EU Transport GHG: Routes to 2050?

Methodological issues related to assessing cost effectiveness of climate change abatement options

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10 March 2009 - Final

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Executive Summary

This study examines why studies to assess the cost effectiveness of policies addressing the climate impact of transport have yielded such widely different results to date. Our analysis of the costing methodologies in use shows there are three types of choice having a major influence on results. The first concerns the perspective adopted. Are costs being considered from the perspective of the end user, society or government? Secondly, there are a series of choices to be made in calculating direct expenditures, with respect to depreciation rates and prior estimates of investments, among other things. Finally, there is a basic choice as to whether only direct expenditures are to be included, or a comprehensive welfare-economic analysis carried out. Are the welfare effects of behavioural change or additional externalities to be included, for instance?

The conclusions are the following:

1 Particularly in the transport sector, the cost effectiveness of an abatement measure can be very different when assessed from the perspective of the end user or that of society as a whole. This is first of all because measures designed to reduce vehicle fuel consumption also affect the flow of tax revenue from road users to government, and when it comes to transport, fuel duty and other taxes make up a substantial proportion of total costs. From the perspective of the end user, savings on these costs definitely count and should be included, while from the perspective of society as a whole they do not. Secondly, climate policy measures that reduce the aggregate annual mileage of the vehicle fleet also have a substantial impact on the overall welfare of society, because they also reduce other externalities (such as air pollution and noise), which should be included from society's perspective but not from the end user's. Although the choice of perspective adopted in analysing the transport sector has a major impact on results, the choice in itself is *unproblematical*. Generally speaking, researchers and policymakers clearly distinguish that the two perspectives serve different purposes and that results cannot therefore be compared. Consequently, many studies present results for both the end user's and society's perspective.

2 The pivotal items in any calculation of cost effectiveness, whether from the end user or social perspective, are the direct expenditures associated with implementing the measure in question, in other words the capital costs, operating costs (including costs due to changes in fuel use) and regulatory costs. In this study we have examined in more detail three choices that influence calculations of direct expenditures. Are calculations based on costs ex-works or on end user (i.e. retail) prices? What baseline scenario is used, with respect to fuel price trends, for example? And how are cost price trends for new technologies estimated? The choices made with respect to these issues are found to have a major impact on estimates of direct expenditures.

3 In many national and international studies the cost effectiveness of environmental measures are calculated on the basis of direct expenditures only. However, a growing number of reports are appearing, in both policy and research circles, in which a comprehensive welfare-economic analysis is recommended. In this kind of analysis it is not only direct expenditures that are regarded as costs, but also losses of welfare associated with enforced behavioural change, the indirect costs of the measure, and additional externalities, i.e. other than those the measure is designed to reduce. This kind of analysis has been carried out for a number of individual transport policy measures. Studies in which the cost effectiveness of a wide range of measures are compared from a broader, welfare-economic angle are rare, though. .
There may be two reasons for this. First, a welfare-economic analysis is more complex and thus time-consuming than an analysis of direct expenditures. This is obviously a problem if a large number of measures are to be assessed. Second, the costs and possible benefits that a welfare-economic analysis adds to an analysis of direct expenditures follow from derivative calculations and models and are consequently more open to debate. There are two extra 'cost items' in a welfare-economic analysis that may make the outcome very different from a calculation on direct expenditures only:

a In the realm of transport, particularly, climate measures have a substantial impact on other externalities, too. Measures to cut vehicle fuel consumption reduce not only CO_2 emissions but also those of NO_x and particulates, for example. Measures to reduce aggregate annual mileage affect not only emissions but also noise, congestion and the number of road traffic injuries and deaths. As the majority of studies take most of the cited external effects to be broadly similar in terms of importance to society, whether or not the impact of a measure on these externalities is included in calculations of cost effectiveness is of major influence on results.

