, i.e. the effect outside the occupied area.
- We assess 'green urban land' as equal to 'continuous urban land', since we define PDF in terms of the potentially disappeared fraction (PDF) of endemic species, i.e. not including alien species.
- We assess that only 30 % of naturally occurring species in pasture areas (meadow lands) are negatively affected by grazing (Landsberg et al. 1997), where the Ecoindicator99 method suggests an impact close to that of other agricultural land uses, mainly as an effect of fertilizer and herbicide use.
- To align this impact category to the marginal approach generally used for other impact categories (implying that the impact measured is that of an additional unit of the stressor, as opposed to an average approach that measures the average impact per unit of stressor), we only include the difference in impact relative to the marginal use of each land type. The marginal land use for arable land is assumed to be conventional agriculture. The marginal land use for pasture and forest lands is assumed to be the natural situation.
- Since all occupation of arable land (all land with potential for agriculture) contributes to the overall pressure leading to current global deforestation, we include an additional severity of 0.88 to represent the secondary impacts from this deforestation, calculated as the nature occupation during the later relaxation from deforestation (Weidema & Lindeijer 2001). Current global deforestation is estimated to 1.5E11 m^2/year. In absence of an adequate characterisation model, we estimate the relaxation time for biodiversity to 500 years (range 300 to 1 300 years), and the average severity during relaxation as 0.2. The resulting value is

allocated over the current global use of arable land (1.7E13 m^2) to arrive at the additional severity of 0.88 for all current uses of arable land.

The resulting characterisation values are shown in Table 13.3.

Table 13.3: Characterisation factors for $1m^2$*year land occupation for different intensities of occupation.

Intensity of occupation	Direct impact	Direct impact relative to marginal land use	Deforest-ation impact	Sum of direct relative & deforestation impacts	Midpoint indicator
	PDF*m^2 *years	PDF*m^2 *years	PDF*m^2 *years	PDF*m^2 *years	m^2-equivalents arable land
Urban and intensive agricultural use of arable land					
Continuous urban land	0.80	0	0.88	0.88	1.00
Construction and dump sites	0.80	0	0.88	0.88	1.00
Green urban land	0.80	0	0.88	0.88	1.00
Conventional agriculture	0.80	0	0.88	0.88	1.00
Integrated agriculture	0.80	0	0.88	0.88	1.00
Intensive meadow land	0.80	0	0.88	0.88	1.00
Less intensive uses of arable land [a]					
Organic agriculture	0.76	-0.04	0.88	0.84	0.95
Organic meadow land	0.71	-0.09	0.88	0.79	0.9
Discontinuous urban land	0.67	-0.13	0.88	0.75	0.85
Industrial area	0.58	-0.22	0.88	0.66	0.75
Rail or road area	0.58	-0.22	0.88	0.66	0.75
Use of non-arable land					
Pasture in high productivity areas	0.30	0.30	0.00	0.30	0.34
Forest land	0.10	0.10	0.00	0.10	0.11

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

13.4 Normalisation

[Note that Chapter 9.2 of the main report is a more relevant text on normalisation for the purpose of this report.]

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

In Stepwise2006, the normalisation reference currently available is the impact per person in Europe for year 1995. The normalised results are therefore expressed in person-years or rather person-year-equivalents of each category impact.

Normalisation values for Europe in year 1995 are provided in Table 13.4. A more recent normalisation reference (for year 2000) is published in a separate paper (Weidema & Wesnaes 2007b).

Table 13.4: **Normalisation references and factors per person in Europe for 1995.**

Impact category	Unit of characterised values	Normalisation factors (Europe 1995)		Source
		Characterised unit/ person-year (normalisation references)	Person-year/ characterised unit (normalisation factors)	
Acidification	m^2 UES	2 200	4.55E-04	[a]
Ecotoxicity, aquatic	kg-eq. TEG water	1 360 000	7.37E-07	[b]
Ecotoxicity, terrestrial	kg-eq. TEG soil	2 350	4.25E-04	[b]
Eutrophication, aquatic	kg NO_3-eq.	58	1.72E-02	[a]
Eutrophication, terrestrial	m^2 UES	2 100	4.76E-04	[a]
Global warming	kg CO_2-eq.	10 620	9.41E-05	[c]
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	45.5	2.20E-02	[b]
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	173	5.77E-03	[b]
Injuries, road or work	fatal injuries-eq.	0.000142	7.04E03	[d]
Ionizing radiation	Bq C-14-eq.	533 000	1.88E-06	[b]
Mineral extraction	MJ extra	292	3.42E-03	[b]
Nature occupation	m^2 arable land	3 140	3.18E-04	[e]
Non-renewable energy	MJ primary	152 000	6.58E-06	[b]
Ozone layer depletion	kg CFC-11-eq.	0.204	4.91E00	[b]
Photochemical ozone – Vegetation	m^2*ppm*hours	140 000	7.14E-06	[a]
Respiratory inorganics	kg PM2.5-eq.	8.8	1.14E-01	[b]
Respiratory organics	person*ppm*hours	10	1.00E-01	[a]

[a] Hauschild & Potting (2005). UES = Unprotected Eco-System.
[b] IMPACT 2002+ v.2.1 (Annex 3 in Humbert et al. 2005). For terrestrial ecotoxicity, the normalisation reference does not include emissions to soil cf. the argumentation in Table 13.2.
[c] Gugele et al. (2005).
[d] Calculated from 39 400 fatal and 1 390 000 nonfatal road injuries (data from the CARE road accident database for EU-15 extrapolated to EU-25 using a factor 1.32 from Eurostat road fatality data), and 6 460 fatal and 5 740 000 non-fatal work injuries (Eurostat data for EU-15 extrapolated to EU-25 using a factor 1.2).
[e] Calculated from the normalisation data of Humbert et al. (2005), using the characterisation factors from Table 13.3.

Since the normalised results do not express any statement of importance for each impact category, normalised results should not be aggregated or compared across impact categories.

13.5 Damage modelling

13.5.1 Impacts on ecosystems

For acidification and terrestrial eutrophication, the damage model has been taken from the EcoIndicator99 (Goedkoop & Spriensma 2001) and related to the EDIP2003 midpoint characterisation factors, as shown in Table 13.5 and Table 13.6. In the calculation of the damage factors, it is a complication that the EcoIndicator99 factors for NO_x and NH_3 represent the sum of the contributions from acidification and terrestrial eutrophication since Ecoindicator99 does not separate these at the midpoint level. A pragmatic solution has been to calculate the acidification damage factor for SO_x (by dividing the EcoIndicator99 damage factors by the EDIP2003 characterisation factors) and assume that this relationship of midpoint to damage is representative also for the acidification damage caused by NO_x and NH_3 (Table 13.5). Furthermore it is assumed that the rest of the damage caused by these emissions are due to the terrestrial eutrophication, and hence that there is no overlap between the areas damaged by acidification and terrestrial eutrophication. Comparison of the ratios between the S- and

N damaged areas according to EcoIndicator99 and according to EDIP2003 (area of unprotected eco-system (UES) due to acidification from SO_x divided by area of UES due to acidification or eutrophication by NO_x and NH_3) indicates that this assumption is reasonable. Since the midpoint that is modelled in EDIP2003 is the variation in area of unprotected ecosystem due to the eutrophication exposure, one would expect that the damage factor is independent of the substance, as was also assumed for acidification. The damage to midpoint ratios found in Table 13.6 are a bit different for NO_x and NH_3. This is probably due to modelling artefacts residing in the fact that the damage modelling (in EcoIndicator99) is performed in one model assuming a Dutch ecosystem sensitivity distribution to be representative for Europe, while the midpoint modelling (in EDIP2003) is performed in another, spatially differentiated, model with information about the actual ecosystem sensitivity distribution for grid cells all over Europe. What can be achieved by this approach is thus the order of magnitude of the damage to midpoint ratio, determined as the average of the ratios found for NO_x and NH_3.

Table 13.5: Calculation of damage (endpoint) characterisation factors for acidification impacts.

Substance	EI99 damage factor for Europe (Goedkoop & Spriensma 2001)	EDIP2003 acidification characterisation factor (Potting & Hauschild 2005)	EDIP2003 acidification damage factor for Europe [a]
	PDF·m²·year / g	m²UES / g	PDF·m²·year / m² UES
SO_x	1.04E-3	1.9E-2	5.47E-2
NO_x	5.71E-3	6.4E-3	5.47E-2
NH_3	1.56E-2	3.0E-2	5.47E-2
Acidification	-	-	5.47E-2

[a] The EDIP2003 damage factor appears by dividing the EI99 damage factor by the EDIP2003 characterisation factor for SOx. This damage factor is then applied as a general damage factor for all contributing substances.

Table 13.6: Calculation of damage (endpoint) characterisation factors for terrestrial eutrophication impacts.

Substance	EI99 damage factor for Europe (Goedkoop & Spriensma 2001)	EDIP2003 terrestrial eutrophication characterisation factor (Potting & Hauschild 2005)	EDIP2003 terrestrial eutrophication damage factor for Europe [a]
	PDF·m²·year / g	m²UES / g	PDF·m²·year / m² UES
SO_x	1.04E-3	0	0
NO_x	5.71E-3	3.3E-2	0.12
NH_3	1.56E-2	0.14	5.7E-2
Terr. eutrophication	-	-	8.85E-2

[a] The EDIP2003 damage factor for terrestrial eutrophication appears as an average of the calculated damage factors for NOx and NH_3. These damage factors for the individual substances are calculated by dividing the EI99 damage factor by the EDIP2003 characterisation factor, subtracting the 5.47E-2 PDF*m²*years already attributed to acidification in Table 58.

For aquatic eutrophication, a damage model has until now been missing. (For further details on our specific developments for this project, see Annex III.)

All the damage characterisation factors for ecosystem impacts are provided in Table 13.7. Normalisation references are also provided.

Table 13.7: Damage (endpoint) characterisation factors and normalisation references for impacts on ecosystems.

