Working Party on Economic and Environmental Policy Integration

INSTRUMENTS AND TECHNOLOGIES FOR CLIMATE CHANGE POLICY

ANNEX 2 - SECTORAL RESULTS

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<th>Description</th>
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<tbody>
<tr>
<td>ABS</td>
<td>Acrylonitrile Butadiene Styrene</td>
</tr>
<tr>
<td>Al₂O₃</td>
<td>Alumina</td>
</tr>
<tr>
<td>APME</td>
<td>Association of Plastics Manufacturers in Europe</td>
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<tr>
<td>BF</td>
<td>Blast Furnace</td>
</tr>
<tr>
<td>BOF</td>
<td>Basic Oxygen Furnace</td>
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<tr>
<td>BTX</td>
<td>Benzene Toluene Xylene</td>
</tr>
<tr>
<td>CCF</td>
<td>Cyclone Converter Furnace</td>
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<tr>
<td>CHP</td>
<td>Combined Heat and Power production</td>
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<tr>
<td>CFCs</td>
<td>Chlorofluorocarbons</td>
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<tr>
<td>CF₄</td>
<td>Carbontetrafluoride</td>
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<tr>
<td>C₂F₆</td>
<td>Carbonhexafluoride</td>
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<tr>
<td>CH₄</td>
<td>Methane</td>
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<tr>
<td>CO₂</td>
<td>Carbon Dioxide</td>
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<tr>
<td>CaO</td>
<td>Calcium Oxide</td>
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<tr>
<td>CaCO₃</td>
<td>Calcium Carbonate (limestone)</td>
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<tr>
<td>DFE</td>
<td>Design For the Environment</td>
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<tr>
<td>DMT</td>
<td>DiMethylTerephthalate</td>
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<tr>
<td>DRI</td>
<td>Direct Reduced Iron</td>
</tr>
<tr>
<td>EAF</td>
<td>Electric Arc Furnace</td>
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<tr>
<td>ECN</td>
<td>Netherlands Energy Research Foundation</td>
</tr>
<tr>
<td>EUR</td>
<td>Euro, formerly known as European Currency Unit (ECU)</td>
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<tr>
<td>EU</td>
<td>European Union</td>
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<tr>
<td>EU-12</td>
<td>Former European Union of 12 countries: Belgium, Denmark, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Portugal, Spain, UK</td>
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<tr>
<td>EU-15</td>
<td>Current European Union of 15 countries: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Portugal, Spain, Sweden, UK</td>
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<tr>
<td>FAO</td>
<td>Food and Agriculture Organisation of the United Nations</td>
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<tr>
<td>GDP</td>
<td>Gross Domestic Product</td>
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<tr>
<td>GER</td>
<td>Gross Energy Requirement</td>
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<td>GFPs</td>
<td>GlassFiber reinforced Plastics</td>
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<tr>
<td>GHG</td>
<td>GreenHouse Gas</td>
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<tr>
<td>GWP</td>
<td>Global Warming Potential</td>
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<tr>
<td>HCFCs</td>
<td>HydroChloroFluoroCarbons</td>
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<td>HFCs</td>
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<tr>
<td>HNO₃</td>
<td>Nitric acid</td>
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<tr>
<td>HTU</td>
<td>HydroThermal Upgrading</td>
</tr>
<tr>
<td>HYL</td>
<td>Hojalata y Lamina SA</td>
</tr>
<tr>
<td>IGCC</td>
<td>Integrated Coal Gasifier Combined Cycle</td>
</tr>
<tr>
<td>IEA</td>
<td>International Energy Agency</td>
</tr>
<tr>
<td>IISI</td>
<td>International Iron and Steel Institute</td>
</tr>
<tr>
<td>IMAGE</td>
<td>Integrated Model to Assess the Greenhouse Effect</td>
</tr>
<tr>
<td>IPCC</td>
<td>Intergovernmental Panel on Climate Change</td>
</tr>
<tr>
<td>K₂O</td>
<td>Kalium Oxide</td>
</tr>
<tr>
<td>LPG</td>
<td>Liquefied Petroleum Gas</td>
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**ACRONYMS/FORMULAE (continued)**

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<th>Acronym</th>
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<td>LCA</td>
<td>Life Cycle Analysis</td>
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<tr>
<td>LHV</td>
<td>Lower Heating Value</td>
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<tr>
<td>MARKAL</td>
<td>MARKet ALLocation</td>
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<tr>
<td>MEK</td>
<td>MethylEthylKetone</td>
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<tr>
<td>MFA</td>
<td>Material Flow Analysis</td>
</tr>
<tr>
<td>MIT</td>
<td>Massachusetts Institute of Technology</td>
</tr>
<tr>
<td>MSW</td>
<td>Municipal Solid Waste</td>
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<tr>
<td>MTBE</td>
<td>MethylTertiaryButylEther</td>
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<td>MTO</td>
<td>Methanol-To-Olefins</td>
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<td>MUSS</td>
<td>Markal User’s Support System</td>
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<td>NaO</td>
<td>Sodium Oxide</td>
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<td>N-Benzene</td>
<td>Nitro-Benzene</td>
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<td>NEI</td>
<td>Netherlands Economic Institute</td>
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<tr>
<td>NGL</td>
<td>Natural Gas Liquids</td>
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<td>NH$_3$</td>
<td>Ammonia</td>
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<td>Nitrous Oxide</td>
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<td>NO$_x$</td>
<td>Nitrogen Oxides</td>
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<tr>
<td>o-Xylene</td>
<td>Ortho Xylene</td>
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<tr>
<td>OECD</td>
<td>Organisation for Economic Cooperation and Development</td>
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<tr>
<td>PA</td>
<td>PolyAmide</td>
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<td>PBD</td>
<td>PolyButaDiene</td>
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<td>PFCs</td>
<td>PerFluoroCarbons</td>
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<tr>
<td>Phth. Anhydride</td>
<td>Phthalic Anhydride</td>
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<tr>
<td>PHB/PHV</td>
<td>PolyHydroxyButyrate/PolyHydroxyValerate</td>
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<tr>
<td>PP</td>
<td>PolyPropylene</td>
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<tr>
<td>ppmv</td>
<td>parts per million, volume based</td>
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<td>PS</td>
<td>PolyStyrene</td>
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<td>PUR</td>
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<td>PhotoVoltaics</td>
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<td>PolyVinylChloride</td>
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<td>p-Xylene</td>
<td>Para-Xylene</td>
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<td>RIVM</td>
<td>National Institute of Public Health and the Environment</td>
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<td>RME</td>
<td>RapeseedMethylEther</td>
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<tr>
<td>SBR</td>
<td>Styrene Butadiene Rubber</td>
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<tr>
<td>SF$_6$</td>
<td>Sulphurhexafluoride</td>
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<td>SO$_2$</td>
<td>Sulphurdioxide</td>
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<tr>
<td>STAG</td>
<td>Natural gas fired STeam And Gas combined cycle</td>
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<tr>
<td>TO</td>
<td>Technological improvement option</td>
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<tr>
<td>TPA</td>
<td>Terephthalic Acid</td>
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<tr>
<td>UF-Resins</td>
<td>Urea-Formaldehyde Resins</td>
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<tr>
<td>ULSAB</td>
<td>Ultra Light Steel Automobile Body</td>
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<tr>
<td>UN-ECE</td>
<td>United Nations Economic Commission for Europe</td>
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<tr>
<td>UNEP</td>
<td>United Nations Environment Programme</td>
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<tr>
<td>UNFCCC</td>
<td>United Nations Framework Convention on Climate Change</td>
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<tr>
<td>VCM</td>
<td>VinylChloride Monomer</td>
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<tr>
<td>UNITS</td>
<td>SI units</td>
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<tr>
<td><strong>Energy</strong></td>
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<td>MJ</td>
<td>MegaJoule</td>
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<td>GJ</td>
<td>GigaJoule</td>
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<td>PJ</td>
<td>PetaJoule</td>
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<td>EJ</td>
<td>ExaJoule</td>
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<tr>
<td><strong>Weight</strong></td>
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<td>kt</td>
<td>Kilotonne</td>
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<td>Gt</td>
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<tr>
<td><strong>Distance</strong></td>
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<td>km</td>
<td>kilometre</td>
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<tr>
<td><strong>Volume</strong></td>
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<td>Gl</td>
<td>Gigalitre</td>
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<tr>
<td><strong>Pressure</strong></td>
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<tr>
<td>Mpa</td>
<td>MegaPascal</td>
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FOREWORD

The present study has been initiated as part of OECD’s ‘subsidies, taxes and resource pricing’ programme. This volume is Annex 2 of the report which consists of four volumes:

**Main report**: Summary and policy recommendations.

**Annex 1**: Model characteristics and results on the aggregate energy and materials systems level, including intercontinental trade and the aggregated Western European energy and materials systems.

**Annex 2**: Sectoral results, including ancillary benefits of GHG emission reductions in terms of waste management.

**Annex 3**: Database characterisation, detailed description of technical options and the modelling of the Ideal-Policy and Policy-continuation scenarios.

This report has been written by D.J. Gielen, Netherlands Energy Research Foundation, Petten, and by J.H.M. Pieters, Ministry of Housing, Spatial Planning and Environment, and Research Centre for Economic Policy (OCFEB), Erasmus University, Rotterdam.
1. INTRODUCTION

The present study is initiated in the ‘subsidies, taxes and resource pricing’ programme and funded by the Dutch Ministry of Housing, Spatial Planning and the Environment. The four topics that are covered by the OECD sustainable development programme are integrated into this project. This volume is Annex 2 of the reporting which consists of four volumes:

Main report: policy makers summary
Annex 1: model characteristics and results on the aggregate energy and materials systems level
Annex 2: sectoral results
Annex 3: database characterisation and modelling of regulation and pricing instruments

The discussion of results and model characteristics is split into four levels: the intercontinental trade level (Annex 1), the aggregated Western European energy and materials systems level (Annex 1), the level of individual sectors in the economy (this volume) and the technology characterisation (Annex 3).