b Measures to reduce aggregate annual mileage or fuel consumption often mean an enforced change in behaviour: without the measure, people would have driven more kilometres or bought a different kind of car. If only direct expenditures are included, these kinds of measures would be all profit and no loss. After all, those choosing not to make a particular journey or buying a smaller car are left with more money in their pocket. In a welfare-economic analysis the conclusions look rather different, though. Not being able to do something that one would have preferred to do constitutes a loss of welfare. This loss can be expressed in monetary terms, with reference to a price incentive, for example. Such studies show that because of the already relatively high taxes on car ownership and use, additional cuts in transport volumes will be associated with high costs to society. An alternative perspective is to see the currently high costs of car ownership and use as a means of pricing negative transport externalities. In that case, to the extent that the negative externalities of transport are already priced and internalised, additional regulations can no longer bring about an increase in welfare, and may even lead to a loss. However, various arguments can be cited as to why this loss of welfare may well be less pronounced than appears at first sight from a welfare-economic analysis:
   - First, much of people's transport behaviour is conditioned. What was estimated beforehand (*ex ante*) to constitute a loss of welfare, proves subsequently (*ex post*) to be far less problematical (for consumer and researcher alike).
   - Second, the fact that people buy 'gas-guzzling' vehicles has to do with *relative* consumption. People derive personal welfare from having a bigger car than their neighbours. Policies that impinge on the entire vehicle fleet will leave relative consumption unaffected, however, thus causing less loss of welfare than originally anticipated.
   - Third, there is the objection that, as a matter of principle, an inability to engage in consumptive behaviour deemed socially undesirable should not be included as a cost item in calculating policy costs.
   - Fourth, in the case of transport pricing measures, the welfare effects can be partly offset by using the revenue to reduce other, distorting taxes like income tax. There is a growing body of literature that argues on these grounds that pricing measures in the transport sector are particularly cost-effective.

Top-down approaches are generally better equipped to handle the welfare effects described above. A major disadvantage, however, is the fact that insight is lost about the potential of individual technical options.

Recommendations

One should be careful with the conclusions drawn from cost-effectiveness analyses. First, as has become apparent, different points of departure in the calculation of cost effectiveness lead to very different results, which implies that often only the order of magnitude (is it -10, 0, 10, 25 or 100 €/ton?) can be considered significant or meaningful. Second, it should be noted that GHG abatement costs do not cover all economic, societal and political motives that lead to a decision to apply or promote a certain option, such as distributional motives and energy security.

When results from cost-effectiveness analyses are applied or compared, it is therefore recommended always to look carefully at the explicit or implicit assumptions regarding oil prices, interest rates of chosen perspective (society, end user).

Finally, it should be noted that cost-effectiveness analysis is presently more suitable for short-term than long term effects. This is particularly the case when policy measures have far reaching consequences for spatial, social and economic structures. In the long term, the palette of options to reduce emissions widens, but becomes also more unpredictable. Such options include unexpected technological breakthroughs, 'spatial adaptation' to new transport modes and prices, and habituation. Policy restrictions to already available options for consumption and transportation may cost more welfare in the short term than in the long term, when preferences adapt to the new situation. It is recommended that more research is performed in this area.

1 Introduction

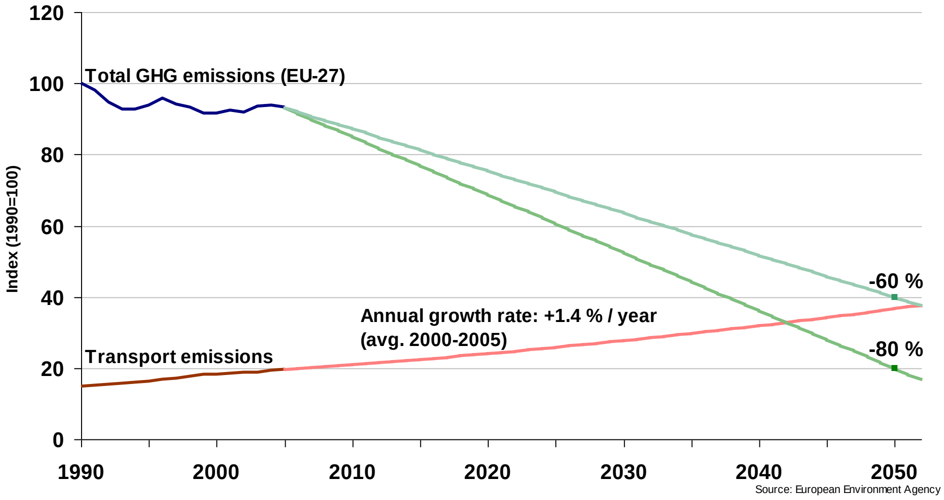
1.1 Topic of this report

This report is one of the reports drafted under the *EU Transport GHG: Routes to 2050?* The report examines why studies to assess the cost effectiveness of policies addressing the climate impact of transport have yielded such widely different results to date.