Impact category	Unit of characterised values	BAHY / characterised unit	BAHY / person-year (normalisation references)	Source
Acidification	m^2 UES	5.47E-06	1.20E-02	[a]
Ecotoxicity, aquatic	kg-eq. TEG water	5.02E-09	6.8E-03	[b]
Ecotoxicity, terrestrial	kg-eq. TEG soil	7.91E-07	1.9E-03	[b]
Eutrophication, aquatic	kg NO_3-eq.	7.2E-05	4.17E-03	[a]
Eutrophication, terrestrial	m^2 UES	8.85E-06	1.86E-02	[a]
Global warming	kg CO_2-eq.	5.82E-05	6.18E-01	[a]
Nature occupation	m^2 arable land	8.8E-05	2.76E-01	[c]
Photochemical ozone – Vegetation	m^2*ppm*hours	6.59E-08	9.3E-03	[d]

a. See the text for details.

b. Humbert et al. (2005). Normalisation references calculated from accompanying spreadsheet. For terrestrial ecotoxicity, the normalisation reference does not include emissions to soil; cf. the argumentation provided in the description of the midpoint impact category.

c. See Table 13.3.

d. An adequate damage model for photochemical ozone impacts on vegetation is not available. In order not to omit this potentially important impact category, we have assumed a proxy value corresponding to 1 % of the total European area (4E8 ha) or 4E6 BAHY / year, based on an assumed proportionality between the estimated net primary production (NPP) loss of 10 % and the loss of plant species from the exposed ecosystems, applying an uncertainty factor of 10 between NPP loss and species disappearance. Using the European normalisation reference for m^2*ppm*hours from EDIP2003 and a European population of 431 000 000, as in IMPACT 2002+, this gives a damage factor of 6.59E-8 BAHY/ m^2*ppm*hour and a normalisation reference of 9.3E-03 BAHY / person-year.

A damage modelling for global warming impacts on nature has hitherto been missing. Our below attempt is only a first rough estimate, which should primarily be seen as preferable to an omission of this potentially important impact category from the damage modelling. We assume that more elaborated models will soon become available from the climate change research community.

Thus, we estimate the impacts of global warming as the consequences of a 2.5 K temperature increase corresponding to a central estimate (IPCC 2001, Watson et al. 2001) for a doubling of the CO_2 concentration in the atmosphere, equal to a global concentration increase of 370 ppm by volume or an emission of 8E14 kg C or 2.93E15 kg CO_2. For mid-range climate scenarios, and assuming perfect dispersal, Thomas et al. (2004b) calculate that 4-13 % of all species will lose 100 % of their climatically suitable areas by year 2050, and 9-32 % will lose over 90 % of their climatically suitable areas, with 4 and 14 % of all species as the central estimates. A loss of 90 % of the climatically suitable area was estimated to give a 44 % chance of extinction (Thomas et al. 2004b). As we do not seek to model species extinction, but rather lost species-area (which may eventually lead to extinction), we have applied 4 % + 0.9*(14 %-4 %) = 13 % of the global species-area as the central estimate. With a global terrestrial area of 1.3E10 ha, this corresponds to a lost area of 1.7E9 ha. If we had also included species losing over 50 % of their climatically suitable area, this would correspond to 27.5 % of the global species-area or 3.6E9 ha, based on the 47 % of species affected according to Thomas et al. (2004b). The forecasts of Thomas et al. were recently confirmed by Malcolm et al. (2006) calculating an average extinction of 12 % of endemic species, using a slightly different method.

Although relaxation from the climate effect (understood as a return to the previous climate vegetation) is less likely to occur than relaxation from deforestation, we have applied the same assumptions on relaxation from global warming as applied for relaxation from deforestation, i.e. 500 years relaxation time, and an average severity during relaxation of 0.2 (see the description of the impact category 'nature occupation', above). Thereby, we arrive at a characterisation factor of 5.82E-5 BAHY/kg CO_2-equivalents (0.2* 500 years* 1.7E9 ha / 2.93E15 kg CO_2-equivalents).

Although we generally advocate the use of best estimates for calculation of characterisation factors (rather than low or high estimates), the estimate made here for global warming is a rather low estimate, since a number of modest assumptions are made by Thomas et al. (perfect dispersal, only including species losing >90 % of their climatically suitable area, full relaxation). However, even with this low estimate, our trial runs of the Stepwise impact assessment method show that the impacts from global warming will dominate the assessments of most product systems.

13.5.2 Impacts on human well-being

For most midpoint impact categories, the modelling from midpoint indicator results to human well-being impacts in QALY is documented in the same sources as mentioned above in the description of the midpoint impact categories, noting that 1 QALY = -1 DALY. For the midpoint impact categories derived from IMPACT 2002+, the endpoint characterisation models (damage models) are described in Humbert et al. (2005). For respiratory organics, the damage modelling is described in Hofstetter (1998). All the damage characterisation factors for human well-being are provided in Table 13.8. Normalisation references are also provided.

Table 13.8: Damage (endpoint) characterisation factors and normalisation references for impacts on human well-being.

Impact category	Unit of characterised values	QALY / characterised unit	QALY / person-year (normalisation references)	Source
Global warming	kg CO_2-eq.	2.11E-08	2.24E-04	[a]
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	2.80E-06	1.28E-04	[b]
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	2.80E-06	4.86E-04	[b]
Injuries, road or work	fatal injuries-eq.	43	6.09E-03	[c]
Ionizing radiation	Bq C-14-eq.	2.10E-10	1.12E-04	[b]
Ozone layer depletion	kg CFC-11-eq.	1.05E-03	2.14E-04	[b]
Respiratory inorganics	kg PM2.5-eq.	7.00E-04	6.16E-03	[b]
Respiratory organics	person*ppm*hours	2.64E-06	2.64E-05	[d]

a. See the text for details.
b. Humbert et al. (2005).
c. Mathers et al. (2004).
d. Damage modelling is performed with the epidemiological approach of Hofstetter (1998), in parallel to Ecoindicator99, applying the normalisation reference from Hauschild & Potting (2005).

As for impacts on ecosystems, the impacts of global warming has been estimated as the consequences of a 2.5 K temperature increase corresponding to a central estimate for a doubling of the CO_2 concentration in the atmosphere, equal to a global concentration increase of 370 ppm by volume or an emission of 800 Gt C or 2.93E15 kg CO_2. The uncertainty range on the temperature increase at CO_2 doubling (known as the climate sensitivity) is 1.5-4.5 K and the temperature response to increasing CO_2 concentration is logarithmic (IPCC 2001, Watson et al. 2001). We summarize the impacts on human well-being in terms of 2.1E-8 QALY/kg CO_2-equivalent, caused by 4.8E5 additional cases of vector-borne diseases at 50 QALY/case, 8.8E6 QALYs as a net change in heat and cold related diseases, 4.8E6 relocations due to sea-level rise at 1 QALY per case, and 2.4E7 QALYs as the impact from additional diarrhoea. The QALYs/case for the different diseases are the same as in Ecoindicator99 (Goedkoop & Spriensma 2001) and the incidence values are rough estimates based on our interpretation of Tol (2002). A comparison of our resulting value to that of Ecoindicator99 (2.1E-7 DALY / kg CO_2) shows that our value is an order of magnitude lower. The difference between our interpretation and that of Ecoindicator99 value is likely to be in the interpretation of the number of cases of malaria, since this value dominates the Ecoindicator99 value. An additional explanation for a difference is that Ecoindicator99 does not include negative damage (i.e. benefit), except when it compensates positive damage within the same region (Goedkoop & Spriensma 2001). Although the difference between the estimates may seem large, it should be noted that the importance of health impacts is only a small part of the total impact from global warming in our method (0.76 % with our estimate and 7 % with that of Ecoindicator99), since the overall impact is dominated by the impact on nature; see above. This means that even if the health impacts may be underestimated with our interpretation, this would only have a small influence on the overall assessment of the importance of global warming.

13.5.3 Impacts on resource productivity

In the widest sense, the term resource signifies an available means of production or consumption. Resources available for production are also known as production factors. In this sense, human resources are the available labour force with its different productive abilities, biotic resources are the natural or manipulated biota with its inherent or artificially enhanced abilities to grow and propagate, and abiotic resources are the natural or manufactured raw materials or catalysts for human or biotic production. Although not entering into production itself, the social conditions, such as institutions, rule of law, trust, and human networks, are also prerequisites or catalysts for production, and may therefore be seen as social resources. The term capital is often used as a synonym for resources, typically divided in human capital, social capital, natural capital, and manufactured capital; and when monetised: financial capital.

Life Cycle Assessment has traditionally ignored impacts on resources, with the exception of impacts of dissipation of abiotic natural resources. In contrast, impacts on human and manufactured capital have been the primary focus of cost-benefit analyses. We have analysed the different estimates provided in the RED database (www.red-externalities.net), and found that the most important impacts are human health impacts and the impacts on agricultural production from global warming and photochemical ozone. In the Stepwise method, we therefore limit ourselves to providing characterisation factors for these impacts and – following the tradition in Life Cycle Impact Assessment – the impact of dissipation of abiotic natural resources. In particular, we have currently not spent any effort to include the impact on buildings, nor the fertilisation effect of NOx emissions, due to the relatively small size of these impacts compared to the health impacts, as shown by the ExternE study (Bickel & Friedrich 2005).

All the damage characterisation factors for impacts on resource productivity are provided in Table 13.9. Normalisation references are also provided.

In addition to the direct impact on human well-being recorded in Table 13.8, the direct health impacts listed there also impact indirectly on human productivity in terms of lost labour and/or treatment costs. For each of the midpoint impact categories it would be possible to model this impact on human productivity specifically (see for example Miller et al. 1998), taking into account the severity and treatment costs for the involved disabilities and taking into account only life-years lost in the productive age. For the Stepwise method, we have currently not had enough resources to perform such detailed modelling, and have therefore resorted to the general observation that when expressing losses in economic production output in percentage of GDP per capita (e.g. one lost work year = 100 % of GDP per capita) the corresponding QALY value (1 QALY per lost work year) is a good proxy for the economic impact (i.e. 1 QALY equals an economic loss of 1 per capita GDP) when applying the same discounting rates for both (Miller et al. 2000). As a general proxy, we therefore estimate the loss of economic production from a health impact of 1 QALY in Europe to be 23 000 EUR_{2003}, which is the 2003 GDP per capita for EU-25.

Global warming has both positive and negative influences on agricultural yields. Tol (2002) summarises the available global studies for impacts until year 2200 and for a central 2.5 degrees temperature increase we interpret his conclusion as a net positive impact of approx. 2.5E12 EUR_{2003} or 8.5E-4 EUR_{2003} / kg CO_2.