In Section 2, the results for emission mitigation strategies are discussed on the level of the whole economy in order to draw the background for the sectoral results. The contribution of individual energy and materials strategies is analysed in more detail. Next, the following sectors are discussed in more detail in separate chapters:

1. Iron and steel production
2. Aluminium production
3. Petrochemical production
4. Building materials production
5. The transportation sector
6. Agriculture and forestry
7. Waste

These sectors have been selected because of their high relevance from a GHG emission point of view and their relevance from a materials flow point of view.

The discussion focuses primarily on the ideal policy framework (I) model runs for the integrated energy and materials system (see Annex 1 for an explanation of I and the policy continuation framework C). Some important differences between I and C are elaborated with regard to material flows. These results illustrate the importance of the proper selection of GHG policy instruments for the materials sector, an issue which has received little attention as of yet.

The following questions will be answered:

- which technology shifts can be expected in the selected sectors?
• which change in production volume will occur for materials and for agricultural products because of GHG policies?

• what are the consequences of GHG policies for other policy areas, such as waste policies and agricultural policies?

• is the optimal strategy mix sector specific?

**Box 1. Interpretation of results**

The analyses in this report are based on the results of the MARKAL MATTER model. They represent an approach where the whole energy and materials system ‘from cradle to grave’ is optimised at once. As a consequence, the results represent the outcome of general policy approaches that choose the optimal set of options from a Western European regional perspective. The results do not represent the results of separate sectoral policies which can be applied independently. Note that the emission penalties which are shown in this paper do not represent the emission reduction costs for individual sectors. In many cases, it is more cost-effective to reduce emissions through technological change instead of paying the penalties. Earlier analyses have shown that for the economy as a whole, the average GHG emission mitigation costs represent approximately one fifth of the marginal emission mitigation costs [1]. If industries get tradable emission permits and they can reduce their emissions at a cost price below the value of the permit (i.e. below the marginal costs for emission mitigation for the whole economy), they may even make a profit out of GHG emission policies.
2. GENERAL RESULTS

In order to understand the results on a sectoral level it is necessary to understand the general changes in the system for increasing GHG penalties. The interactions between sectors are considered in the optimisation, and the analysis shows that these interactions matter. For example the emissions in electricity production are reduced significantly. These general results are discussed in this Section. The following issues are discussed:

2.1 General results for GHG emission reduction for the whole economy
2.2 General analysis of the contribution of materials options
2.3 Changes in electricity production

2.1 General results for GHG emission reduction

Figure 2.1 shows the trend for the West European gross domestic product in real terms (GDP) and the trend in GHG emissions for the base case and the emission reduction cases. GDP increases 3.5-fold while the GHG emissions increase moderately in the base case (by 30%) and decrease significantly in the emission reduction cases. In the base case the decoupling of GDP and GHG emissions is 1.7% per year (the gap between the GDP index and the BC index) This gap is more important for future GHG emissions than the GHG emission reduction that can be achieved through emission penalties. One should note that 1.7% decoupling is well above historical decoupling trends, implying a significant structural change.

The gap between GDP and GHG emissions can be attributed to a physical product service demand that does not grow as fast as GDP. The demand for energy and product services is in this model estimated in a bottom-up approach. Growth in the existing product demand categories is estimated on the basis of logistic penetration curves in time. Consequently, the growth in all demand categories levels off in time. New product service demand categories have not been identified. However, the additional income must be spent. The question is whether additional consumption will really be irrelevant from a GHG emission point of view. Where this additional income will be spent is not clear at present. On the supply side, the trend towards a service economy is obvious [2]. Trends such as the rapid development of telecommunication and computer technology may result in new products that are very GHG-intensive. However, whether these trends can really result in such a strong decoupling over a period of decades should be analysed in more detail on the basis of other socio-economic analysis methods. The problem extends beyond the scope of this study, but it is of major importance for GHG emission reduction. Less decoupling implies more reliance on technological solutions, a costly but feasible solution (see e.g. [3])
Figure 2.1. GDP and GHG emissions in the base case BC and for increasing GHG penalties

The emission reduction for the integrated energy and materials system (E+M) is shown in Figure 2.2. This result suggests that a factor 4 emission reduction is feasible, even in a situation with a growing demand. However, the penalty level where such a goal can be achieved exceeds the penalty levels that are currently being considered.

Note the high contribution of CO₂ to the total GHG emissions in the base case. Autonomous developments reduce the emissions of other GHGs (compared to 1990 levels, e.g. because of closure of the German and UK coal mines and because of the ban on landfilling of organic waste reduce methane emissions). Increasing CO₂ emissions are linked to a growth in demand in sectors such as transportation. As a consequence of the dominance of CO₂ emissions in total GHG emissions, a significant GHG emission reduction is largely a CO₂ emission reduction problem.
The contribution of individual strategies to the total GHG emission reduction is shown in Figure 2.3. The figure shows that the bulk of the emission reduction can be attributed to energy and materials efficiency (including all types of improvements on the materials demand side and demand reductions, up to 1900 Mt CO₂ equivalents). CO₂ storage and the non-CO₂ GHG contribute together approx. 1100 Mt CO₂ equivalents. Fuel switches (from coal to natural gas, nuclear and renewables) contribute up to 640 Mt CO₂ equivalents.
2.2 General analysis of the contribution of materials options

The contribution of the demand reduction to the emission reduction is shown in Figure 2.4. This result is based on the comparison of a set of calculations for the MATTER2.0 model (with fixed demand) and MATTER3.0 model calculations (with elastic demands). The difference in emission reduction can be attributed to the change in demand. The results show that this contribution increases gradually to 550 Mt in the 200 EUR/t case. This represents a contribution of 15% to the total emission reduction at this penalty level.

Figure 2.4. Aggregated emission reduction including elastic demands (MATTER3.0) and excluding elastic demands (MATTER2.0), 2030

![Graph showing emission reduction versus penalty](image)

The contribution of individual materials strategies is shown in Figure 2.5 (based on MATTER2.0 model runs). These emission reductions were calculated with the integrated energy and materials system model. Certain groups of options have been set to zero in these calculations, and the results have been compared to the model results for the full set of emission mitigation options. For example: in order to calculate the contribution of the strategy category ‘end-of-pipe’, the options for CH₄ recovery from landfills, N₂O emission mitigation in the chemical industry, and CO₂ removal and underground storage in industry (excluding refineries and electricity production) have been removed from the model. The difference in emission reduction in the calculations with and without these options is the emission reduction which is attributed to this strategy. The end-of-pipe strategy proves to be important. The contribution of biomass feedstocks for the petrochemical industry is also significant (both fermentation and pyrolysis processes). ‘Other resources substitution’ in Figure 2.5 refers to the use of slag materials and Pozzolan for the production of cement and the use of tropical hardwood substitutes. Some materials substitution occurs (the substitution of steel for aluminium in the transport sector and the substitution of concrete for wood in the building sector). Materials substitution proves to be important because it induces emission reductions in the energy system (e.g. lightweight vehicles reduce transportation fuel demand). Moreover, the calculations show an additional 35 Mt CO₂ equivalent of carbon storage in wood products in the 200 ECU/t penalty case, compared to the base case (mainly in the building sector). On the waste management side, waste from renewable materials (wood, paper) is increasingly used for energy recovery,
while waste from synthetic organic materials (plastics) is increasingly recycled. Waste management strategies for GHG emission mitigation become especially important in the 200 ECU/t penalty case. Sensitivity analysis shows that the model would opt for disposal of these materials as a carbon storage strategy if no restrictions were added. However such strategies conflict with pending European regulations (a ban on disposal of combustible waste). Improved materials quality (the category ‘materials efficiency’ in Figure 2.5) is introduced in the 200 ECU/t penalty case. One must consider that the model representation of materials efficiency is incomplete. The strategy has only been considered for steel and for concrete because of the scarcity of reliable data for other materials. Some case studies indicate similar efficiency potentials for other materials. As a consequence, its potential is probably more significant than Figure 2.5 suggests.

**Figure 2.5. Contribution of individual materials strategies, 2030 [1]**

![Graph showing contributions of individual materials strategies](image)

Strategies such as higher industrial energy efficiency and emission reduction in electricity production for materials production have not been considered in Figure 2.5 (but are included in the emission reduction in the materials system as defined in Annex 1).

In the framework of this study there was considerable debate regarding the definition of the ‘materials system’. Some experts may argue that the definition of ‘materials system’ in this study is artificial because it encompasses materials production, which is generally considered in energy policy making and thus already considered for to GHG policies.

In order to meet this critique, the relevance of optimisation of materials demand has been considered separately in order to show that the contribution of these truly new strategies is substantial. The model has been run without all emission reduction options on the materials demand side (Figure 2.6). The results can be compared to the results in Figure 2.5. Figure 2.6 shows a maximum contribution of the materials demand side of 700 Mt CO₂ equivalents, compared to a maximum contribution of 800 Mt in Figure 2.5 (excluding the end of pipe strategy in Figure 2.5, which is partially materials supply related). If these emission reductions are compared to the emission reductions in Annex 1, Section 2.3 (770 Mt, which encompass both materials production and materials demand), the conclusion could be drawn that the emission reductions on the materials production side have little additional impact. However it is obvious that this conclusion is not correct, given the significant changes on the materials production side (see
Sections 3-9). The problem with this analysis is the impact of materials options on the emission reductions in the energy system. For example by-products from the use of biomass feedstocks (biomass crackers etc.) are used for energy production. Light weight materials in the transportation sector reduce the transportation fuel demand. In this analysis, these emission reductions are completely allocated to the materials demand side. If these emission reductions are subtracted from the 800 Mt emission reduction (given the current GHG intensity of the energy system), the contribution of the materials supply side is in the range of 200-300 Mt CO₂ equivalents. This is again an underestimation because the GHG intensity of the transportation system and the building sector change simultaneously. In conclusion, many allocation schemes can be derived, resulting in a emission reduction contribution of the materials system between 200 and 1200 Mt CO₂ equivalents.

**Figure 2.6. Contribution of technological emission reduction options on the materials demand side, 2030, ideal policy framework I**

The preceding analysis shows the importance of an integrated energy and materials systems modelling approach. The interaction of emission reduction strategies - both in the materials system between the supply and demand side and between the energy system and the materials system - decrease the effectiveness of individual strategies considerably.