1.2 The contribution of transport to GHG emissions

The EU-27's greenhouse gas (GHG) emissions from transport have been increasing and are projected to continue to do so. The rate of growth of transport's GHG emissions has the potential to undermine the EU's efforts to meet potential, long-term GHG emission reduction targets if no action is taken to reduce these emissions. This is illustrated in Figure 1 (provided by the EEA), which shows the potential reductions that would be required by the EU if economy-wide emissions reductions targets for 2050 of either 60 or 80% (compared to 1990 levels) were agreed and if GHG emissions from transport continued to increase at their recent rate of growth. The figure is simplistic in that it assumes linear reductions and increases. However it shows that unless action is taken, by 2050 transport GHG emissions alone would exceed an 80% reduction target for all sectors or make up the vast majority of a 60% reduction target. This illustrates the scale of the challenge facing the transport sector given that it is unlikely that GHG emissions from other sectors will be eliminated entirely.

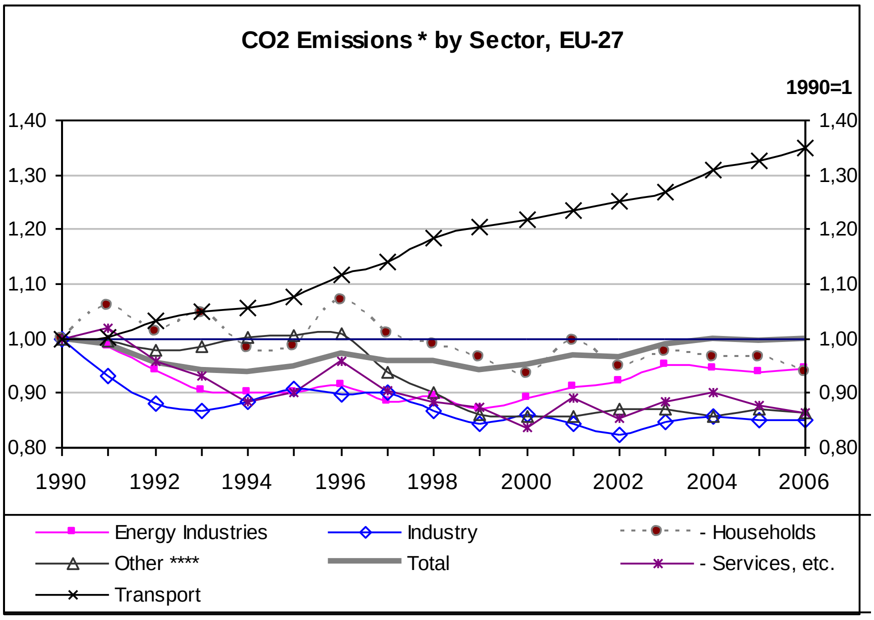
Figure 1 EU overall emissions trajectories against transport emissions (indexed)[1]



The extent of the recent growth in transport emissions is reinforced by Figure 2, which presents a sectoral split of trends in CO_2 emissions over recent years. Whilst the CO_2 emissions from other sectors have levelled out or have begun to decrease, transport's CO_2 emissions have risen steadily since 1990. It should be noted that whilst Figure 2 is presented in terms of CO_2 emissions, very similar trends are evident for GHG emissions (in terms of CO_2 equivalent) since CO_2 emissions represent 98% of transport's GHG emissions.