The midpoint indicator 'mineral extraction' measures the difference between the current energy requirement for extraction and an estimated future energy requirement for extraction from lower grade ores. As alternative energy sources to fossil fuels are currently becoming competitive, there is no reason to assume that long-term energy prices will exceed the current energy prices. We therefore apply a damage (endpoint) characterisation factor of 0.004 EUR_{2003} / MJ extra, based on current energy prices, without discounting of future costs. The total impact is 1.2 EUR_{2003} / person-year, using the normalisation reference from Table 13.4.

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

For impacts from photochemical ozone on agricultural crop production, we apply a rough estimate of a 10 % reduction in crop yields caused by the current emission levels in Europe (Hauschild & Potting 2005), and apply this to the annual crop production value of 1.7E11 EUR_{2003}.

Table 13.9: Damage (endpoint) characterisation factors and normalisation references for impacts on resource productivity.

Impact category	Unit of characterised values	EUR_{2003} / characterised unit	EUR_{2003} / person-year (normalisation references)	Source
Global warming	kg CO_2-eq.	-3.65E-04	-3.9	[a]
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	6.44E-02	2.9	[b]
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	6.44E-02	11	[b]
Injuries, road or work	fatal injuries-eq.	9.89E+05	140	[b]
Ionizing radiation	Bq C-14-eq.	4.83E-06	2.6	[b]
Mineral extraction	MJ extra	4.00E-03	1.2	[c]
Non-renewable energy	MJ primary	0	0	[c]
Ozone layer depletion	kg CFC-11-eq.	24	4.9	[b]
Photochemical ozone – Vegetation	m^2*ppm*hours	2.80E-04	39	[d]
Respiratory inorganics	kg PM2.5-eq.	16.1	142	[b]
Respiratory organics	person*ppm*hours	6.07E-02	0.6	[b]

a. The negative damage (i.e. benefit) to resource productivity is the net effect of the health impact on human productivity calculated as in note [b] and a net increase in agricultural production of 8.5E-4 EUR_{2003} / kg CO_2, based on our interpretation of Tol (2002).

b. The QALY values recorded in Table 13.8 multiplied by 23000 EUR_{2003}.

c. See the text for explanation.

d. Applying the rough estimate of a 10 % reduction in crop yields to the annual European crop production value of 1.7E11 EUR2003, we obtain a total impact on crop production of 1.7E10 EUR2003 per year or 39EUR / person-year. With the normalisation values from Table 4, this gives us a damage characterisation factor of 2.8E-4 EUR2003 / m^2*ppm*hour.

13.6 Uncertainty in the impact assessment methods

Estimates of uncertainties on the characterisation factors are generally available in the methods supplying the characterisation factors, i.e. EDIP2003 (Hauschild & Potting 2005, Potting & Hauschild 2005) for acidification, eutrophication and photochemical ozone formation, and IMPACT 2002 for human toxicity and ecotoxicity (where Jolliet et al. 2003 and Humbert et al. 2005 suggest a precision of a factor 100). For the remaining impact categories taken from IMPACT 2002+ (ionising radiation, mineral extraction, non-renewable energy, ozone layer depletion and respiratory inorganics) as well as for the endpoint characterisation factors for the EDIP2003 impact categories (with the exception of aquatic eutrophication), the characterisation models are taken over directly from Ecoindicator99, for which the uncertainties are provided in Goedkoop & Spriensma (2001).

The damage model for aquatic eutrophication is based on rather small data sets, both geographically and temporally, and we therefore estimate the uncertainty on the overall damage factor to be at least +/- 50 %. The 30:1 ratio between N and P is also subject to an uncertainty of the same size.

For the midpoint global warming characterisation factors (kg CO_2-equivalents / kg substance), the IPCC suggests an uncertainty of 30 % for other substances than CO_2. For the endpoint characterisation factors for global warming, the uncertainties are large, as indicated in Tol (2002) and Thomas et al. (2004b). The uncertainty on the temperature effect of CO_2 doubling is 1.5-4.5 K around the central estimate of 2.5 K and the temperature response to increasing CO_2 concentration is logarithmic (IPCC 2001, Watson et al. 2001). As mentioned above, we have deviated from the principle of applying best estimates for calculation of the endpoint impacts of the temperature increase. Our rather low estimate for the dominating ecosystem effects implies that the effects corresponding to 1.5 K should be seen as a lower bound, while the upper bound will be well beyond the effects corresponding to a 'linear' interpretation of the 4.5 K estimate.

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

For nature occupation, the uncertainty for the impact category 'land use' of Goedkoop & Spriensma (2001) can be applied as a basic uncertainty. For occupation of arable land (all land with potential for agriculture), we have included an additional severity of 0.88 to represent the secondary impacts from current deforestation. For this additional severity, the most critical assumption is the relaxation time. According to Dobben et al. (1998), the relaxation time to reach potential biomass varies from 50 to 220 years depending on latitude and altitude. Weidema & Lindeijer (2001) suggest that the relaxation times for biodiversity are a factor six higher than for biomass, i.e. 300 to 1 300 years. This may be taken as a rough estimate of the uncertainty around our applied central estimate of 500 years relaxation time.

13.7 Discussion and conclusion

From our application of the Stepwise2006 method to different case studies (Weidema & Wesnaes 2006, 2007a, Weidema et al. 2006, 2007), we particularly note the importance for the results of quantifying the global warming damage to ecosystems, and the more modest role of resource inputs when these are placed in the context of other impacts on resource productivity. This can also be seen from the comparison in Table 13.1.

Although many of our modifications and additions to the characterisation models have the nature of first rough attempts and estimates, we find that these are justified by the increased consistency and completeness obtained. The described uncertainties should of course be heeded in any practical application.

13.8 Outlook

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

- A specific characterisation model for groundwater emissions is missing.
- The speciation of metal emissions in the inventory databases is deficient.
- The characterisation models for metal toxicity do not adequately reflect the changes in bioavailability of the metal emissions over time in different environments.
- Midpoint characterisation models for traffic noise and invasive alien species still need to be integrated into the impact assessment methods.
- The endpoint characterisation models for ecotoxicity should be better calibrated to reflect the overall importance of ecotoxicity relative to other impacts on ecosystems.
- The endpoint characterisation model for marine eutrophication should be further elaborated with respect to the relationship between oxygen depletion level and species disappearance, especially over longer periods, and calibrated with a larger geographical and temporal dataset.
- An endpoint characterisation model for ozone impacts on vegetation is missing. This affects both the assessment of ecosystem impacts and impacts on agricultural crop production.
- A separate impact category for agricultural crop production should be created, which should include both the impact of ozone and the impacts of other ecotoxic substances on crop yields, the fertilisation effect of CO_2 and the different mineral nutrients in emissions, as well as soil losses through erosion.
- A characterisation model for ecosystem impacts during relaxation after deforestation and climate impacts is missing.
- The lack of a site-dependent characterisation model for respiratory inorganics is seen as a major shortcoming for the site-specific impact assessment.
- The available normalisation reference for Europe is from 1995. It should be updated, preferably on a continuous basis.
- The endpoint characterisation model for global warming should be improved and better documented.

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14 ANNEX III. DAMAGE ESTIMATES FOR AQUATIC EUTROPHICATION

Eutrophication damage from nutrient enrichment of aquatic ecosystems

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14.1 Introduction

In the life cycle impact assessment (LCIA) of nutrient emissions, eutrophication impacts are targeted in the form of changed species composition as the ecosystem changes from oligotrophic (nutrient poor) to eutrophic (nutrient rich).

The elements that have the ability to cause these changes are the macronutrients nitrogen and phosphorus, which are essential to the growth of algae and macrophytes in the aquatic systems. The algae are essential for the eutrophication impacts since increased growth of algae has the potential to

- reduce the visibility of the water and thereby reduce the extension of macrophytes and change the composition of the fish community in lakes by favouring herbivores over carnivores (predators) among the fish;
- use oxygen in the bottom strata of lakes and coastal waters through their mineralisation. In combination with stratification of the waters (due to stable thermal or saline gradients throughout the water column), this may lead to oxygen depletion of the bottom strata, which in extreme cases eliminates all life in the impacted regions.

Typically, it is either N or P that limits the growth of algae during the summer months with high light intensity. Although there are specific exceptions to the rule, it is generally observed that P is the limiting nutrient in freshwater systems (lakes) while N is the limiting nutrient in most marine systems.

In midpoint characterisation of nutrient enrichment impacts, the results are often reported as N or P-equivalents. EDIP2003 (the Carmen model) gives a spatially differentiated prediction (at the level of countries) of the fraction of the emitted substances that will expose aquatic ecosystems (freshwater or coastal). Combined with information about the nutrient load in the different substances (as given in the EDIP97 characterisation factors for nutrient enrichment) this allows predicting the amount of macronutrients (N and P) that reaches the aquatic ecosystems. The metric is kg N/kg emitted or kg P/kg emitted, and for endpoint characterisation of nutrient enrichment there is thus a need to translate the midpoint score, expressed as kg N or P-equivalents into the damage it causes to the exposed ecosystems.

Damage is difficult to quantify for eutrophication impacts since the aquatic systems are dynamic and may be naturally transient over time, i.e. an oligotrophic lake may over the centuries become eutrophied through natural leaching of nutrients from the surroundings and gradually turn into a bog with dramatic consequences for its species composition. It is difficult to identify any of the intermediary stages as the ‘natural’ stage and hence difficult to identify the reference against which impacts of anthropogenic nutrient emissions shall be measured.

The metrics of damage characterisation for the area of protection ‘natural environment’ (sic!) in LCIA is typically the product of the potentially disappeared fraction of species (the pdf), the area over which this fraction is disappeared and the duration over which it is disappeared, hence: pdf·m^2·yr ([1]).

([1]) In the main report, this technical term is replaced by the term ‘species-weighted m^2*years”

14.2 Marine eutrophication

Eutrophication of marine waters is typically observed in the more shallow coastal waters, and in terms of nutrient emissions, it is generally attributed to the anthropogenic emissions of N.