Significant differences occur in the changing materials consumption between the I and C policy framework (Figures 2.7 and 2.8). The significant increase in wood consumption is related to more wood use for buildings (coupled to less cement use), for feedstock substitution and for paper. The increased wood use for paper production is related to the increased energy recovery from waste paper (instead of recycling, see Figure 9.2). It is interesting to note the generally less significant changes in demand in the regulation model runs. This can be explained by the fact that industry is excluded from GHG policies in this scenario. As a consequence there is no demand reduction (because product prices remain constant) and no materials substitution.
2.3 Electricity

Electricity production will be significantly affected by GHG penalties. Approximately 23% of all GHG emissions can be attributed to the electricity production (see Annex 1). Changes in electricity production are also of key importance for the assessment of materials strategies, for example the GHG intensity of aluminium depends largely on the GHG intensity of the electricity which is applied in the production process. Important strategies for emission reduction in electricity production include:

- increased efficiency (including combined heat and power generation CHP)
- fossil fuel switch (from coal to gas)
- CO$_2$ removal and underground storage
- nuclear energy
- renewables (biomass, PhotoVoltaics, wind, hydro)

Electricity demand is not constant during the day and during the year. The time of peak demand depends on the local conditions. In northern countries without air conditioning, the peak demand occurs during a winter morning. In southern countries with air conditioning, it occurs during a summer day. The difference between peak demand and minimum demand may be a factor 4-5. The variable demand is a problem because of the electricity storage problem. Contrary to other commodities, electricity can only be stored at considerable expense (e.g. water pumped into high water reservoirs or batteries). Renewables such as photovoltaics (PV) have their peak supply during a specific period of day and/or year. Depending on the supply pattern, e.g. renewables may or may not fit into the existing electricity demand pattern. These problems can complicate the introduction of e.g. renewables significantly and should be considered in the analysis for proper assessment. The MARKAL approach is especially suited for the analysis of this.
type of interactions, because supply and demand are matched in the analysis on a seasonal and diurnal basis.

**Figure 2.7. The impact of GHG penalties on materials consumption, 2030, policy continuation framework C**

A GHG emission penalty will result in significant price increase for electricity, with detrimental effects on the electricity consumption. However many technological options exist to reduce these emissions. GHG emission reduction can even be positive for total electricity production because electricity is a CO$_2$-free energy carrier (at least during in its use stage). Electrification can be a way to reduce GHG emissions in sectors which consist of a large number of small energy users such as residential heating, heating of industrial kilns and for the transportation sector (electric vehicles). These strategies are also considered in the analysis.

**Model structure**

The electricity production technologies that have been considered in this study are discussed in [4]. They cover the full range of existing and future supply options. For supply options with regionally different characteristics such as PV and biomass, a split has been applied for North, Middle and Southern Europe, characterised by different technology parameters and/or costs. Regarding supply options such as nuclear energy, the main feasibility problem is outside the GHG policy area. In the model calculations, it has been assumed that the minimum nuclear capacity is half the current capacity, while the maximum contribution reflects the current capacity [4].

**Results**

Figure 2.8 shows the electricity production mix in the pricing model calculations. The results show an almost constant electricity production for increasing GHG emission penalties. Reduced consumption due to increased efficiency of equipment is balanced by increasing demand due to new applications (e.g. in the transportation sector, see Section 7). The results show a significant fuel shift. In the 20 EUR/t penalty case, natural gas increases significantly compared to the base case (at the expense of coal). However at higher penalty levels, gas is substituted by nuclear energy and by renewables: hydro,
wind, PV and biomass. Especially wind and hydro are important renewable options, while the use of biomass is remarkably low (see also Section 8). The remaining coal and gas fired power plants are equipped with CO\textsubscript{2} removal and underground storage. As a consequence, GHG emissions from electricity production are reduced to less than a tenth of the emissions in the base case at penalty levels from 50 EUR/t upward. This change has also important consequences for the emissions related to materials production. For example materials whose production requires large amounts of electricity, such as aluminium and chlorine, become less problematic from a GHG emission point of view. As a consequence, the GHG impact of materials substitution is affected for these materials.

Figure 2.8. Electricity production, ideal policy framework I, 2030

The future of electricity production is influenced by the assumptions regarding the future of nuclear energy, which is to a lesser extent a cost issue. A sensitivity analysis was done without limitation on nuclear supply. The result is shown in Figure 2.9.
Comparison of Figures 2.8 and 2.9 shows that total electricity demand is approximately 2 EJ higher in the 200 EUR/t penalty case. This can be attributed to an increased electrification (more electric vehicles). Nuclear power plants substitute renewables. Total GHG emissions are hardly affected in the 100 EUR/t penalty case.
MARKAL MATTER model results show a very significant emission reduction in electricity production if GHG emission mitigation strategies are introduced (see Figure 2.10). CHP, new high efficiency gas fired power plants, renewables such as biomass and wind energy, CO₂ removal and underground storage and nuclear energy will reduce the emissions in electricity production by a factor 10, if permit prices from 100 EUR/t upward are introduced. An average annual emission reduction of 3% to 7% is feasible for the next 40 years from the current Western European average level of 0.1 GJ/t (which is comparable to the emission for a gas based modern power plant without CHP). Such changes must be considered if the GHG impact of changes in electricity consumption is analysed.

2.4 Conclusions

The MATTER modelling results show a moderate emission growth in the base case. Some other - econometric - simulation models, show higher emission growth rates for Western Europe. The difference can be explained by the tendency of bottom-up models, like MARKAL, to underestimate growth (e.g. because future new demand sectors are not adequately modelled) and the tendency to overestimate efficiency gains (all options that are cost-effective are included in the base case, while these options may not be introduced because of non-financial barriers like knowledge gaps etc.). On the other hand, econometric models may underestimate the rate of technological change and underestimate the autonomous dematerialisation trends in energy supply due to the rapid introduction of natural gas, increasing conversion efficiencies and rapid structural change, e.g. because of a rapid shift to an information society.
Regarding emission reductions, the non-CO\(_2\) GHGs and CO\(_2\) removal and underground storage contribute up to 30\% of the total emission reduction. Approximately 20\% of the emission reduction is caused by fuel switches (from coal to gas, nuclear and renewables). The remaining 50\% is caused by the increased efficiency in energy and materials use.

MATTER modelling results suggest that a significant emission reduction potential exists with regard to the materials demand side (up to 500-700 Mt CO\(_2\) equivalents, depending on the calculation method). This potential constitutes up to 20\% of the total GHG emission reductions. The relevance of the materials system is even higher if the emission reductions on the materials supply side are also allocated up to the materials system: 800 Mt CO\(_2\) equivalents, 35\% of the total GHG emissions [1].

Technological progress is one of the key driving forces that determine the system configuration. Technology will change significantly within a time horizon of half a century. In the framework of the energy and materials system, major changes can be expected in power generation (gasification technology, new technology for renewables), in the transport sector (biofuels, fuel cells, electric vehicles) and for materials production (e.g. smelting reduction processes for iron production, new paper drying technology, biomaterials). For recycling, cheap materials separation technology (in order to prevent downcycling, e.g. for metals) and improved recycling technology for plastics pose a challenge. These technological changes will affect the systems configuration and must be considered in long-term strategy development.

End-of-pipe technology for CO\(_2\), CH\(_4\) and N\(_2\)O and industrial fuel substitution do already receive a great deal of attention in many European countries. More attention should be devoted to feedstock substitution in the petrochemical industry, to improved materials quality, materials substitution/product re-design, and to the GHG consequences of future waste management technologies.

The results show that it is important to consider the interactions between improvement options in the optimisation. This includes interactions between energy and materials options and interactions between different types of materials options. If such interactions are neglected, long term emission reduction potentials are over-estimated.

Greenhouse gas emission mitigation will have much more impact on materials production and waste handling than on materials consumption. This hypothesis has been approved by this case study. Aluminium production and wood production can significantly benefit from GHG emission reduction. The results indicate a moderate decline in cement production. For other materials, the impact on materials consumption is limited. Generally speaking the impacts on materials flows are more significant in case a techno-economic optimal mix of emission reduction options is selected.
3. IRON AND STEEL

3.1 Introduction

The iron and steel industry will be affected by GHG emission reduction policies because it is a major source of CO₂ emissions. A number of studies have addressed this issue before (e.g. [5,6,7]). These studies have analysed new steel production technologies and emission reduction options within the iron and steel industry. In this study, the emission reduction in the iron and steel industry is analysed within the framework of the changing energy and materials system configuration. This includes the whole environmental life cycle ‘from cradle to grave’. Not only steel production, but also steel consumption and steel scrap management is considered in the optimisation.

3.2 Model structure

Figure 3.1 shows the model structure for steel production. Three coal based technologies are considered for production of liquid iron: the blast furnace, Corex and CCF (the Cyclone Converter Furnace). The blast furnace represents the current technology for iron production. Corex is a recent technology that is nowadays rapidly introduced on a number of locations throughout the world (but not yet in Western Europe). Its main advantage is the use of (low quality) coal instead of coke, so the coke production step can be avoided. CCF is still in a development stage¹. Successful development will require at least another decade. The CCF process uses coal and ore fines, so both coke making and ore agglomeration can be avoided. CCF is a representative of a family of smelting reduction processes that are currently developed such as e.g. Finex. Both Corex and CCF are more energy efficient technologies than the current blast furnace process.