[1] Graph supplied by Peder Jensen, EEA

Figure 2 Carbon dioxide emissions by sector EU-27 (indexed)[2]



Notes:

i) *The figures include international bunker fuels (where relevant), but exclude land use, land use change and forestry*

ii) *The figures for transport include bunker fuels (international traffic departing from the EU), pipeline activities and ground activities in airports and ports*

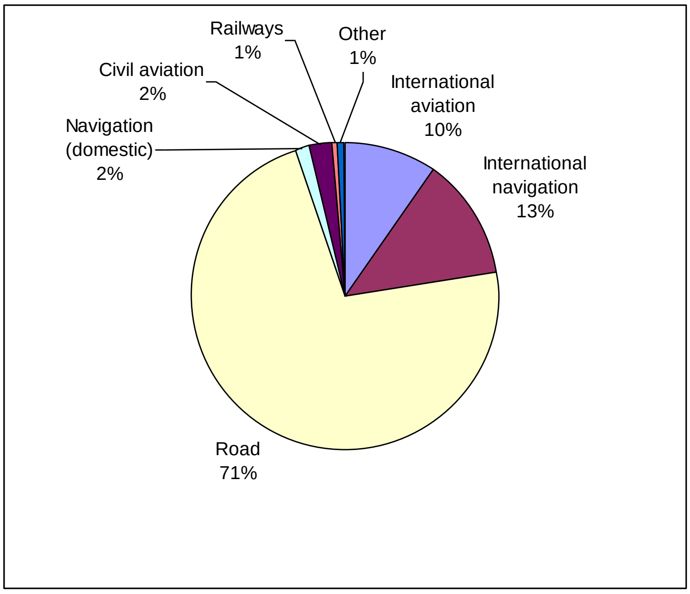
iii) *"Other" emissions include solvent use, fugitive emissions, waste and agriculture*

The vast majority of European transport's GHG emissions are produced by road transport, as illustrated in Figure 3, while international shipping and international aviation are other significant contributors.

Recent trends in CO_2 emissions from transport are also expected to continue, as can be seen from Table 1 below. Between 2000 and 2050, the JRC (2008) estimates that GHG emissions from domestic transport in the EU-27 will increase by 24%, during which time emissions from road transport are projected to increase by 19% and those from domestic aviation by 45%. It is important to note that these projections do not include emissions from international aviation and maritime transport, which are also expected to increase due to the growth in world trade and tourism.

[2] Graph based on figures in DG TREN (2008) *EU energy and transport in figures 2007-2008: Statistical Pocketbook* Luxembourg, Office for Official Publications of the European Communities.

Figure 3 Greenhouse gases emissions by transport mode (EU-27; 2005)[3]



Note: The figures include international bunker fuels for aviation and navigation (domestic and international)

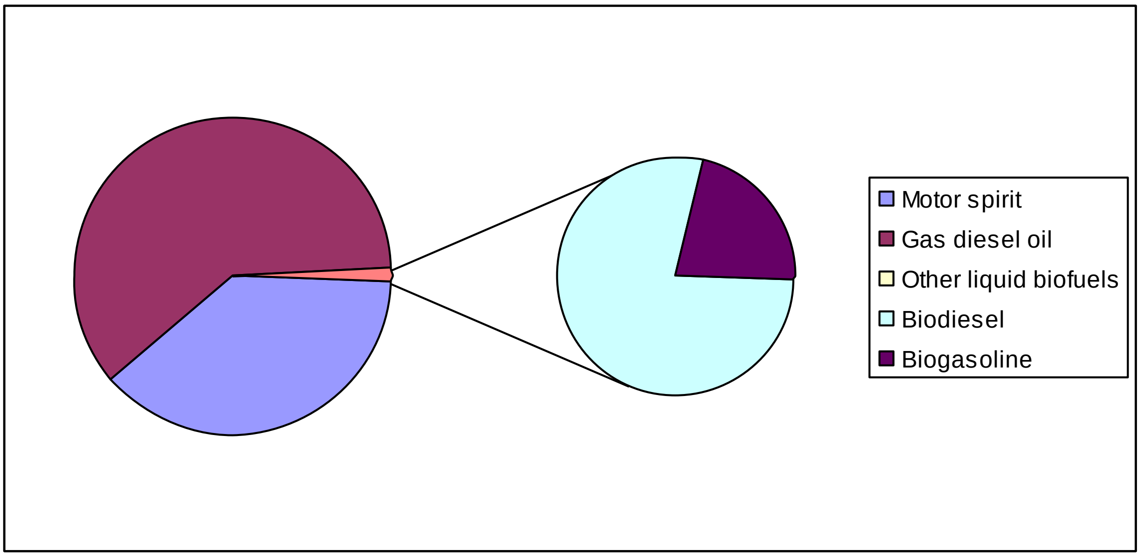
Table 1 CO_2 emissions projection for 2050 by end-users in the EU-27, in Millions tonnes of Carbon[4]