For quantification of the damage from nitrogen emissions to marine waters, a modification of the Danish MIKE 3 model developed by the Danish Hydraulic Institute (DHI) has been found useful. The model is a 3D dynamic deterministic (process-based) model developed for simulation of the consequences of different future emission scenarios for nutrients, in terms of eutrophication impacts on Danish coastal waters. The model includes hydrodynamic conditions (currents, water level, temperature, salinity) and predicts eutrophication indicators (nutrient concentrations, primary production, and oxygen concentrations). It covers roughly the coastal waters shown in Figure 14.1.

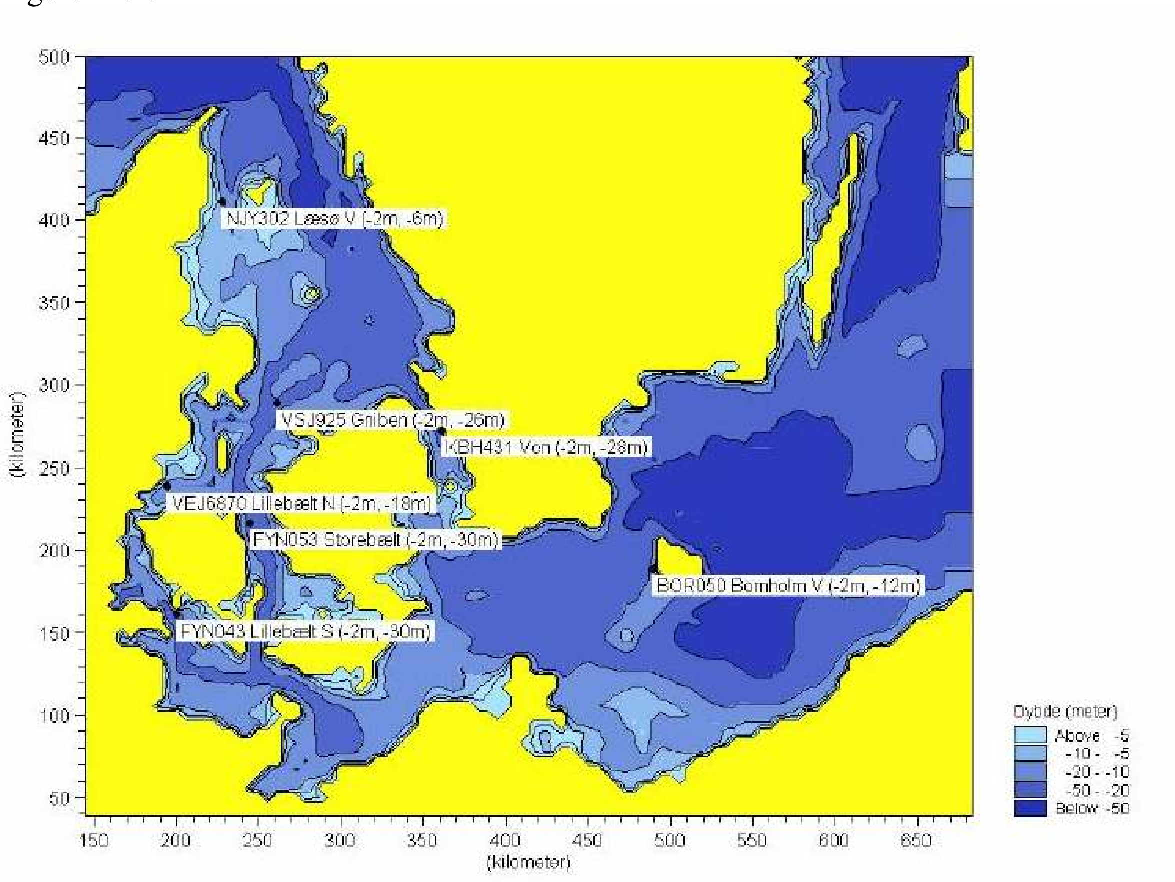

Figure 14.1: Approximate extension of the coastal waters covered by the simulation model. The labels indicate monitoring stations providing the data against which the model was calibrated (Hansen et al., 2003).

Figure 14.2 and Figure 14.3 present model simulations of the area of sea bottom exposed to oxygen depletion once or repeatedly throughout the year as a function of the annual nitrogen load to the modelled region. The load levels indicated by labels on the graph correspond to different nitrogen loading scenarios. The points indicated by 'År 2000' and 'År 2002' represent the actual loading and actual occurrence of oxygen depletion in the years 2000 and 2002. The difference in N loading between the two years is negligible, but the difference in the area of observed oxygen depletion in the Baltic and North Sea region is significant reflecting the strong influence by factors other than N, particularly the weather (which influences run-off and leaching through precipitation patterns and water exchange through wind patterns). As visible from the graph, the model is calibrated against the boundary conditions of year 2000.

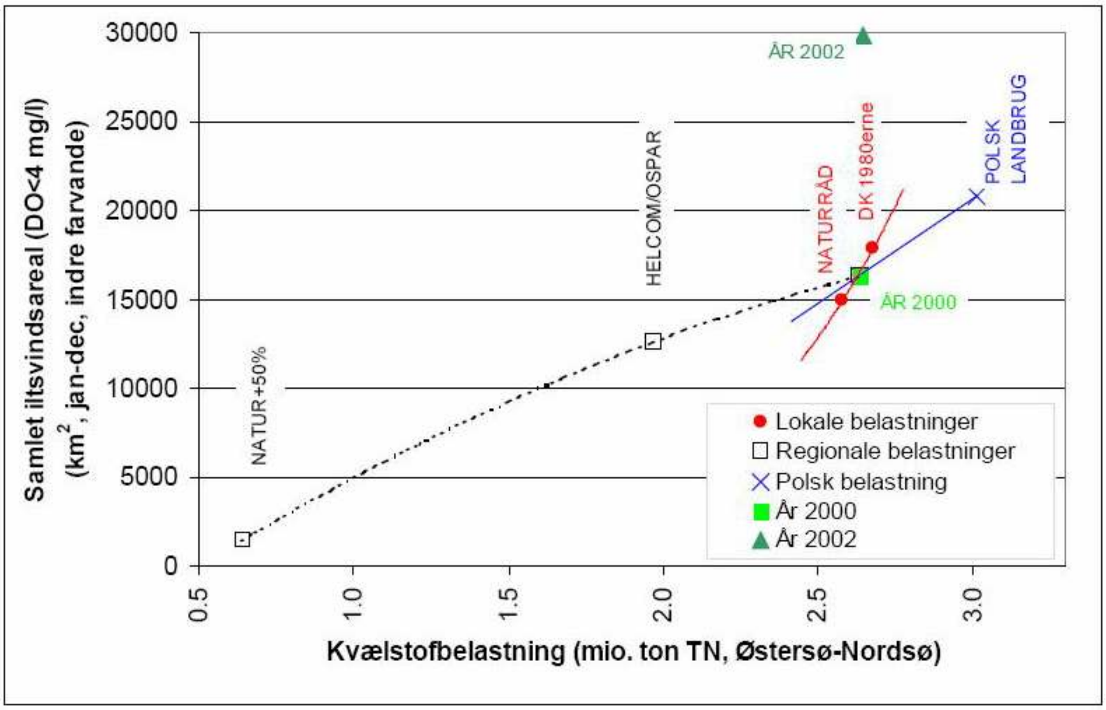

Figure 14.2: **Annual extent of oxygen depletion (< 4 mg O_2/l) in the Danish coastal waters as a function of the nitrogen load (water and air-borne).**

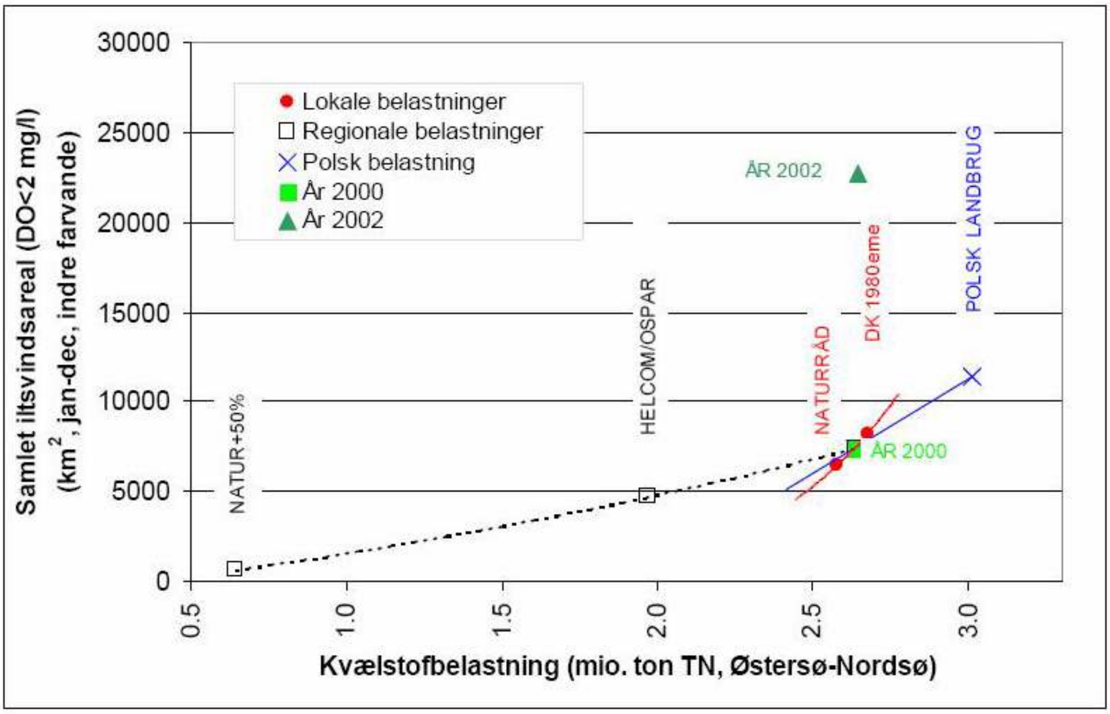

Figure 14.3: **Annual extent of oxygen depletion (< 2 mg O_2/l) in the Danish coastal waters as a function of the annual nitrogen load (water and air-borne).**

14.2.1 Area exposed to damage

The curves in Figure 14.2 and Figure 14.3 show the area of the sea bottom which is exposed to oxygen depletion once or repeatedly through the year, reaching bottom stratum concentrations of oxygen below 4 mg/l (Figure 14.2) or below 2 mg/l (Figure 14.3). The slope of the curves represents the increase in damage in terms of oxygen depletion (the ultimate eutrophication damage) per increase in nutrient load, i.e. a potential damage factor for nutrient enrichment impacts.