Increased coal injection is considered as autonomous development for blast furnaces. Charcoal and waste plastic are considered as substitutes for (limited amounts of) coal in iron making. Two steel making routes are considered: BOF (the Basic Oxygen Furnace) and EAF (the Electric Arc Furnace). DRI (Direct Reduced Iron, a solid iron product that is produced with natural gas) can substitute scrap in EAF steel making. The HyL-III process has been considered for DRI production (HyL is an acronym for the Hojalata y Lamina SA company which has developed the process). Three steel qualities are considered. A very pure quality (which is required for cold rolled sheet with less than 3 mm thickness and for coated sheet), pure (e.g. sheet > 3mm, wire and tubes) and conventional (e.g. bars, sections). The more scrap is added, the lower the steel quality. Scrap based EAF can only be used for the pure and the conventional quality. The very pure steel quality can only be produced from virgin material: from iron or from DRI. A mixture of DRI and steel scrap input into EAF steel making results in a pure steel quality. Higher quality grades can be used for finished products with lower quality requirements, the other way around is not possible (a ‘quality cascade’). A number of finished iron and steel products are modelled:

1 Investment plans for CCF have been postponed by Hoogovens, the company who has developed the technology. This postponement is possibly related to the recent merger with British Steel into a new company called Corus.
Figure 3.1. Materials model structure for the iron and steel industry, detailed for steel production

Conventional quality
- hot rolled sections
- hot rolled concrete reinforcement bars

Pure quality
- hot rolled heavy plate
- alloyed steel
• wire rod
  Very pure quality
  • hot rolled coil
  • cold rolled coils
  • cold rolled sheet, galvanised or tinplated

The life cycle of 40 (groups of) products is modelled that contain steel. The steel products include bulk products such as cars, steel frames for buildings, steel reinforcements, machinery and other capital equipment, and household equipment. Competing materials such as concrete (for buildings), aluminium (for cars, for trucks and for beverage cans), plastics (for cars and household equipment), and glass (for food containers) are considered.

3.3 Results

Figure 3.2 shows a split of iron and steel production into different process technologies and into ore based steel or scrap based steel. The figure shows the impact of an increasing GHG penalty in 2030. The figure shows in the base case a significant shift from the current blast furnace technology to Corex and CCF based iron production. In the case with a penalty of 50 EUR/t CO₂, more DRI and more CCF based iron production is introduced as substitute for Corex based steel production. At this penalty level, CO₂ removal and underground storage is introduced for all processes. At the penalty level of 100 EUR/t CO₂, the production of iron from CCF increases at the expense of DRI based steel production. The reason for this shift is the possibility to produce significant amounts of virtually CO₂-free natural gas substitute and CO₂-free heat.

Figure 3.2. Changing iron and steel production, 2030, due to CO₂ penalty
The CO₂ impact of smelting reduction technologies is significantly affected by the electricity production technology mix. For example Corex produces more than 15 GJ of gas and steam by-products per tonne iron. These products will generally be used for large-scale power generation. In such a situation power production in the reference energy system becomes virtually CO₂-free at higher emission reduction penalties. Power production based on the energy by-products of steel production avoids no CO₂ emissions in electricity production.

In conclusion, the iron and steel industry will be affected significantly by CO₂ policies. The Western European steel industry emits currently 7% of the total CO₂ emissions in this region. MARKAL calculations suggest that blast furnaces will be replaced by new technologies in the first half of the next century. Smelting reduction technologies are more energy efficient and more cost-effective than blast furnaces, but they result in significant amounts of energy by-products. The other option is EAF technology, based on scrap and/or DRI. The model calculation show in the base case without CO₂ emission penalties a stabilised total steel production. EAF based steel production increases further. The blast furnace is replaced by smelting reduction processes (Corex and CCF). The Corex process gains in the base case a dominant position because of the positive value of its large amount of energy by-products. The increasing steel quality poses in the model calculation a constraint for EAF steel production; the quality issue deserves more attention.

The steel stock in the economy is increasing. This limits the potential for a recycling strategy. Moreover, significant amounts of semi-finished steel, steel products, used steel products and steel scrap are exported. Exported steel is not available for recycling in Western Europe unless costly import strategies are considered. Such effects must be considered for development of regional emission abatement strategies.

The regional energy and materials systems engineering approach provides additional information for GHG policy making and industrial investment decisions regarding the interaction of changing steel production technologies and the changing energy and materials systems configuration. Such changes must be considered in the case of the iron and steel industry. Especially the changing reference electricity production is relevant for the analysis of the GHG emission impacts of the industry. In the case of the iron and steel industry, new smelting reduction technologies and EAF technologies will have a significant impact on the future electricity balance of Western Europe. As a consequence, technological changes in the iron and steel industry must also be considered in the assessment of the future electricity production.
4. ALUMINIUM

4.1 Introduction

Aluminium is still in the growth phase of the economic product life cycle. Aluminium demand is increasing, mainly due to substitution of other materials in the transportation sector and other light-weight applications. Because of the increasing demand, there is little incentive to invest into new technology and rapid product innovation. The aluminium production is characterised by an oligopolistic structure with a global market. The high product price (5-10 times the steel price) allows for global aluminium trade. The primary aluminium import into Western Europe is likely to increase in the next decades. Competing primary aluminium producers are primarily located in countries with low electricity prices. However cheap electricity is only part of the picture. For example the availability of cheap hydropower in developing countries seems not sufficient to initiate large investments in primary smelters (at least on the short term). The stability of these favourable conditions on the long term is equally important. As electricity markets become more developed and liberated, both in developed and developing regions, extremely low or high electricity prices cannot last. Within Western Europe, the substantial hydropower potential on Iceland may pose an interesting alternative.

The aluminium market is a global market. Foreign producers can enter the European market, the reverse seems unlikely given the high European production costs. Because the aluminium demand is still rapidly increasing, both within Europe and abroad, new aluminium capacity can be included in the market without long term price impacts. Aluminium is gradually gaining a position in the traditional steel markets. Its light weight, corrosion resistance, processing possibilities and easy recycling will strengthen its position on the long run.

The main energy use is related to the electrochemical conversion of alumina (Al₂O₃) to aluminium. The Hall-Héroult electrolysis process, which is currently applied for this conversion, is a mature technology, but gradual improvements of both productivity and environmental performance are still possible. A new production technology for aluminium from bauxite seems currently not in sight. The only ongoing research in primary aluminium production is aimed at inert anodes and inert cathodes. Both options allow a re-design of the electrolysis cells which can increase the energy efficiency, reduce GHG emissions and reduce production costs. The technological feasibility of such developments is still unclear. Improved technology can be applied in order to increase scrap recovery rates, especially with regard to aluminium in municipal solid waste (MSW). This will result in increased recycling.

4.2 Model structure

GHG emissions in the life cycle of aluminium can be split into CO₂ emissions that are related to the energy use and process emissions (PFCs) that are related to process interruptions in primary aluminium smelting. The PFC emissions are related to the so-called ‘anode effect’. If the concentration of alumina in the bath becomes too low, CF₄ and C₂F₆ are generated, both gases with a high global warming potential (GWP). The total emissions of PFCs for modern smelters (applying to the best available technology) are

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2 Inert cathodes have recently been proven on a pilot plant scale. For inert anodes, development has stagnated during the last two decades, despite significant research efforts.
equivalent to 0.4 t CO₂ equivalents/t aluminium [8]. For older smelters, these emissions may be as high as 10-20 t/t aluminium [9]: emissions for cells with point feeders are one order of magnitude lower than the emissions for side-fed cells. The average European smelter emits 2-3 t CO₂ equivalents of PFCs per tonne aluminium [10]. The CO₂ emissions for the anode production and anode use and for the alumina production amount together to approximately 3.5 t CO₂/t. The emissions that are related to the electricity consumption depend highly on the fuel mix for electricity production. For coal fired power plants, they can amount up to 12 t CO₂/t aluminium. For hydropower plants, they are close to zero. The European average electricity mix results in an emission of approximately 5 t CO₂/t aluminium for electricity production. Based on these data, total emissions for aluminium production in Western Europe (approx. 4 Mt per year) are estimated to be in the range of 40-50 Mt CO₂ equivalents, i.e. about 1% of the total Western European GHG emissions. Note that Western Europe is a net importer of primary aluminium, resulting in emissions abroad for European consumption.

The model structure for aluminium is shown in Figure 4.1. Iceland has been considered as separate production site because of its significant CO₂-free hydroelectricity and geothermal electricity potential at a remote location. Aluminium which is produced on Iceland and which is transported to other parts of Europe can be considered as a kind of “electricity storage medium”. A number of aluminium product applications are separately modelled, together with competing materials. The bipolar cell design (based on inert anodes and inert cathodes) represents a very attractive technology both from an environmental point of view and from a cost point of view. The electricity consumption is lower than for existing smelters, and the use of carbon anodes is avoided. Moreover, the PFC emissions are reduced to zero simultaneously. However the ultimate technical feasibility of this technology is still not proven, despite 25 years of research regarding the inert anodes, the key technological problem for this process route. This is a typical example of a “breakthrough” technology that will be applied on a large scale if it is commercially proven, either with or without GHG penalties because the aluminium production costs are reduced by 15-20%.

**Figure 4.1. Model structure for aluminium**
4.3 Results

Aluminium production increases from in the base case from 5 Mt per year in the first half of the 1990s to 14 Mt in 2030, almost a threefold increase. No major shift towards imports has been assumed. This is probably a valid assumption for recycling; for primary aluminium, increasing imports are part of an existing trend (that may be accelerated by GHG policies, see Annexe 1). However rising demand in other regions can reverse this trend. A global model is required for proper assessment of primary aluminium markets. Figure 4.2 indicates that total consumption increases moderately. The results indicate a switch from the current Hall-Heroult smelters with carbon anodes to smelters with inert anodes and, in the case of penalties of 100 EUR/t and higher, a relocation of aluminium smelters to Iceland. Recycling increases in the base case from currently 30% to 55% in 2030. The total amount of recycled aluminium increases with increasing GHG penalties. However due to the increasing aluminium production, the primary aluminium demand increases simultaneously. The figures indicate a limited potential for increased aluminium recovery. The increase of recycling from 55% in the base case to 65% in the case with a 200 EUR/t penalty can be attributed to the increased aluminium use for transportation (with comparatively high recycling rates and a comparatively short life span of 1-2 decades).

Figure 4.2. Changing aluminium production with increasing emission penalties, 2030

The impact of GHG emission reduction on aluminium prices and aluminium use can be significant. However contrary to common sense the aluminium production will be positively affected by GHG emission reduction, notwithstanding the currently high emissions in aluminium production. A combination of emission mitigation in production and significant GHG emission reduction effects further down the product chain cause this effect.