End user Catergory	1990	2000	2010	2020	2030	2050
Road transport	695	825	905	980	1,002	1,018
Rail	29	29	27	27	21	20
Domestic Aviation	86	134	179	206	237	244
Inland navigation	21	16	16	17	17	17
Total	**810**	**988**	**1,110**	**1,213**	**1,260**	**1,299**

Figures from the EEA (2008), illustrate the recent growth in GHG emissions from international aviation, as they estimate that these increased in the EU by 90% (60 Mt CO_2e) between 1990 and 2005; international aviation emissions will thus become an ever more significant contributor to transport's GHG emissions if current trends continue. Furthermore, the IPCC has estimated that the total impact of aviation on climate change is currently at least twice as high as that from CO_2 emissions alone, notably due to aircrafts' emissions of nitrogen oxides (NO_x) and water vapour in their condensation trails. However, it should be noted that there is significant scientific uncertainty with regard to these estimates, and research is ongoing in this area.

[3] Graph based on figures in EEA (2008) *Climate for a transport change – TERM 2007: Indicators tracking transport and environment in the European Union* EEA Report 1/2008, Luxembourg, Office for Official Publications of the European Communities.
[4] Taken from JRC (2008) *Backcasting approach for sustainable mobility* Luxembourg, EUR 23387/ISSN 1018-5593, Office for Official Publications of the European Communities.

Figure 4 Final transport energy consumption by liquid fuels in EU-27 (2005), ktoe5



The principal source of transport's GHG emissions is the combustion of fossil fuels. Currently, petrol (motor spirit), which is mainly used in road transport (e.g. in passenger cars and some light commercial vehicles in some countries), and diesel, which is used by other modes (e.g. heavy duty road vehicles, some railways, inland waterways and maritime vessels) in various forms, are the most common fuels in the transport sector (see Figure 4). Additionally, liquid petroleum gas (LPG) supplies around 2% of the fuels for the European passenger car fuel market (AEGPL, 2009[6]), while the main source of energy for railways in Europe is electricity, neither of which are included in Figure 4. While, alternative fuels are anticipated to play a larger role in providing the transport sector's energy in the future, currently they only contribute 1.1% of the sector's liquid fuel use.

1.3 Background to project and its objectives

The context of the *EU Transport GHG: Routes to 2050* is the Commission's long-term objective for tackling climate change, which entails limiting global warming to 2°C and includes the definition of a strategic target for 2050. The Commission's President Barosso recently underlined the importance of the transport sector in this respect be noting that the next Commission "needs to maintain the momentum towards a low carbon economy, and in particular towards decarbonising our electricity supply and the transport sector"[7]. There are various recent policy measures that are aimed at controlling emissions from the transport sector, but these measures are not part of a broad strategy or overarching goal. Hence, the key objective of this project is to provide guidance and evidence on the broader policy framework for controlling GHG emissions from the transport sector. Hence, the project's objectives are defined as to:

- Begin to consider the long-term transport policy framework in context of need to reduce greenhouse gas (GHG) emissions economy-wide.
- Deal with medium- to longer-term (post 2020; to 2050), i.e. moving beyond recent focus on short-term policy measures.
- Identify what we know about reducing transport's GHG emissions; and what we do not.
- Identify by when we need to take action and what this action should be.

[5] Graph based on figures in DG TREN (2008), page 206
[6] European LPG Association (2009) *Autogas in Europe, The Sustainable Alternative: An LPG Industry Roadmap,* AEGPL, Brussels. See http://www.aegpl.eu/content/default.asp?PageID=78&DocID=994
[7] http://ec.europa.eu/commission_barroso/president/pdf/press_20090903_EN.pdf

Given the timescales being considered, the project will take a qualitative and, where possible, a quantitative approach. The project has three Parts, as follows:

- Part I ('Review of the available information') has collated the relevant evidence for options to reduce transport's GHG emissions, which was presented in a series of Papers (1 to 5), and is in the process of developing four policy papers (Papers 6 to 9) that outline the evidence for these instruments to stimulate the application and up take of the options.
- Part II ('In depth assessment and creation of framework for policy making') involves bringing the work of Part I together to develop a long-term policy framework for reducing transport's GHG emissions.
- Part III ('Ongoing tasks') covers the stakeholder engagement and the development of additional papers on subjects not covered elsewhere in the project.