As seen from the two curves, the relationship between the size of the oxygen depleted area and the N loading, which is predicted by the complex model, turns out to be closely linear at this macro level, which means that regardless the background load, the slope (and hence the damage factor) is roughly the same; roughly 7 500 km^2 per million ton N, which is equivalent to 7.5 m^2 per kg N (oxygen depletion defined as bottom concentration below 4 mg/l O_2), and roughly 3 200 km^2 per million ton N, which is equivalent to 3.2 m^2 per kg N (below 2 mg/l O_2).

14.2.2 Affected fraction of species

In order to estimate the order of magnitude of the damage factor for marine aquatic eutrophication (under Danish conditions), assume that the pdf for areas exposed to O_2 concentrations below 2 mg/l is 1, i.e. the bottom becomes lifeless at these low oxygen concentrations. This assumption, as well as the duration of a situation with a pdf of 1, evidently depends on the duration of the oxygen depletion event, but it still serves to determine an order of magnitude of the damage factor.

14.2.3 Duration of damage

Due to the annual cycle of the region's weather, the oxygen depletion is always broken during autumn where fresh oxygen-rich surface waters are mixed with the depleted bottom waters and life may recolonize the former lifeless bottom regions. If severe oxygen depletion hits the same areas every year for a long time, this leads to lasting changes in the species richness, but for estimating the order of magnitude of the damage factor we may assume that the duration of the damage (the pdf reduction) is one year.

14.2.4 Damage factor for marine eutrophication

Combining the information on area (3.2 m^2/kg N for O_2 concentrations lower than 2 mg/l), the potentially disappeared fraction of species (1 for $O_2 < 2$mg/l) and the duration (1 yr), gives the following damage factor for marine eutrophication: 3.2 pdf·m^2·year/kg N.

This factor should only be taken as an indication of the order of magnitude of the marine eutrophication damage – a number of aspects deserve further consideration, including:

- Relationship between oxygen depletion level and pdf;
- Long-term trends in pdf (pdf relative to species composition if a partial annual restoration occurs);
- Weather conditions (cf. difference between 2000 and 2002);

- Topography and water exchange – the model and the derived values are valid for Danish coastal waters and might look different for other coastal waters.

14.3 Freshwater eutrophication

Eutrophication of freshwater systems is mainly observed in lakes and slowly running rivers. It is generally attributed to P, which tends to be the limiting nutrient in most freshwater systems.

Based on a database of the Dutch organisation STOWA (http://www.stowa.nl/) compiling a large number of observations of the species occurrence in Dutch freshwater systems and concomitant concentrations of P in the water phase, Struijs and co-workers have derived the data in Figure 14.4, showing the macrofauna species diversity versus the P concentration.

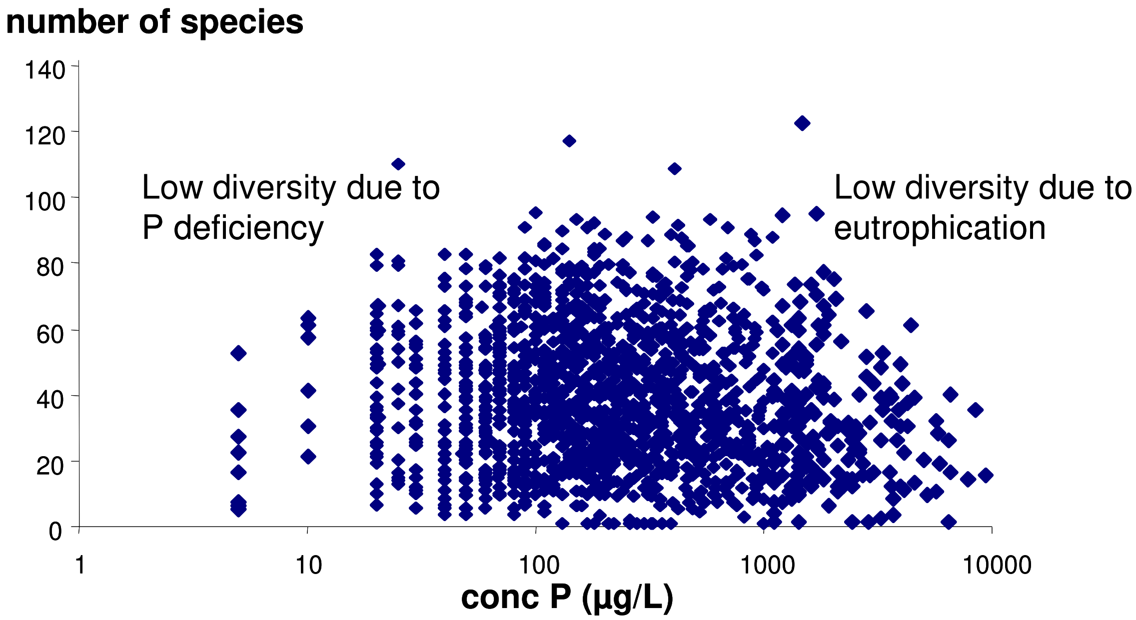

Figure 14.4: Number of species versus P-concentration in freshwater ecosystems in the Netherlands (Struijs et al., 2007).

From the data in Figure 14.4, Struijs and co-workers discarded all biosurvey data with P-concentration CP below 100 µg/l and aggregated species into genera (in total 251). For each genus the relative abundance was evaluated for 20 intervals of log CP. For simplicity, a relative abundance below 0.05 was considered absence (= 0), and otherwise the species is considered present (= 1). The variation among the 251 *genera* in response to log CP represents a Species Sensitivity Distribution (SSD). For each CP interval the Potentially Disappeared Fraction (PDF) is computed according to: PDF = 1 – (number present/251) and plotted versus CP as illustrated in Figure 14.5.

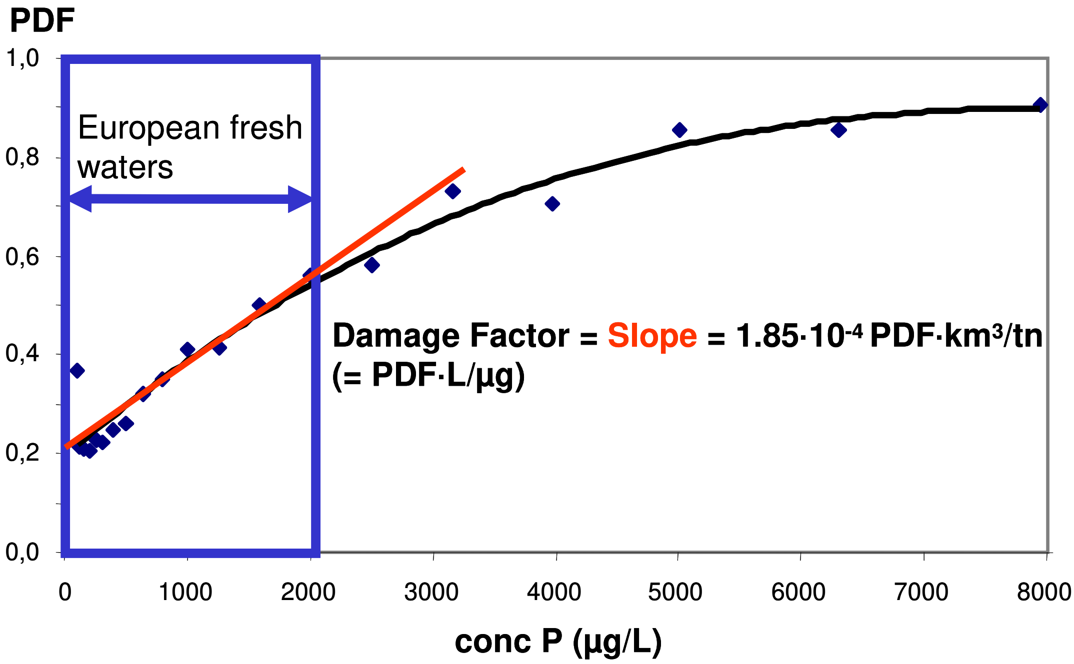

Figure 14.5: Potentially disappeared fraction of species, pdf (of macrofauna genera) versus P-concentration in freshwater ecosystems in the Netherlands (Struijs et al., 2007).

From the plot of pdf versus the P concentration, a damage factor expressing the increase in pdf per P concentration increase is determined as the slope of the curve in Figure Figure 14.5. In the concentration interval observed in European fresh water systems, the slope is around $1.85 \cdot 10^{-4}$ pdf/(µg P/l) (Struijs et al. 2007).

Assuming that

- the removal or immobilisation of P, which occurs between the emission location and the freshwater body where eutrophication is observed, is considered in the midpoint characterisation (as is the case applying EDIP2003 exposure modifier on the characterisation factors)
- the average depth of the freshwater systems is 2 m,

this damage factor translates into:

$$1.85 \cdot 10^{-4} \text{ pdf/(µg P/l)} = 1.85 \cdot 10^{-4} \text{ pdf·l/µg P} = 1.85 \cdot 10^{-4} \text{ pdf·m}^3\text{/mg P} = 0.93 \cdot 10^{-4} \text{ pdf·m}^2\text{/mg P} = 93 \text{ pdf·m}^2\text{/kg P}$$

Assuming a duration of one year, with the reasoning as given for marine eutrophication, and rounding this damage factor can be translated into: 93 pdf·m^2·year/kg P

As for the damage factor for marine eutrophication, this factor should be interpreted only as an order of magnitude for aquatic eutrophication in freshwater systems. The assumptions behind the disappearance curve in Figure 14.5 are crude, and many other factors equally influencing the species number are disregarded in this approach. This should be further investigated in further studies.