Aluminium recycling rates will probably further increase. Because large amounts of aluminium are stored in long life products and because of the significant losses of (part of the) aluminium that is used in the packaging sector, recycling can cover only a part of the aluminium market in the next two to three decades.
The emissions per tonne of material change significantly with increasing penalty levels. Emissions decrease because upstream emissions decrease (e.g. in electricity production) and because of introduction of new technology with higher energy efficiency and lower process emissions.

Aluminium production benefits in a situation with GHG emission reduction because of additional aluminium demand from the transportation sector, where substitution of steel results in weight reductions which result in significant fuel savings that off-set the remaining emissions in aluminium production.
5. PETROCHEMICALS

5.1 Introduction

The petrochemical industry in western Europe is largely based on the conversion of oil and gas feedstock into synthetic organic materials. The structure of this complex industry is illustrated in Figure 5.1. The most important group of synthetic organic materials is plastics, which are produced through steamcracking of naphtha and of other oil refining fractions.

5.2 Model structure

The modelling of the petrochemical industry is illustrated for polyethylene in Figure 5.2. The model includes a number of emission reduction strategies:

- increased energy efficiency in the petrochemical industry (not illustrated in Figure 5.2)
- feedstock substitution: naphtha, gas oil, LPG, ethane, natural gas, methanol and ethanol
- materials substitution (e.g. in the packaging sector)
- recycling of waste plastics: improved collection schemes and new recycling technology
5.1 Structure of the Western European petrochemical industry, 1994
(figures indicate material and energy flows in tons; exports and one-end arrows refer to exports-imports; acronyms are explained in the glossary)
5.3 Results

The discussion of results is limited to two elements of the petrochemical industry: the production of ethylene (Figure 5.3) and the handling of waste plastics (Figure 5.4). The total production of ethylene is not affected significantly by GHG penalties. However the mix of feedstocks changes. In the 20 EUR/t scenario, gas oil is substituted by LPG. At higher emission penalties, biomass feedstocks substitute fossil fuel feedstocks. One must emphasise that this technology has only been tested on a laboratory scale, so the data are speculative with regard to the feasibility and the characteristics of such a process on a commercial scale. More recent analyses suggest similar biomass potentials, but possibly based on other conversion technologies (see e.g. [3], [11]).
The constant production of ethylene from naphtha can be explained by a lower bound on naphtha cracking in the model input.

The results for plastic waste handling are illustrated in Figure 5.4. The amount of plastic waste decreases at higher emission penalties due to a reduced consumption of plastics. The main change is a substitution of grate firing by increased back-to-feedstock recycling, in this case hydrogenation for production of a naphtha substitute.
Figure 5.4. Waste plastic handling, 2030, with increasing emission penalties
6. BUILDING MATERIALS

6.1 Introduction

Building materials constitute the bulk of all materials applications from a mass point of view. Between 215 and 315 Mt CO$_2$ equivalents can be attributed to this materials category [12]. Cement constitutes the single most important material in this category, between 110 and 125 Mt CO$_2$ equivalents. Only cement will be discussed in more detail.

6.2 Model structure

The model structure for cement is shown in Figure 6.1. The most important cement type is Portland cement, which is produced through calcination of limestone (calcium carbonate) at temperatures above 950 °C and subsequent sintering into cement clinker. During the calcination, the calcium carbonate is dissociated into calcium oxide and CO$_2$. This process is a source of inorganic CO$_2$ (not related to the combustion of fossil fuels). Because of the high temperatures and the chemically basic environment, cement kilns are suited for combustion of waste materials. Residues such as metals are immobilised in the cement. In a coal fired cement kiln, the fossil fuel combustion represents an emission of approximately 0.3 t CO$_2$/t cement, while the calcination represents an additional 0.5 t CO$_2$/t cement. There is already a lot of attention for fuel substitutes in the cement industry, driven by cost savings. These trends reduce fossil fuel related CO$_2$ emissions. However this poses no solution for the reduction of the emissions of inorganic CO$_2$. A number of resource alternatives can reduce the inorganic emissions. Slag residues from industrial processes (blast furnace slag and fly ash from coal fired power plants) can substitute Portland cement clinker. Pozzolana is a natural volcanic fly ash. Finally, synthetic clinkers can be produced on the basis of K$_2$O and Na$_2$O as substitute for the CaO in the clinker, so-called “geopolymeric cement”.

Concrete is the main application for cement. Two main types of concrete products can be discerned: prefab elements which are produced on industrial sites and liquid ready-mix concrete, which is used to fabricate elements on the building site. The model is a simplification of a practice with many different cement quality types. In certain applications high strength concrete can substitute conventional concrete, thus reducing cement demand. This option has been considered in the category “materials quality improvements”.

6.3 Results

Figure 6.2 shows the results for cement. Total cement production decreases by 25% at higher penalty levels. This reduction is partially accounted for by a switch to high strength cement (a materials efficiency strategy). Another factor is materials substitution (mainly substitution by wood materials in the building sector). The mix of production processes is also significantly affected. The availability of fly ash is limited by the use of coal for electricity production. In the 20 EUR/t case, this use is reduced to zero, hence the Portland cement production increases. At higher penalty levels, coal is reintroduced (ICGCC with CO₂ removal). It is assumed that the fly ash from this type of coal fired power plant is also suited for cement production. The changing fly ash availability is typically an example where changes in different sectors must be considered simultaneously for proper assessment of emission reduction potentials. In the 200 EUR/t case, the use of Pozzolans and the use of alternative cement types increases at the expense of Portland cement (“geopolymeric” cement types, based on K₂O and NaO as substitute for CaO in cement).
Figure 6.2. Changing cement production with increasing emission penalties, 2030
7. TRANSPORTATION

7.1 Introduction

Transportation constitutes currently 19% of the western European GHG emissions (see Annex 1). This fraction is still increasing, because the transportation sector shows higher-than-average growth rates. The market can be split into a number of transportation modes:

- passenger cars
- vans
- trucks
- trains
- inland vessels
- sea ships
- air planes

Emission reductions can be achieved through:

- fuel switches
- increased engine efficiencies
- light weight vehicles
- intelligent traffic control systems
- changes in the intermodal split
- reduction of mobility

Only the first three types have been considered in the model, because techno-economic optimisation makes no sense for the latter three types and/or they cannot be modeled properly.

7.2 Model structure

The markets segments have been modelled separately. The passenger car market has been spilt into large cars and heavy cars. For passenger cars, vans and trucks the vehicle production and materials selection has been explicitly modelled. Four types have been considered:

- a standard type
- an optimised steel type
- an optimised aluminium type
- an optimised plastic type
The weight of the designs differs, hence the fuel efficiency is affected (e.g. for passenger cars 1% weight reduction results in 0.5% fuel savings). The lowest weight (hence the highest fuel efficiency) is achieved for the aluminium design.\textsuperscript{3}

The following fuel types have been considered:

- gasoline
- diesel
- liquefied petroleum gas LPG
- kerosene
- methanol
- ethanol
- electricity
- hydrogen

\textbf{Figure 7.1. Model structure for road vehicles, example passenger cars}

\textsuperscript{3} Compared to plastics, aluminium has the advantage of heat resistance and increased strength. Not all car parts can be made from plastics. Combined plastic-aluminium designs have not yet been considered, while some studies suggest they may show lower weight (a plastic car body combined with maximised aluminium use in other car parts).
7.3 Results

Figure 7.2 shows the results for transportation fuel use. The figure shows a significant decline in the total fuel demand. This decline can be explained by a mix of declining transportation demand (including a switch to smaller cars), increased engine efficiencies and a switch to fuels with higher efficiency (such as electricity). Especially the gasoline market declines at higher emission penalties in favour of methanol, ethanol, and electricity. The remaining diesel demand can be attributed to inertia of the system, related to the refinery supply of a complete product package where diesel is one products out of a refinery product mix which cannot be substituted separately without considering the other products. However the results show that significant changes can be expected in the transportation fuel market.

Figure 7.3 shows the changing materials consumption. Note the significant increase of steel use in the base case, compared to the current situation. This growth is caused by the still increasing demand for vehicles. Aluminium use increases at higher emission penalties in favour of steel. Because of the lower weight of aluminium vehicles and the introduction of smaller cars, total materials demand declines.

![Figure 7.2. Transportation fuel use, 2030](image-url)
Figure 7.3. Materials consumption for vehicle production, 2030
8. AGRICULTURE AND FORESTRY

8.1 Introduction

Agriculture and forestry are closely linked to the GHG problem. On one hand, agriculture is an important source of methane CH$_4$ and nitrous oxide N$_2$O emissions (both approximately 200 Mt CO$_2$ equivalents out of a total of 4250 Mt CO$_2$ equivalents [13]). On the other hand, both agriculture and forestry can be used for biomass production. Biomass constitutes a CO$_2$-neutral energy source or a material. The problem whether energy or materials should be produced is currently investigated with the MATTER model in the framework of the EU BRED project (Biomass for greenhouse gas emission REDuction). The discussion in this Section is based on interim results; final results are expected at the end of 1999.

The basic strategies for reduction of CH$_4$ emissions focus on the ruminants: cows and sheep are the main sources. The strategies are based on increased conversion efficiency and new fodder types. For N$_2$O, the main emission source is the use of natural and organic nitrogen fertilisers, which are partially converted into N$_2$O by micro-organisms in the soil. Strategies for the reduction of these emissions are based on the increased efficiency of nitrogen fertiliser gifts. However the situation does significantly differ among European countries. Nitrogen fertiliser gifts are highest in the intensively farmed areas, such as the Netherlands. Fertiliser gifts are 5-10 times higher than the European average, mainly driven by regional manure surpluses. These differences have significant impacts on attractive emission mitigation strategies. At lower N-fertilisation rates, the crop yield is proportional to the fertiliser gift. As a consequence reduced N$_2$O emissions are off-set by increased land requirements.