As noted under Part III, stakeholder engagement is an important element of the project. The following meetings were held:

- A large stakeholder meeting was held in March 2009 at which the project was introduced to stakeholders.
- A series of stakeholder meetings (or Technical Focus Groups) on the technical and non-technical options for reducing transport's GHG emissions. These were held in July 2009.
- A series of Technical Focus Groups on the policy instruments that could be used to stimulate the application of the options for reducing transport's GHG emissions. These were held in September/October 2009.
- Two additional large stakeholder meetings at which the findings of the project were discussed.

As part of the project a number of papers have been produced, all of which can be found on the project's website, as can all of the presentations from the project's meetings.

1.4 Background and purpose of the paper

The gravity of the risk of climate change may require far-reaching and costly abatement options. Citizens demand, however, that governments do not impose higher costs upon them than necessary, i.e. that governments design climate policy in the most cost-effective manner. Therefore, insight in the costs of emission abatement plays an important role in environmental policy making. This is particularly true in the case of (private) transport, a part of life where society is not very eager about government intervention.

Over the last years many studies have been performed in which the costs of emission abatement in transport have been assessed. A variety of approaches exists, however, to the determination of cost effectiveness leading to diverging conclusions and policy recommendations. While some studies conclude that transport is the social sector where the most emission abatement can be achieved at zero or *negative* costs (see e.g. Greene and Schafer, 2003; NRC, 2002; T&E, 2005; McKinsey, 2007), other studies conclude that transport is a relatively expensive sector compared to for example the energy sector (EEA, 2005; IEA, 2006; Stern, 2006: annex 7c).

1.5 Structure of the paper

The report is structured as follows. The purpose of the present report is first to explain why different calculation methods to the determination of cost effectiveness lead to different results (chapter 2). In chapter 3, the most common presentation of inventories of abatement options and their costs are discussed, the so called marginal abatement cost curves (MACCs). In chapter 4, recommendations are given how to improve the methodologies.

2 Different calculation methods

2.1 Introduction

The purpose of this chapter is to review and discuss the principal choices that explain the differences in the results obtained by the various calculations of policy cost effectiveness in the transport sector. This chapter builds further on previous reports (CE Delft, 2006, 2007; OECD, 2009). We do not provide a systematic, step-by-step description of a full cost effectiveness analysis (CEA), however. Aspects that are more or less uncontroversial are discussed only briefly.

In section 2.2, we shall first explain the general definition of cost effectiveness. In the following sections, we discuss the methodological issues. An analysis of costing methodologies shows that three clusters of significant choices can be distinguished. The first is the perspective adopted: are costs considered from the perspective of the end user, society as a whole, or government? This topic is discussed in Section 2.3. Second are the choices made in calculating direct expenditures with respect to depreciation rates, estimated investments, and so on. These are the subject of Section 2.4. Third is the choice of whether only direct expenditures are to be included or a comprehensive welfare-economic analysis carried out. Are the welfare effects of behavioural changes or externalities included, for example? These issues are discussed in Section 2.5.

2.2 Cost effectiveness

The cost effectiveness of greenhouse gas abatement options is defined as the costs of an option divided by its greenhouse gas abatement potential, and is expressed in €, $ or £ per ton of C or CO_2 equivalents abated. For example, if an option abates 100 ton CO_2 and costs €1,000 its cost effectiveness is €10 per ton CO_2.

Cost estimates typically include such items as the capital costs, operating and maintenance costs, and costs or benefits due to changes in fuel use. However, other items may be included as well, such as regulatory costs and the welfare costs related to changes in behaviour. Therefore, it should be noted that although the general definition of cost effectiveness is straightforward, there are various views on the precise calculation leading to divergent results, which will be discussed in the following sections.