This being said, it is reassuring for the validity of the above calculations that the resulting ratio between the specific (per kg) eutrophication damages for P and N (93 pdf·m^2·years / 3.2 pdf·m^2·years = 29) is in the same order of magnitude as the Redfield ratio, expressing the ratio between P and N in the composition of plankton algae. The molar Redfield ratio between P and N is 16, or when expressed in weight units, i.e. adjusted for the molar weights, 16*14/31 = 7.2.

References for Annex III

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Struijs J, Beusen A H W, De Zwart D, Huijbregts M A J. (2007). Ecological damage factors for eutrophication in European freshwaters in Life Cycle Impact Assessment (LCIA). Poster presented at the 17th annual meeting of SETAC Europe, Porto, 20-24 May, 2007.

15 ANNEX IV. USING THE BUDGET CONSTRAINT TO MONETARISE IMPACT ASSESSMENT RESULTS ([1])

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15.1 Abstract

Recent developments in Life Cycle Impact Assessment (LCIA) provide a basis for reducing the uncertainty in monetarisation of environmental impacts. The LCIA method 'Ecoindicator99' provides impact pathways ending in a physical score for each of the three safeguard subjects: humans, ecosystems, and resources. We redefine these damage categories so that they can be measured in terms of Quality Adjusted Life Years (QALYs) for impacts on human well-being, Biodiversity Adjusted Hectare Years (BAHYs) for impacts on ecosystems, and monetary units for impacts on resource productivity.

The monetary value of a QALY can be derived from the budget constraint, i.e. the fact that the average annual income is the maximum that an average person can pay for an additional life year. Since a QALY by definition is a life-year lived at full well-being, the budget constraint can be determined as the potential annual economic production per capita at full well-being. We determine this to be EUR 74 000 with an uncertainty estimate of EUR 62 000 to EUR 84 000. This corresponds well to the EUR 74 627 willingness-to-pay estimate of the ExternE project. Differences to other estimates can be explained by inherent biases in the valuation approaches used to derive these estimates.

The value of ecosystems can be expressed in monetary terms or in terms of QALYs, as the share of our well-being that we are willing to sacrifice to protect the ecosystems. While this trade-off should preferably be done by choice modelling, only one such study was found at the level of abstraction that allows us to relate BAHYs to QALYs or monetary units. Stressing the necessity for such studies, we resort to suggest a temporary proxy value of 1 400 EUR/BAHY (or 52 BAHY/QALY), with an uncertainty range of 350 to 3 500 EUR/BAHY.

The practical consequences of the above-described monetarisation values has been investigated by combining them with the midpoint impact categories of two recent LCIA methods, thus providing a new LCIA method with the option of expressing results in both midpoints and an optional choice between QALY and monetary units as endpoint. From our application of the new method to different case studies, it is noteworthy that resource impacts obtain less emphasis than in previous LCIA methods, while impacts on ecosystems obtain more importance. This shows the significance of being able to express impacts on resources and ecosystems in the same units as impacts on human well-being.

([1]) Invited manuscript for Ecological Economics. Submitted 27 January 2007. Revised 19 November 2007.

15.2 Introduction

The applicability of cost-benefit assessments (CBA) is affected by the high uncertainty in relation to monetarisation of environmental impacts (e.g. Turner et al., 2004). CBA has also been criticised for incompleteness (e.g. Bos & Vleugel, 2005). Recent developments in Life Cycle Impact Assessment (LCIA) offer a basis for reducing both the uncertainty in monetarisation and the completeness problem.

The UNEP/SETAC framework for LCIA (Jolliet et al., 2004) operates with three overall safeguard subjects (humans, ecosystems, resources), fundamentally parallel to the 'people, planet, profit' distinction for sustainability made popular by WBCSD (World Business Council for Sustainable Development). Since the three safeguard subjects are logically exhaustive (any item must be human or non-human, biotic or abiotic, intrinsic or instrumental), they provide a complete framework for all imaginable values for protection.

The LCIA method 'Ecoindicator99' (Goedkoop & Spriensma, 2001) was the first to provide impact pathways that ended at a physical score for each of the three safeguard subjects humans, ecosystems, and resources. In the following, we shall elaborate a novel procedure for monetarising these physical scores. The procedure is aimed at reducing some of the previously encountered uncertainty and incompleteness in monetarising environmental impacts.

15.3 Defining the damage categories

For each of the three safeguard subjects (humans, ecosystems, resources) we specify a common measurement unit. Our measurement units are slightly adjusted compared to the units for impact or damage applied in the 'Ecoindicator99' (Goedkoop & Spriensma, 2001).

Within the safeguard subject 'humans' we define the damage category as 'human well-being' with the measurement unit Quality Adjusted Life Year (QALY). This measurement unit is identical to the Disability Adjusted Life Year (DALY) used by Goedkoop & Spriensma (2001), except for a reversal of signs (QALY measures a positive state, while DALY measures damage, i.e. 1 QALY = -1 DALY) and that while the disability adjustment is limited to health issues, the quality adjustment may also apply to social aspects, such as infringements on autonomy and equal opportunities (Weidema 2006). The reversal of sign is of little consequence and is mainly made to ensure consistency with the traditional definitions and usage of EUR and QALYs in previous work in the field of CBA and health economics. The most critical value choice in the DALY and QALY concepts is that all individuals are given equal weight irrespective of socioeconomic status.

Within the safeguard subject 'ecosystems', we define the damage category as 'biodiversity' with the measurement unit Biodiversity Adjusted Hectare Year (BAHY). This measurement unit is identical to the PDF*m^2*years used by Goedkoop & Spriensma (2001), where PDF is an abbreviation of Potentially Disappeared Fraction of species, except for the more convenient size of the unit (1 hectare = 10 000 m^2), a reversal of signs (BAHY measures a positive state, while PDF*m^2*years measure damage, i.e. 1 BAHY = -10 000 PDF*m^2*years), and that we specify the damage relative to the number of <u>endemic</u> species under natural conditions. It would be possible to define the damage category wider, e.g. in terms of Quality Adjusted Hectare Years, to capture also other aspects of ecosystems quality than just biodiversity. However, in practice, the currently available operational measures of ecosystems quality are all related to biodiversity, so a more encompassing name would be presumptuous. The most critical value choice in the PDF*m^2*years and BAHY concepts is that all species are given equal weight.

In giving equal weight to all individuals or species, the QALY and BAHY concepts have a level of abstraction that may complicate their application for valuation in, for example, choice modelling, but at the same time gives them the level of neutrality required to reduce arbitrariness and uncertainty from specific contexts.

For the safeguard subject 'resources' we define the damage category as 'resource productivity' measured as the future economic output in monetary units. In practice, we use 'EUR_{2003}', i.e. the currency unit euro at its average value in 2003. The conversion factor to USD is close to one for this year. To measure the impact of mineral resource use on future generations, Goedkoop & Spriensma (2001) used 'MJ' additional energy required for future extraction as a result of current dissipation. However, dissipation of mineral resources is only a small part of the non-internalised impacts on resources caused by current human activities. Examples of much more important impacts are the lost production due to health impacts on the labour resource, and the lost agricultural output resulting from photochemical ozone impacts; see Table 15.3, the notes to Table 15.4, and Weidema et al. (2007b). Since all losses of resource productivity, including the additional efforts needed for future extraction of mineral resources can be measured directly as the economic production value foregone, it appears reasonable to use a monetary rather than a physical unit as the common unit of measurement.

15.4 Using the budget constraint to obtain the monetary value of a QALY

In this section, the monetary value of a QALY is derived from the overall budget constraint, and the resulting value compared to and discussed in the context of the results of other methods to derive the monetary value of a QALY.

The budget constraint, i.e. the fact that the average annual income is the maximum that an average person can pay for an additional life year, provides an upper limit for the monetary value of a QALY. Since a QALY by definition is a life-year lived at full well-being, the budget constraint can be determined as the potential average annual income at full well-being, which is equal to the potential annual economic production per capita.

Since a QALY conceptually covers all aspects of human well-being that one would be willing to pay for, all income will on average be spent on total production to maintain full well-being, providing that there is no long-term change in capital stock. Therefore, the potential average annual income at full well-being also provides a lower limit for the monetary value of a QALY. We may thus conclude that there is a conceptual equivalence between the monetary value of a QALY and the potential economic production per capita ([2]).

([2]) One reviewer suggested that part of the potential might be realised in the form of increased leisure, in which case the potential economic production would be reduced. However, from a valuation perspective, the value (shadow price) of such a potential change in leisure preference should also be included in the value of a QALY, which means that this would remain unaffected.

The potential annual economic production per capita is calculated by Weidema (2005), arriving at a value equivalent to 74 000 EUR_{2003}. An uncertainty range of 62 000-84000 EUR_{2003} per QALY can be estimated. The potential annual economic production is calculated by taking the current Gross Economic Product (GEP) ([3]) of USA (39 500 EUR_{2003}) as a starting point – noting that USA has the highest GEP in the World, when ignoring a few untypical economies based heavily on oil or banking – and multiplying it by the factor 1.87 derived in Table 15.1 to take into account current impacts from unemployment and underemployment, health impacts, trade barriers and missing education. Except for these impacts, the current difference between the USA and the global average is assumed to be due to lacking physical and social infrastructure. There are no other apparent reasons that the GEP of countries should differ.

Table 15.1: Ideal economic production relative to the current economic production of the USA.

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

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

It is interesting to note that the willingness-to-pay studies performed as part of the recent update of the ExternE methodology (Markandya et al. 2004) result in a recommended undiscounted value of a life year of 74 627 EUR, i.e. practically the same as our value of 74 000 EUR_{2003} for the budget constraint calculated as the potential annual economic production. While this is purely a coincidence, it confirms that our value is in a reasonable range. The ExternE update is characterised by specifically seeking to address small risk increases from involuntary exposure and is therefore regarded as more relevant for policy analysis of pollution impacts than previous studies.