It has been stated before that biomass can be used for GHG emission reduction. However, its potential is limited by the biomass availability. There is some additional potential for wood recovery from western European forests (current recovery is 70% of the annual regrowth). Western European has reached a status where its agricultural production exceeds food and fodder demand. This is largely accounted for by the steadily increasing agricultural productivity. If this trend continues, 10-20% of the agricultural land (both arable land and pastures) may become available for other purposes. As a consequence, biomass can constitute an important option for GHG emission reduction. Current European policies are aiming for bioenergy, especially large scale electricity production based on gasification technology. Biomaterial strategies have received little attention as of yet from a GHG emission point of view, while transportation fuel activities have been reduced because of the negative and costly experiences during the last decades. The MATTER model has been used in order to investigate in more detail whether this strategy preference is still warranted, given R&D progress during the last decade [3].

8.2 Model structure

Figure 8.1 provides a general model overview of the model structure, showing the close relation between food, energy and material crops. A detailed overview of the energy and materials model structure is provided in [14]. A detailed overview of the food module is provided in [15]. Approximately 10 crop types are covered, split into North, Middle and Southern Europe in order to account for climatological productivity differences. The main markets are covered:
The model covers 120 million hectares agricultural land out of 150 million hectares. The remaining 30 million hectares are assumed to remain constant.

8.3 Results

Figure 8.2 shows the changes in agricultural land use which are induced by GHG penalties. Up to 28 million hectares are used for energy crops in the 200 EUR/t penalty scenario. Consequently there is no physical supply constraint at lower penalty levels. More recent calculations suggest a significant competition from afforestation as a carbon storage strategy, which has not been considered in this model version [3]. This competition may affect biomass availability. Figures 8.3-8.5 show how this biomass is applied. Figure 8.3 shows that materials applications are of similar importance such as energy applications.
at penalty levels up to 100 EUR/t CO₂. Both energy and materials applications show a strong increase compared to the base case. In the 200 EUR/t penalty scenario, energy applications dominate materials applications. This can be attributed to the higher potential market volume for energy applications.

Energy applications are detailed in Figure 8.4. The figure shows especially strong growth in the transportation fuel market, both for methanol and for ethanol. Growth in the electricity and heating market is more moderate. The limited growth in the electricity market can be attributed to the fierce competition from other supply options with low GHG emissions (see Section 2).

Materials applications are detailed in Figure 8.5. The figure shows especially strong growth in the feedstock market and in the pulp production. The growth in the pulp market is related to an increased waste paper incineration/gasification: as long as supply is no problem, this is an option to increase the biomass use in the energy system. The increased use for feedstocks makes sense because biomass is the only renewable carbon source. This feature is important in case of biomass use for feedstocks and for transportation fuels. Other renewable and carbon free energy sources (wind, PV, hydro) can more easily be applied for electricity production.

Figure 8.2. Agricultural land use change with increasing GHG emission penalties, 2030
Figure 8.3. Biomass use for energy and materials, 2030

Figure 8.4. Biomass use for energy production (expressed as primary biomass input), 2030
Figure 8.5. Biomass use for materials production (expressed as primary biomass input), 2030
9. WASTE

9.1 Introduction

The preceding chapters have shown that GHG emission reduction policies will change energy and material flows through the economy significantly. They will also affect the waste flows. The impact of GHG penalties on waste can be split into a number of elements:

- materials substitution results in a changing waste composition;
- increased materials use efficiency results in a reduction of waste volumes;
- changing consumption patterns (‘dematerialisation’, characterised through demand elasticities) results in a reduction of the waste volumes;
- the value of many waste types increases. The cost-effectiveness of waste collection and waste upgrading improves;
- the cost-effectiveness of waste management options changes due to GHG policies. As a consequence other waste treatment technologies are introduced;
- end-of-pipe technologies for landfill gas recovery is introduced on a large scale;
- new waste management problems will emerge. For example underground CO$_2$ storage and oceanic CO$_2$ storage constitute new types of waste disposal, where new regulations must be developed.

These elements will be elaborated in more detail below.

Waste on the energy balance

A first problem for any waste study is the data quality: data for waste flows in Western Europe are not consistent. In [16] the amount of Municipal Solid Waste (MSW) is estimated to be 141 Mt in 1990. 34 Mt waste was incinerated in 1992 according to this source. 83% of the combustion capacity was equipped with energy recovery. The total MSW creation in Western Europe amounted to 225.3 Mt in 1993 according to [17]. 17% of this waste (38 Mt) was incinerated according to this source. The amount that is incinerated is similar according to both sources, but the amount of MSW differs. The difference is probably accounted for by a different definition of MSW. A recent analysis has shown that different national definitions before 1994 are a major cause of inconsistent waste figures [18]. A proper comparison for 1994, based on consistent definitions, has resulted in MSW figures between 460 and 585 kg per person per year for 8 Western European countries, with an average of 537 kg per person per year. Assuming this figure can also be applied to the other countries results in an estimate of 190 Mt MSW in Western Europe for 1994. This figure is in between both earlier estimates. Municipal construction and demolition waste not originating from households is excluded from these figures [18]. Some of this waste category may also be considered MSW in a broader definition.
The energy content of MSW ranges from 9 to 13 GJ per tonne for individual countries. The MSW heating value is largely determined by the plastic content, the paper content, and the amount of kitchen waste. In some countries, separate collection and recycling for these flows has reached high levels. A typical MSW composition for Western Europe is shown in Figure 9.1.

Figure 9.1. Average European MSW composition, 1994 [19]

Another approach for estimation of waste quantities is based on a balance for individual materials:

Materials consumption - losses - stock increases in the product use phase = Waste arising

A combination of both approaches has been used for Table 9.1. Total waste volume figures are higher than previous estimates because for metals and for wood, the bulk of the waste does not end up in MSW. For the other materials, the bulk is accounted for in MSW statistics. Note that the definition of recycling in Table 9.1 differs from the definition in some other statistics in order to generate comparable data for different materials. The definition that is applied in this study for recycling rates is the ratio of waste input into production processes and the waste arising. The waste arising is defined as the total of recycling, incineration, and disposal (this excludes all kinds of ‘losses’ due to oxidation and net exports and it excludes the increasing materials stock in the use phase). For some materials, the amount of waste released in a given year is considerably lower than the materials consumption.

The heating value of the total waste flows is indicated. The total energy content is approximately 2700 PJ. Total current Western European primary energy use is approximately 55,000 PJ per year. Comparison of both figures shows that waste materials can cover 5% of the primary energy consumption. However, recycling makes often more sense, as both feedstock energy and process energy use can be reduced. The selection of the best strategy from an environmental and cost point of view depends on the policy goals and the systems configuration.
Table 9.1. Waste balance for important groups of materials, Western Europe (EU+EFTA), 1993/1994 [20,21,22,23]

<table>
<thead>
<tr>
<th></th>
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<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Paper and board</td>
<td>67</td>
<td>60</td>
<td>15</td>
<td>900</td>
<td>50</td>
<td>10</td>
<td>40</td>
</tr>
<tr>
<td>Kitchen waste/ garden waste</td>
<td>68</td>
<td>68</td>
<td>8</td>
<td>544</td>
<td>10</td>
<td>15</td>
<td>75</td>
</tr>
<tr>
<td>Glass</td>
<td>24</td>
<td>20</td>
<td>-</td>
<td>-</td>
<td>40</td>
<td>12</td>
<td>48</td>
</tr>
<tr>
<td>Metals</td>
<td>175</td>
<td>100</td>
<td>-</td>
<td>-</td>
<td>80</td>
<td>4</td>
<td>16</td>
</tr>
<tr>
<td>Textiles</td>
<td>25</td>
<td>16</td>
<td>35</td>
<td>560</td>
<td>5</td>
<td>16</td>
<td>79</td>
</tr>
<tr>
<td>Wood products</td>
<td>82</td>
<td>34</td>
<td>16</td>
<td>544</td>
<td>15</td>
<td>26</td>
<td>59</td>
</tr>
<tr>
<td>Total</td>
<td>307</td>
<td>2773</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Methane emissions from waste disposal sites represent approximately 200 Mt CO₂ equivalents per year. Apart from methane, waste incineration is an important source of CO₂ from long life petrochemical products (plastics, elastomers, resins, synthetic fibres). This emission was equivalent to 15 Mt CO₂ in 1994, but this figure may rise to more than 150 Mt during the next decades, if all waste is incinerated and the amount of waste petrochemicals doubles.

9.2 Model structure

The model contains about 40 waste material qualities. The characterisation is based on the chemical composition: steel scrap, aluminium scrap, PVC waste etc.. For several materials such as plastics and metals, the quality of waste is further detailed (e.g. clean plastic waste, mixed plastic waste, plastic in municipal solid waste, see Figure 5.2). Each waste treatment process has minimum requirements with regard to the waste quality input. It is possible to change the waste quality with new collection and separation methods (e.g. through disassembly of plastic products or separate collection of waste paper).

The main strategies: disposal, energy recovery and recycling have been modelled. For incineration and recycling and number of technologies have been modelled, depending on the waste type. A detailed discussion of the model input parameters can be found on the internet [24].

9.3 Results

The following elements will be discussed:

- The impact of a GHG penalties via changing materials consumption on waste quantities and waste qualities
- The impact of GHG penalties on the selection of waste management technologies
- The impact of GHG penalties on the introduction of landfill gas recovery
- Emerging waste problems caused by GHG policies

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4 Includes anaerobic digestion (for food/garden waste) and recycling abroad (e.g. for textiles).
5 Both with and without energy recovery.

50
The impact of materials substitution, materials efficiency and dematerialisation

The impact of GHG policies on materials consumption has been discussed in Section 2. Generally speaking, the total consumption declines as prices increase. This decline is especially pronounced in products with a comparatively high CO₂/materials intensity per Euro of product value (e.g. concrete building elements). However for many products, the materials costs constitute only a minor part of the total product life cycle costs.