The following generic formula is often used (e.g. Blok, 2001; AEA, 2001; INFRAS, 2006):

$$\text{Cost effectiveness} = \frac{I^{an} + \Delta_{O\&M} - \Delta_{\text{fuel costs}} - \text{secondary benefits}}{\text{annual } CO_2 \text{ emission abatement}}$$

In the formula I^{an} is the annuity of the total investment costs I:

$$I^{an} = I * \frac{(1+r)^l * r}{(1+r)^l - 1}$$

where l is the lifetime of the option, r the discount rate (generally 4% for calculating social costs) and I the total investment.

$\Delta_{O\&M}$ represents the additional annual operating and maintenance costs and $\Delta_{\text{fuel costs}}$ the annual savings on fuel costs. The formula also includes monetised secondary benefits (e.g. cuts in air pollutant emissions through use of more efficient technology).

A different approach would be to calculate the cost effectiveness of an abatement option on the basis of *total* costs and benefits instead of the annual costs and benefits (see e.g. TNO, 2006). Such a formula would include total investment costs, lifetime fuel cost savings and lifetime CO_2 emission abatement. In that case the net present value (NVP) has to be calculated of both future streams of costs and benefits.

2.3 Differences in perspective

In the literature three perspectives on cost effectiveness are distinguished, differing with respect to the party or parties from whose perspective the costs are considered: those directly affected by the option in question (the end user), society as a whole, or government. We first discuss these various perspectives and their application individually before going on to examine the importance of the choice in Section 2.3.4.

2.3.1 The end user

From the perspective of the end user, the costs calculated are those that are incurred by those directly affected by the environmental option, i.e. from the perspective of companies, institutions or households. This perspective yields insight into how the costs of a given environmental policy instrument are distributed over the various actors in society and, with it, an idea of the likely degree of support it will enjoy among them. From the end user's perspective:

- externalities such as pollution and congestion do not count, as these by definition fall on others;
- the effects of taxes and subsidies do count, even though from the perspective of society as a whole these represent no more than a redistribution;
- government implementation costs do not count;
- the depreciation period for investments is generally shorter (or the discount rate higher) than that from the perspective of society as a whole.

In the transport and environment context, the end user perspective has been adopted in a number of international academic studies on, among other things, the cost effectiveness of fuel-saving vehicle technologies (see, for example: AEA, 2001; Decicco & Ross, 1996; IEEP *et al*., 2004; TNO, 2006). An example of a study on the cost effectiveness of biofuels in which the end user perspective is taken is S&T Consultants (2003).

The end user perspective is also often adopted by regional and international policymakers. OECD/ECMT (2007) cites cost effectiveness data compiled by CE Delft (CE, 2005b), in which the (additional) expenditures for industry involved in implementing various emission abatement technologies are reviewed. This method can be categorised as a restricted end user approach.

2.3.2 Society

In a social perspective on cost effectiveness it is the costs to society as a whole that are calculated. Although this is generally at the national scale, an international perspective may also be adopted, as in CONCAWE (2007), where the analysis is at the European level. The social perspective is useful in the macro-economic context, when the focus is on impact on overall social welfare, irrespective of distribution effects. From the perspective of a given society:

- in principle, external (environmental or congestion) benefits do count;
- the effects of taxes and subsidies do not generally count, to the extent that these merely entail redistribution;
- government implementation costs do count (human resources, outsourcing of information services, consultancy, monitoring, etc.);

- investments are generally written off over a longer period (i.e. a lower discount rate is assumed) than from the end user perspective.

In recent studies of the cost effectiveness of climate abatement options in the transport sector it is society's perspective that has generally been adopted. This is the case for the assessments of vehicle fuel-saving technologies reported in IPCC (2001), AEA (2001), ECMT (2006), IEEP *et al.* (2005), Johansson & Aahman (2002), Kleit (2004), T&E (2005) and TNO (2006). In 'Energy Technology Perspectives: Scenarios and Strategies to 2050' the IEA (2006a) also opts for a social perspective, as evidenced by the adjustment of fuel savings for taxes and the low discount rate employed in calculating net present value. The IEA study plays a pivotal role in IPCC (2007) as well as the Stern Review (2006). In studies on the cost-effectiveness of biofuels, too, it is society's perspective that is generally adopted (e.g. CONCAWE, 2007; CE, 2005). Finally, in the Dutch government's 'Option Document on Transport Emissions' (RIVM/CE, 2004), reviewing the CO_2 emission cuts and costs of a range of abatement options and policy instruments, among other issues, a social perspective is also taken.