Other estimates in the value-of-life literature – 42 in total – were reviewed by Hirth et al. (2000), who found a strong dependency on the method applied. They found median values of 25 000 USD_{1997} (approx. 23 000 EUR_{2003}) per QALY for studies using the human capital approach, and 160 000 USD_{1997} (approximately 150 000 EUR_{2003}) per QALY for contingent valuation studies, when using a 3 % discounting rate (corresponding to 90 000 EUR_{2003} / QALY without discounting). For studies using revealed preferences, the median values were 93 000 USD_{1997} for non-occupational safety and 428 000 for job-risk studies, both calculated for a 3 % discount rate. The human capital approach only includes the value of the earning ability under current

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

economic conditions. It is therefore expected that the values derived by this method are lower than our value derived from the potential economic production, which takes into account the full earning ability when current barriers for full economic production are removed. The higher values of the willingness-to-pay studies can be explained by the difficulties to adequately account for the budget constraint in this type of studies. Also, studies based on contingent valuation and revealed preferences most often assess voluntary risk or risk aversion behaviour, and the derived values can best be interpreted as the individuals' evaluation of impacts that occur to themselves, rather than a value that is applicable for general policy purposes, see also the discussion in Markandya et al. (2004). It is obvious that some people in some situations will be willing and able to pay more than the global average budget constraint for an extra QALY, and that other people will be less able (and possibly also less willing). However, the global nature of the QALY concept, i.e. that a QALY has the same value for all individuals, supports that the value of a QALY should be derived from the global average budget constraint, rather than the budget constraints and valuations of specific individuals.

The willingness-to-pay estimate of the ExternE project of EUR 74 627 is provided with an uncertainty estimate of EUR 27 000 – EUR 225 000 (Markandya et al. 2004). An important cause of the uncertainty found in willingness-to-pay studies is that the results vary with the geographical location, population and context. While this may indeed provide relevant values for a specific context, it is less useful for deriving values for an abstract concept like QALYs, which is intended to be globally applicable for aggregation of impacts in many different contexts. When applying the overall budget constraint, the uncertainty on the monetary value of a QALY is reduced to a range of 62 000 – 84 000 EUR_{2003} per QALY, derived by applying low and high estimates for each of the constituting components in Table 15.1, where the range for the overall factor is calculated to 1.57-2.12.

15.5 Expressing ecosystem impacts in terms of human well-being

Lack of willingness-to-pay values for general impacts on ecosystems has been a major obstacle for the inclusion of such impacts in CBAs. For example, in his otherwise thorough study of the economic impacts of climate change, Tol (2002) resorts to applying a fixed value based on 'warm glow', i.e. a value that does not change with increasing impacts on ecosystems.

As a more solid alternative to willingness-to-pay estimates, choice modelling is gaining ground as a way to value ecosystem impacts (see the survey of Hanley et al., 2001, for examples for specific ecosystems and geographically limited ecosystem services, and Itsubo et al. 2004 for an example using species-extinction). Choice modelling is already widely used for the health state evaluations that allow us to aggregate different impacts in human well-being into the common unit of QALYs, see e.g. Hofstetter (1998), Goedkoop & Spriensma (2001), Jolliet et al. (2003). For a decision-maker that accepts the use of choice modelling to obtain health state evaluations for environmental midpoint indicators, it should also be acceptable to apply choice modelling as a procedure to obtain an expression of the severity of ecosystem impacts in terms of QALYs or monetary units. For example, it could be investigated what sacrifice in terms of disabilities or lost life years would be acceptable to protect a certain ecosystem area, or put in other terms: what reduction in life quality is regarded as equivalent to the loss of a certain ecosystem area.

However, although choice modelling has been applied to specific ecosystems and geographically limited ecosystem services, only one study (Itsubo et al. 2004) has yet been made at the level of abstraction that allows us to obtain a measure of BAHYs in terms of QALYs or monetary units. In anticipation of, and to stimulate the execution of, more such choice modelling studies, we resort to suggest a proxy value.

In an initial attempt (Weidema et al. 2006), we started our derivation of a proxy value by comparing the global terrestrial species-area of 13 10^9 hectare*years to the global human population 6.2 10^9 people, noting that if these were given an equal weighting in a valuation, this would result in an 'exchange rate' of 2.1 hectare*years per human life-year. We could also express this as 2.1 BAHY/QALY, since QALYs represent human life years at full well-being, corresponding to BAHYs representing hectare*years of nature in its unaffected state. To adjust for the fact that ecosystem biodiversity and humans are not in practice given equal weight, we suggested that the protection target of 10 % of the global ecosystems called for in the Convention on Biological Diversity could be compared to an ultimate protection target for human well-being of 100 %, giving us an adjustment factor of 10 for the 'exchange rate' between biodiversity and human well-being. The resulting value of 21 BAHY/QALY or 0.048 QALY/BAHY could be understood to mean that the full protection of an ecosystem of 21 hectares (210 000 m^2) for one year has the same value as an extra life-year at full health for one person.

To express BAHYs in monetary units, we used the above-derived value of a QALY, thus arriving at a value of 3 500 EUR/BAHY (74 000 EUR/QALY divided by 21 BAHY/QALY). Noting that the current human activities engage approximately 50 % of natural ecosystems (37 % of NPP according to Imhoff et al., 2004; 13 % as a central estimate of the global species-area lost due to climate change, following Thomas et al., 2004), the adjustment factor of 10 implies that this the overall damage would be equivalent to a 5 % loss of all potential QALYs or 0.05 QALY/person-year. In monetary terms, this may be interpreted as 5 % of the potential income or 3 700 EUR/person-year.

However, we note that the value of 3 500 EUR/BAHY is one order of magnitude larger than the range of EUR 63 – EUR 350 per ha of ecosystem protected suggested by the ExternE study (Bickel & Friedrich 2005) for acidification and eutrophication. This value was derived from what they call a 'second-best' method of revealed preferences from political negotiations.

The choice modelling study of Itsubo et al. (2004) used the normalised environmental impacts of an average Japanese (0.54 million DALYs versus the extinction of one species annually) and obtained monetarised values of 9.7 million JPY/DALY (approx. 68 000 EUR/DALY) and 4.8E+12 JPY (34 E+9 EUR) per species-extinction, or a weighting factor of 1.2 on the normalised values. Itsubo et al. (2003) present values for different land uses (e.g. road construction) with an average impact of 4E-8 species-extinctions per ha. With a corresponding value of 0.88 BAHY/ha for similar land uses, we obtain 4.5 E-8 species-extinctions/BAHY or 1 500 EUR/BAHY.

Finally, we note that the current environmental protection expenditures in developed countries are at 1-2 % of GDP. Although this is <u>not</u> the same as the <u>marginal</u> willingness-to-pay for additional ecosystem protection above the current level, it may – together with the above observations – indicate that our initial suggested value of 5 % of the potential income for ecosystem protection is likely to be an upper bound. Using the ExternE values as a lower bound, we have an order of magnitude range for the ‘correct’ value of a BAHY, i.e. it is likely to be anywhere between 350 and 3 500 EUR/BAHY. In the following exemplary applications, we used a value corresponding to valuing the current global ecosystem impacts at 2 % of the potential income, i.e. 1 500 EUR/person*year or 1 400 EUR/BAHY, stressing that this is purely a proxy value in order to show the importance of being able to express ecosystem damage in monetary terms, waiting to be replaced by better estimates to be made directly by choice modelling.

15.6 On the additivity of the three damage categories

That the impacts on the three damage categories are additive is demonstrated by the following reasoning: in a world without externalities, the GDP would be 74 000 EUR/capita, as shown in the previous section. This would also be the money we could spend. The potential value of production and consumption is thus 2*74 000 EUR = 148 000 EUR/capita. We loose some of our production value of 74 000 EUR because of impacts on production, i.e. our present education-corrected global GDP is not 74 000 EUR/capita but only 10 300 EUR (the relationship 10 300/74 000 EUR = 14 %, which could be called our current production efficiency). Furthermore, we loose some of our potential 74 000 EUR worth of life quality because of impacts on human health and ecosystems. Let us assume that these impacts can be calculated on a global scale to be approx. 51 000 EUR (not all of them attributable to products or even to human activities). The ratio (74 000-51 000)/74 000 = 31 % could be called our consumption efficiency. The overall production and consumption efficiency is therefore currently (14+31)/200 = 23 %, which indeed shows ample room for improvements.

15.7 Choosing QALYs or monetary units to express overall impact?

The relationship between QALYs and potential human economic production is an equivalence, i.e. while the potential annual per capita economic production of 74 000 EUR_{2003} puts a limit on our ability and willingness to pay for a QALY, an additional life year at full well-being (i.e. an additional one QALY) provides us an additional potential economic production of 74 000 EUR_{2003}. In comparison to other monetarisation methods, our procedure of using the budget constraint has the advantage that the resulting values can be interpreted as a proportion of the potential human economic production, and thus directly comparable to the impacts on resource productivity (production output lost due to health impacts, lost agricultural output resulting from pollution, etc.). We may therefore use this equivalence to translate the impacts on economic production into QALYs, rather than translating QALYs into monetary units. This has the advantage that QALYs express an (ultimate) intrinsic value, while monetary units merely represent instrumental values. This option may be of particular interest when the endpoint results are to be communicated to persons that do not favour monetarisation. Another advantage is that impacts expressed in QALYs are relatively stable over time, while monetary units are more volatile.

15.8 Findings from applying the endpoint modelling to case studies

The practical consequences of the above-described endpoint modelling has been investigated by integrating it with the midpoint impact categories of two recent LCIA methods (EDIP2003 and IMPACT 2002+), extended to the damage categories of 'Ecoindicator99', thus providing a new LCIA method (named Stepwise2006) with the option of expressing results in both midpoints and an optional choice between QALY and monetary units as endpoint. The full documentation of Stepwise2006 is available via www.lca-net.com/projects/stepwise_ia/ or in Weidema et al. (2007b).

We have applied the Stepwise2006 method at different stages of development to a number of case studies (Weidema & Wesnaes 2006, 2007, Weidema et al. 2006, 2007a). From these experiences, we find that the impact category for natural resource use is now assigned less importance than in previous LCIA methods, as a result of expressing impacts on resource productivity in comparable monetary units rather than in physical values. Conversely, impacts on ecosystems now obtain higher importance in the results than in previous LCIA methods. This shows the importance of being able to express impacts on the three safeguard subjects in the same units.

15.9 Estimating the relative importance of environmental impact categories

Table 15.2 provides a summary of the characterisation factors for each of the midpoint impact categories of the Stepwise2006 LCIA method. As mentioned, the relationship between QALY and EUR, as applied in Table 15.2, is an equivalence. Thus, all values in EUR in Table 15.2 may as well be expressed in QALY, by using the conversion ratio 1.35E-5 QALY/EUR.