Table 9.2. Rules of thumb for environmentally conscious design process [25,26,27]

<table>
<thead>
<tr>
<th>Life Cycle Stage</th>
<th>Materials use</th>
<th>Product use</th>
<th>Waste handling</th>
</tr>
</thead>
<tbody>
<tr>
<td>Materials use</td>
<td>Materials selection/saving:</td>
<td>Design product alternatives:</td>
<td>Close materials cycles:</td>
</tr>
<tr>
<td></td>
<td>- reduce weight</td>
<td>service substitutes</td>
<td>- develop recovery scenarios</td>
</tr>
<tr>
<td></td>
<td>- reduce materials diversity</td>
<td>- shared use of product</td>
<td>- develop a disassembly plan</td>
</tr>
<tr>
<td></td>
<td>- use recyclable materials</td>
<td>Design for long product life:</td>
<td>- avoid constructions that cannot be disassembled</td>
</tr>
<tr>
<td></td>
<td>- substitute materials</td>
<td>- quality and reliability</td>
<td>- select cost-effective recycling technologies</td>
</tr>
<tr>
<td></td>
<td>- use renewables</td>
<td>- modular design</td>
<td>- avoid stickers etc.</td>
</tr>
<tr>
<td></td>
<td>- use biodegradable materials</td>
<td>- improvement of weak points</td>
<td>- organise recycling by materials producer</td>
</tr>
<tr>
<td></td>
<td>- reduce non-renewables</td>
<td>- reduce complexity</td>
<td>- use materials coding (system)</td>
</tr>
<tr>
<td></td>
<td>- use waste materials</td>
<td>- easy assembly/disassembly</td>
<td>- design for high quality waste materials</td>
</tr>
<tr>
<td></td>
<td>- reduce spare parts/auxiliaries</td>
<td>- reduced maintenance</td>
<td>- packaging recovery</td>
</tr>
<tr>
<td></td>
<td>- multifunctional products</td>
<td>Design for weight reduction</td>
<td>- reusable containers</td>
</tr>
<tr>
<td></td>
<td>Energy saving:</td>
<td>- increase liquid concentration</td>
<td>- leasing programs</td>
</tr>
<tr>
<td></td>
<td>- select materials with low GER</td>
<td>- design thinner enclosures</td>
<td>- use compatible materials</td>
</tr>
<tr>
<td></td>
<td>- minimise the use of materials with high GER</td>
<td>- reduce product dimensions</td>
<td></td>
</tr>
<tr>
<td></td>
<td>- avoid energy intensive production processes</td>
<td>- use electronic documentation</td>
<td></td>
</tr>
<tr>
<td></td>
<td>- use processes with high energy efficiency</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>- use renewable energy</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>- reduce transportation volume</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

This results in a moderate price increase of products and product services due to the price increase of materials, caused by GHG penalties. For products such as packaging, the effect is well below
10% of the product price in the base case. This price increase is generally not sufficient to cause any major demand reductions.

Increased materials efficiency can be based on a long list of design and engineering strategies (see Table 9.2). For many products, these strategies can result in a 20-50% increased materials efficiency. However, the model is not exhaustive in this respect. The impact of GHG policies and other policies aiming for reduced materials consumption can be much more substantial than the modelling results suggest.

The demand reductions due to increasing prices interact with the demand effects of materials substitution. The results suggest that in certain sectors such as the transportation sector and the building sector, the impact of substitution can be substantial. The results suggest a decreasing cement consumption and an increasing wood and aluminium consumption due to substitution. Such changes in consumption will affect the waste composition. However, especially in the building sector, any change in materials consumption will only result in a change of waste consumption after a period of decades because of the long life span of this type of products. As a consequence, its impact on waste composition in the next decades is limited (but for changes in the processing waste composition in the early stages of the environmental product life cycle).

**Changing waste management practices**

The prices of waste materials are significantly affected by GHG penalties (see Table 9.3). One of the main driving forces for this price change is the GHG emission impact of recycling and energy recovery in comparison with emissions for the competing materials and energy supply options.

<table>
<thead>
<tr>
<th>[EUR/t material]</th>
<th>BC</th>
<th>20 EUR/t</th>
<th>50 EUR/t</th>
<th>100 EUR/t</th>
<th>200 EUR/t</th>
</tr>
</thead>
<tbody>
<tr>
<td>Clean steel scrap</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>118</td>
<td>143</td>
</tr>
<tr>
<td>Clean aluminium scrap</td>
<td>1525</td>
<td>1747</td>
<td>1995</td>
<td>2341</td>
<td>2637</td>
</tr>
<tr>
<td>Waste paper</td>
<td>-47</td>
<td>-69</td>
<td>-106</td>
<td>-98</td>
<td>-71</td>
</tr>
<tr>
<td>Mixed waste plastics</td>
<td>-145</td>
<td>-143</td>
<td>-171</td>
<td>-197</td>
<td>-253</td>
</tr>
<tr>
<td>Nylon waste</td>
<td>196</td>
<td>344</td>
<td>539</td>
<td>738</td>
<td>875</td>
</tr>
</tbody>
</table>

If the GHG emission is reduced by recycling in comparison to the production from natural resources, the value of the waste increases (e.g. for aluminium scrap). If the GHG impact is negative, the value decreases (e.g. in the case of plastic waste incineration). If the waste price increases, it becomes more attractive to collect waste, and separate collection schemes will generally benefit from these developments.

The GHG penalties will also affect the choice between disposal, energy recovery and recycling. Generally speaking, the calculations show a shift for synthetic organic materials from incineration to recycling (see below). For natural organic materials, the calculations show a shift from disposal towards incineration with energy recovery.

The results for waste incineration are illustrated in Figures 9.2 and 9.3. The figures refer to the energy content of the materials that are incinerated. Plastic waste incineration declines significantly, while the incineration of waste paper increases. The latter result depends on the assumptions regarding future land availability for biomass production. In case of ample land availability, it makes sense to incinerate the waste paper. In case of limited biomass availability, it makes sense to increase the efficiency through increase recycling. Especially in the 100 EUR/t penalty case there is some difference between the results for the C and I approach: less waste is incinerated in the I approach.
Figure 9.2. Waste incineration, 2030, with increasing GHG penalties, I framework

![Figure 9.2](image1)

Figure 9.3. Waste incineration, 2030, with increasing GHG penalties, C framework

![Figure 9.3](image2)

The plastic waste handling trends are illustrated in Figure 9.4 and Figure 9.5. The figures show a shift from waste incineration to recycling. The total amount of plastic waste declines in the I framework, while the quantity is almost constant in the C framework. The C framework shows slightly higher recycling rates in the 100 EUR/t penalty case. In the 200 EUR/t penalty case, hydrogenation is selected in the I framework while pyrolysis is selected in the C framework.
In conclusion, the results show that selection of waste management technologies is affected by GHG policies. Not only the emission reduction, but also the selection of GHG policy instruments affects the optimal choice of waste management options. This case illustrates that integrated energy and materials policies should be developed.
End of pipe technology

CH\textsubscript{4} emissions from landfill sites are related to the decomposition of organic waste. The decomposition rate depends on the waste type. Kitchen waste decomposes rapidly. Paper and wood require more time. Plastics and other synthetic organic materials do not decompose within a period of 100 years. Cellulose and hemicellulose decompose, while the decomposition of lignin is more difficult. As a consequence, a changing waste composition (e.g. because of separate collection) will change the landfill gas production. Landfill gas is generally produced for 20-30 years after the closure of the landfill site. As a consequence, the landfill gas recovery potential in 2010 is partially determined by existing disposal sites and partially determined by landfilling policies until 2010. Because of the shift from disposal towards incineration with energy recovery (see above), the relevance of landfill gas recovery decreases. However landfill gas recovery is a very cost-effective emission mitigation strategy for the remaining landfill sites, as the costs are in the range of 0-20 EUR/t CO\textsubscript{2} equivalents.

Emerging waste problems

A significant CO\textsubscript{2} removal and underground storage is another important feature if GHG penalties are applied. Up to 500 Mt of CO\textsubscript{2} is stored per year (see below). Thus quantity is equivalent to 2-3 times the quantity of municipal solid waste. This storage can be considered as a redistribution of environmental problems: from emissions to air to increased waste storage. Its impacts on the environment are still unclear, but may be substantial if insufficient restrictions are imposed. For example the storage of CO\textsubscript{2} in the deep ocean may have serious effects on the ecosystem. Figure 9.6 shows the increasing underground CO\textsubscript{2} storage in the pricing approach which is induced by GHG penalties.

Figure 9.6. Underground CO\textsubscript{2} storage, 2030, with increasing GHG penalties, I framework

Apart from CO\textsubscript{2} storage, the calculations include a shift towards increased use of nuclear energy (see Section 2). The increased use of nuclear energy implies an increased nuclear waste volume.
10. CONCLUSIONS AND RECOMMENDATIONS

10.1 General conclusions

- Improvement options regarding material flows (excluding energy efficiency, fuel switch and end-of-pipe technology in materials production, and emission reductions in electricity production) can be broken down into four separate categories:
  1. Substitute fossil fuel feedstocks by biomass feedstocks
  2. Increase the efficiency of materials use
  3. Recycle/reuse materials and products
  4. Substitute materials

- Emissions in most parts of the energy and materials system will be reduced in a scenario with strong GHG emission reduction policies due to a changing technology mix. The results show that it is necessary to consider interactions between improvement options in the long-term strategy development in order to avoid overestimating emission reduction potentials. Many cases of interaction have been encountered. For example: both the reduction of emissions in materials production, and the reduction of the emissions in electricity production reduce the cost-effectiveness of emission reduction at a later stage in the materials life cycle. The choice for lightweight cars reduces the demand for transportation fuel; the selection of building materials affects the heating and cooling energy demand in the building use stage. Several industries compete for energy recovery from waste. If such interactions are not considered, emission reduction potentials are overestimated.

- Greenhouse gas emission penalties will have a far greater impact on materials production and waste handling than on materials consumption. The impact on materials consumption and the technological change in the materials producing industries is elaborated in Table 10.1.