Table 15.2 **Summary of damage (endpoint) characterisation factors for the midpoint impact categories in Stepwise2006 v.1.2, and aggregation of all impacts into a single-score indicator measured in EUR.**

Impact category	Unit of characterised values at midpoint	Impact on ecosystems		Impacts on human well-being		Impacts on resource productivity	All impacts aggregated
		BAHY / characterised unit at midpoint [1]	EUR / characterised unit at midpoint [2]	QALY / characterised unit at midpoint [3]	EUR / characterised unit at midpoint [4]	EUR / characterised unit at midpoint [5]	EUR / characterised unit at midpoint [6]
Acidification	m^2 UES	5.5E-02	7.7E-03				7.7E-03
Ecotoxicity, aquatic	kg-eq. TEG wat.	5.0E-09	7.1E-06				7.1E-06
Ecotoxicity, terrest.	kg-eq. TEG soil	7.9E-07	1.1E-03				1.1E-03
Eutrophication, aq.	kg NO_3-eq.	7.2E-5	0.10				0.10
Eutrophication, terr.	m^2 UES	8.9E-06	1.3E-02				1.3E-2
Global warming	kg CO_2-eq.	5.8E-05	8.2E-2	2.1E-08	1.6E-03	-3.7E-04	8.3E-2
Human toxicity	kg C_2H_3Cl-eq.			2.8E-06	0.21	6.4E-02	0.27
Injuries, road/work	fatal injuries-eq.			43	3.2E+06	9.9E+05	4.2E+06
Ionizing radiation	Bq C-14-eq.			2.1E-10	1.6E-05	4.8E-06	2.0E-05
Mineral extraction	MJ extra					4.0E-03	4.0E-03
Nature occupation	m^2 arable land	8.8E-05	0.12				0.12
Ozone layer deplet.	kg CFC-11-eq.			1.1E-03	78	24	100
Ph.chem. ozone – veg	m^2*ppm*hours	6.59E-08	9.3E-05			2.8E-04	3.7E-04
Respirat. inorganics	kg PM2.5-eq.			7.0E-04	52	16	68
Respiratory organics	pers*ppm*hours			2.6E-06	0.20	6.1E-02	0.26

1. Characterisation factors from Weidema et al. (2007b), based on Goedkoop & Spriensma (2001), Potting & Hauschild (2005), Humbert et al. (2005), and Thomas et al. (2004).
2. Values from column [1] multiplied by 1 400 EUR/BAHY.
3. Characterisation factors from Weidema et al. (2007b), based on Tol (2002), Humbert et al. (2005), Mathers et al. (2004), Hofstetter (1998), and Hauschild & Potting (2005).
4. Values from column [3] multiplied by 74 000 EUR/QALY.
5. Characterisation factors from Weidema et al. (2007b), based on Tol (2002), Miller et al. (2000), and Goedkoop & Spriensma (2001).
6. Sum of values from column [2], [4] and [5].

The relative importance of the different environmental impacts is shown in Table 15.3, obtained by multiplying the monetarised values for each midpoint impact category in Table 15.2 by their respective normalisation references, which express the total midpoint impacts in Europe in year 1995.

This shows that four impact categories (global warming, injuries, nature occupation, and respiratory inorganics) make up 92 % of the total monetarised impacts from the included impact categories. Important impact categories that are not yet included in the Stepwise2006 method are invasive alien species and traffic noise.

Table 15.3: Normalisation references and total impacts in EUR per person in Europe for year 1995.

Impact category	Unit of characterised values	Normalization reference (Europe 1995)	Source	Total impact per person
		Characterised unit / person-year		EUR / year
Acidification	m^2 UES	2 200	[1]	17
Ecotoxicity, aquatic	kg-eq. TEG water	1 360 000	[2]	10
Ecotoxicity, terrestrial	kg-eq. TEG soil	2 350	[2]	2.6
Eutrophication, aquatic	kg NO_3-eq.	77	[4]	7.9
Eutrophication, terrestrial	m^2 UES	2 100	[1]	26
Global warming	kg CO_2-eq.	10 600	[3]	880
Human toxicity	kg C_2H_3Cl-eq.	219	[2]	59
Injuries, road or work	fatal injuries-eq.	0.000142	[4]	590
Ionizing radiation	Bq C-14-eq.	533 000	[2]	11
Mineral extraction	MJ extra	292	[2]	1.2
Nature occupation	m^2 arable land	3 140	[2, 4]	390
Ozone layer depletion	kg CFC-11-eq.	0.204	[2]	21
Photochemical ozone — Vegetation	m^2*ppm*hours	140 000	[1]	52
Respiratory inorganics	kg PM2.5-eq.	8.8	[2]	590
Respiratory organics	person*ppm*hours	10	[1]	2.6
Sum of the above (rounded)				2 650

1. Hauschild & Potting (2005).
2. Humbert et al. (2005).
3. Gugele et al. (2005).
4. Weidema et al. (2007b).

15.10 Comparison to traditional monetarisation methods

Earlier monetarisation studies have primarily obtained their values from stated preferences (via contingent valuation or choice modelling) or from revealed preferences. The method applied for the Stepwise method (i.e. obtaining the monetary values directly via the overall budget constraint in terms of the potential human economic production), requires that all impacts are first expressed relative to an overall concept of well-being (such as QALYs), which has only recently become possible, especially as a result of the pioneering work of Goodkoep & Spriensma (2001) in developing the Ecoindicator99 method.

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

In our approach, we have sought to reduce the need for separate monetarisation exercises, by suggesting that all human health and ecosystem impacts be measured by one indicator (QALY) and by then assessing the monetary value of this overall indicator. This does not eliminate uncertainty and the need for assumptions, but it does increase the consistency and transparency of the assumptions made.

An overview of monetarisation studies has recently been provided by Turner et al. (2004). Table 15.4 shows the values of Stepwise2006 compared to the values in the summary table of Turner et al. (2004) translated to EUR, using the exchange rate of 1.45 EUR/GBP.

Table 15.4: Comparison of the Stepwise monetary endpoint values to the summary values in Turner et al. (2004). All values in EUR_{2003} per Mg emission.

Substance	Previous studies as reviewed by Turner et al.	Stepwise2006	Comment
CO_2	1 – 55	83	[1]
CO	2	450	[2]
NOx	2 200 – 42 000	9 700	[3]
PM2.5	2 900 – 435 000	68 000	[4]
PM10	2 600 – 330 000	36 000	[4]
SO_2	2 500 – 23 000	5 400	[5]
VOC	725 – 2 200	250	[6]

1. 98 % of our value is ecosystem impact, while the previous studies have generally not quantified the ecosystem impact. Thus, the value of previous studies mainly captures health and resource productivity impacts.
2. The value of 450 EUR is composed of health impacts (70 EUR), agricultural impact (171 EUR), ecosystem impact (57 EUR), global warming impact (130 EUR), and human resource impacts (21 EUR). The two EUR values of previous studies is probably due to insufficient physical modelling rather than differences in monetarisation.
3. The value of 9 700 EUR is composed of health impacts (6 600 EUR), human resource impacts (2 100 EUR), ecosystem impacts (600 EUR), and agricultural impact via photochemical ozone (400 EUR). The values of previous studies are dominated by the health impact, but also include small contributions from fertilization effect (a benefit of 200 EUR) and effects on buildings (300 EUR), both of which we have ignored in Stepwise2006, due to their relatively low importance.
4. The PM values are for health impacts, except for a small contribution of 200 EUR / Mg PM10 for impacts on buildings, which we have ignored in this study, due to the low importance.
5. The value of 5 400 EUR is composed of health impacts (4 000 EUR), human resource impacts (1 250 EUR), and ecosystem impact (150 EUR). The values from previous studies are also dominated by the health impact, with 370-962 EUR impacts on buildings, 14 EUR impact on agriculture, and 8 EUR impact on ecosystems.
6. The value of 250 EUR is composed of health impacts (20 EUR) incl. human resource impacts, agricultural impact (170 EUR) and ecosystem impacts (60 EUR), while the previous studies have generally not quantified the ecosystem impact. Turner et al. (2004) also gives recommended values for the UK based on a study by Watkiss et al. (2004), where the values for health impacts are 4-600 EUR and the value for agricultural impact is 380 EUR. These more recent values are thus closer to our estimates.

Important impacts are left un-monetarised in previous studies (see e.g. Bos & Vleugel 2005). Most studies do not provide consistent damage values for ecosystem impacts. This is especially problematic for global warming, where the ecosystem impact is dominating, but also the important impact from land use is left unquantified in most studies.

15.11 Outlook

Expressing all environmental impacts in QALYs and using the budget constraint to establish an equivalence between QALYs and monetary units, opens up for seamless integration of new impact categories, e.g. for social and economic impacts, which may also be expressed in either QALYs or monetary units (Weidema 2006), thus allowing for continuous increases in completeness of LCIA-based CBAs.

As any endpoint method will include a number of assumptions that may be controversial, a wider scientific and stakeholder review procedure is needed to approach consensus on the procedures and values used. This is especially relevant for the application of the overall budget constraint to derive the value of a QALY, a procedure which has not been attempted earlier. Also, to replace our proxy value for the severity of ecosystem impacts, a proper choice modelling study should be performed, preferably in conjunction with a larger study to obtain consistent values for a larger number of issues, and including calibration to the values derived in the 'global burden of disease' study (Mathers et al. 2004).

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Abstract

The report is a scientific contribution to the European Commission's Integrated Product Policy framework, which seeks to minimise the environmental degradation caused throughout the life cycle of products.

This report first presents a systematic overview of the life cycle of meat and dairy products and their environmental impacts, covering the full food chain. It goes on to provide a comprehensive analysis of the improvement options that allow reducing the environmental impacts throughout the life cycle. Finally, the report assesses the different options regarding their feasibility as well as their potential environmental and socioeconomic benefits and costs.

The report shows that meat and dairy products contribute on average 24% to the environmental impacts from the total final consumption in EU-27, while constituting only 6% of the economic value. The main improvement options were identified in agricultural production, in food management by households (avoidance of food wastage), and related to power savings. When all environmental improvement potentials are taken together, the aggregated environmental impacts (external costs) of meat and dairy products may be reduced by about 20%.

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