Table 10.1. The impact of a 200 ECU/t penalty on the materials consumption and production

<table>
<thead>
<tr>
<th>Material</th>
<th>Consumption trend [%]</th>
<th>Cost-effective emission reduction strategy</th>
</tr>
</thead>
<tbody>
<tr>
<td>Iron and steel</td>
<td>-15 - -20</td>
<td>CO₂ removal; recycling; fuel switch; energy efficiency</td>
</tr>
<tr>
<td>Aluminium</td>
<td>+10</td>
<td>PFC emission reduction; recycling; relocation to Iceland; energy efficiency</td>
</tr>
<tr>
<td>Petrochemicals</td>
<td>+10</td>
<td>Biomass feedstocks; recycling</td>
</tr>
<tr>
<td>Cement</td>
<td>-25</td>
<td>Clinker substitution; materials substitution; fuel switch; CO₂ removal</td>
</tr>
<tr>
<td>Sawn wood</td>
<td>+25</td>
<td>Building materials substitution</td>
</tr>
<tr>
<td>Transportation fuels</td>
<td>-30</td>
<td>Biofuels; electricity; energy efficiency</td>
</tr>
<tr>
<td>Electricity</td>
<td>0</td>
<td>Renewables; CO₂ removal; nuclear</td>
</tr>
<tr>
<td>Meat</td>
<td>-10 - -15</td>
<td>Increased production efficiency</td>
</tr>
</tbody>
</table>
The impact of emission reduction penalties on materials consumption is more pronounced in the MED approach than in the 'common' MARKAL approach. The difference accounts for almost 50% of the change in materials demand. The demand for all materials is reduced compared to the common MARKAL calculations without demand elasticities.

Materials strategies can contribute significantly to the reduction of GHG emissions. In the I policy runs, the difference in emission reduction between the stand-alone energy system and the integrated energy and materials system calculations amounts up to 770 Mt CO₂ equivalents in the case of a 100 EUR/t emission penalty. In the C runs, the difference in emission reduction amounts up to 930 Mt CO₂ equivalents. Both values indicate the significance of the materials system for the emission reduction, up to 28-40% of the total GHG emission reduction.

Greenhouse gas emission penalties will have a far greater impact on materials production, the composition of materials use and waste handling than on the overall levels of materials consumption. Increasing GHG penalties will stimulate the use of aluminium in transport and wood in construction and the use of biomass as a substitute for petrochemical feedstocks. The results also show a decline in the production of cement. For other materials, the impact is rather limited.

Model calculations also suggest more recycling and less incineration of plastics. As GHG penalties rise, it becomes increasingly economical to incinerate natural organic materials (including paper) with energy recovery. As a result, paper could become the major part (of a decreasing volume) of waste that will be incinerated (Section 4.4).

Feedstock substitution in the petrochemical industry, improved materials quality, the development of new materials, materials substitution/product re-design, and waste management strategies can result in GHG emission reductions of up to 800 Mt, 25% of the total GHG emission mitigation potential. These options are similar to industrial energy efficiency gains in the sense that many options are characterised by cost savings. The main barriers for the introduction are R&D requirements and insufficient policy attention.

In conclusion, the materials system can make a significant contribution to the total GHG emission reduction. A factor 4 reduction of the emissions from the materials system seems feasible in 2030, compared to the emissions from the materials system in the base case. The modelling of emission reduction options such as product re-design, increased product life, improved materials quality, and changing lifestyles is not exhaustive in this study. This suggests there may be additional emission reduction potential.

Emission reductions in the materials system represent up to 35% in the total emission reduction for a fixed emission penalty. This figure shows that the costs are certainly in line with the costs for emission reduction options in the energy system.

Given the significant emission reduction potential for all types of materials, it is not possible to identify 'good' and 'bad' materials from a GHG emission point of view without consideration if the specific characteristics of the application and without considering future reductions of emission intensity. An integrated dynamic approach is required for proper assessment of long-term materials selection strategies.

GHG emission reduction policies can have significant impact on other policy areas. Only four areas will be discussed in more detail, because they have been touched upon in this report.
R&D policy
industry policy
agricultural policy
waste policy

The analysis in this volume has shown that the selection of technologies is affected significantly by GHG emission policies. GHG emission reduction can be reduced effectively by selection of alternative process routes. However this requires a new definition of research priorities. The emphasis is on biomass feedstocks, recycling technology, Design for the Environment (DfE) strategies. Generally speaking, the main challenges are less of a technological nature but problems of social acceptance and organisational issues.

Regarding industry policy, the shift towards “smart products” with higher value added will be accelerated by a tough GHG policy. Recycling industries deserve more attention, especially efficient waste collection schemes are an area where government can improve the resource basis. Materials efficiency options such as new alloys, composite materials etc. deserve more attention. The total industrial output (in physical units) will decrease. However, the value may actually increase because more expensive technologies are applied in order to reduce GHG emissions. Such a trend may simultaneously increase labour requirements. For example the production of agricultural feedstocks is more labour-intensive than the production of oil feedstocks.

Regarding agricultural policy, the problem of overproduction will be solved by the increasing demand for bioenergy and biomaterials. However this development is not in accordance with the current policy initiatives for extensification of agricultural land use.

Regarding waste policy, the results show for synthetic organic materials a shift away from incineration toward recycling. For natural organic materials, the trend is reversed: from waste paper recycling to waste paper incineration in the emission reduction cases. Significant amounts of CO$_2$ will be stored underground, a new waste policy area.

The study has also contributed new methodological insights

- This study has shown that the MARKAL bottom-up systems engineering approach can also be applied for the analysis of the effectiveness of GHG policy instruments. It could therefore be used as a supplement for econometric models. Significant GHG emission reduction will take decades and because of this long time horizon, consideration must be given to technological development, changing product service demand and materials storage during product life. The analysis of these changes requires a dynamic life cycle approach. The interaction of GHG emission reduction options requires an integrated energy and materials systems approach. Sectoral interactions in the materials life cycle, and interactions between different materials life cycles, require a regional systems approach. These are typically characteristics of the MARKAL approach, but a number of similar systems engineering models can be applied for the same purpose.

- The study has shown how the interactions between policy areas such as agricultural policies, the use of renewable energy, industry policy and GHG policies can be analysed fruitfully with this approach.

10.2 Policy recommendations

Non-energy use of fossil fuels (feedstocks), and the relationship between materials and GHG emissions is not yet considered widely in GHG policies. More attention should be devoted to feedstock substitution in the petrochemical industry, to improved materials quality, the development of new materials, materials substitution/product re-design, and to waste management strategies that can reduce
GHG emissions. The potential for emission reduction in the materials system seems to be of a similar magnitude as the emission reduction potential in the energy system. With regard to their potential savings in cost, these options are similar to industrial energy efficiency gains. Cost-effective efficiency improvement options in the materials system can be considered as hedging options in a situation where the need for substantial GHG emission reduction has not yet been fully accepted. Design for the environment, materials sciences and materials engineering should be promoted in order to increase materials efficiency. End-of-pipe strategies, many waste management options, feedstock substitution, and materials substitution, become cost-effective in a situation with GHG penalties. The potential for cost reduction compared to stand-alone energy policies for GHG emission reduction should in any case generate considerable interest.

The main message for policy-making is that a significant improvement potential exists in the materials system which has not yet been given proper attention. The model calculations show that a long-term policy perspective must be provided in order to avoid short-term optimisation with undesirable long-term consequences. Policy-makers are (as of yet) unable to provide more insight into the GHG emission reduction goals beyond the period 2008-2012. Industry is not willing to take future emission reduction into account in their current investment decisions if governments are unable to specify such goals. Because of the long-term consequences of investment decisions, this is a major barrier standing in the way of a significant move towards sustainable development in the next decades. Industry should be given clear guidelines regarding emission targets to be considered for current investments with a lifespan of 2-3 decades.

Materials prices and the value of waste materials will be affected significantly by GHG emission penalties. As a consequence emission pricing instruments pose an important incentive to steer materials producers and waste management into environmentally friendly decision making. The main advantage of a pricing approach is that difficult accounting problems can be avoided that may be caused by inter-sectoral effects.

GHG emission reduction will have a limited impact on consumer product prices. The price increase for products is limited because of the low fraction of materials costs in the ultimate price of the product. This result suggests that pricing instruments are insufficient to achieve significant changes in materials consumption.

It seems unlikely that emission penalties will steer product designers and consumers into environmentally-favourable decision-making in terms of material choices. Consequently, other policy instruments must be developed in order to guide materials consumption into a sustainable direction (such as legislation and covenants for industry, improved information aimed at industrial designers, and R&D programmes geared towards the materials consuming industries, especially small and medium-size enterprises). For the other actors upstream and downstream in the materials/product chain (materials producers and waste handling companies), emission penalties will have a significant impact. Financial instruments are likely to affect their decision-making.

Increased materials efficiency and materials substitution will require substantial R&D, an adjustment of the existing capital equipment, and an adjustment of labour skills in many cases. These are areas in which government policies can help to establish new practices. In a situation with uncertain long-term policy goals, limited investments made in R&D can assist in achieving a rapid transition if the policy climate, or global climate, requires significant emission reduction. Countries and companies who are prepared for such changes can thus obtain a decisive competitive advantage.

It is not possible to categorise ‘good’ and ‘bad’ materials from a GHG emissions point of view. Each material must be analysed within the context of a certain application. Technological change in the materials life cycle can significantly change the environmental performance. The whole life cycle of this material/product combination must be considered for proper analysis given that emission strategies in the product/materials life cycle interact. The emissions for most materials will decline significantly if GHG penalties are introduced.
Some of the problems regarding materials policies are more of an organisational character. Environmental materials policies are linked to the government department for environmental affairs and the department for economic affairs (and, to a lesser extent: housing, physical planning, transport and agriculture). This division in policy-making is a formidable obstacle for any policy in this area. It is a problem that arises at the level of national governments, at European level, and in most other countries [28]. It is recommended that the integration of economic policies and environmental policies be improved in the governmental structure, e.g. through the development of a materials department that incorporates sections of the current environmental, energy, industry, transport and agriculture departments. Materials policies touch the core of industrial policy. As a result of the combination of these conflicting policy goals and the split responsibility, such policies will only be implemented in the short term if there is a win-win situation for the environment and for the economy. Concepts such as eco-efficiency (including materials efficiency) appeal to the industry because they can simultaneously enhance competitiveness (see, for instance, the World Business Council for Sustainable Development, [29]). Increased efficiency is an autonomous trend which is induced by competition. However, policies may accelerate efficiency gains. Other strategies such as substitution or product life extension are not necessarily beneficial for industry, and they will result in a restructuring of industry. As a consequence, they are less popular. These sensitivities must be considered in the design of materials policies.
11. REFERENCES