It should be noted, though, that not all studies based on a social perspective show the same degree of 'completeness' when it comes to the array of costs considered. Indeed, in many cases it involves no more than introducing a correction for taxes. Various studies on the cost effectiveness of fuel-saving vehicle technologies, for example, include no more than extra vehicle costs and (savings on) fuel costs. See, for example: IEEP *et al*., 2005; T&E, 2005; TNO, 2006. Reductions in transport externalities other than CO_2 emissions (such as air-polluting emissions, noise nuisance and road safety) are left out of the picture. Including these other items in calculating the cost effectiveness of abatement options and policy instruments is discussed in Section 2.5.

In their official national guidelines for analysing the cost effectiveness of abatement options, the United States (EPA, 2000; 2008), the United Kingdom (DfT, 2006) and Flanders (LNE, 2007) have adopted a social perspective.

2.3.3 Government

Here, we take the notion of 'CEA from a government perspective' to mean an analysis of *government expenditures*. In principle, two approaches can be distinguished. In the first, only implementation costs or 'apparatus costs' are considered, that is, all the costs incurred in the proper functioning of a given policy. These include the costs of human resources – for enforcing regulations, implementing subsidy schemes, creating an emissions trading scheme, designing policies, etc. – as well as outsourcing of information services, consultancy, monitoring, training and so on. In the second approach it is not only the apparatus costs that are considered, but also government subsidies. Here, we are concerned with subsidy effectiveness; see, for example: FEM, 2007.

2.3.4 Discussion

There are a number of studies in which the analysis is carried out from both the end user and social perspective, in the conviction that each costing methodology yields its own specific information and has its own particular field of application. Indeed, on theoretical grounds there is no particular perspective that is inherently superior, and the right one to adopt depends on the type of information required. Among the researchers reporting from both perspectives there is consequently little debate as to which is the 'right' one (*cf.* IPCC, 2007).

Particularly in the transport sector, the cost effectiveness of an abatement option can be very different when assessed from the perspective of the end user or that of society as a whole. This is first of all because abatement options or policy instruments designed to reduce vehicle fuel consumption also affect the flow of tax revenue from road users to government, and when it comes to transport, fuel duty and other taxes make up a substantial proportion of total costs. From the perspective of the end user, savings on these costs definitely count and should be included, while from the perspective of society as a whole they do not, because all that is involved is a transfer from end users to government. Secondly, transport policies that reduce aggregate annual mileage of the vehicle fleet also have a substantial impact on the overall welfare of society, because they also reduce certain other externalities, which should in principle be included from society's perspective but not from the end user's.

It is precisely the existence of high rates of duty on transport fuels compared with fuel taxes in other sectors that is one of the reasons why several studies concluded that climate abatement options in the transport sector are less cost-effective than options elsewhere – in the energy sector, for example (IEA, 2006a; Strachan *et al*., 2007). Because of the high taxes currently in place, although further fuel savings may mean additional cost savings from the end user's perspective, this is far less so from society's perspective, as the government then also has less tax revenue (*cf.* van Herbruggen & Proost, 2002).

A simplified illustration: a more fuel-efficient engine costs an additional €1,000 and saves around €1,500 on fuel and 3 tonne CO_2 during the vehicle's service life. Of the €1,500 fuel savings, around €1,000 is tax, however. From the end user's perspective, the cost effectiveness is then (1000-1500)/3 = *minus* €167/tonne CO_2: the option leads to 'earnings'. From society's perspective, though, the cost effectiveness is (1000-500)/3 = *plus* €167/tonne CO_2.

Although the end user and the social perspective clearly serve different purposes, in many of the studies and policy documents reviewed it is far from clear which perspective has been adopted or how the term cost effectiveness is being precisely employed. Such is the case, for example, for studies from Canada (AMG, 2002), Australia (Abare, 2006), the US (CBO, 2003), the UK (HMT, 2003) and Sweden.[8] In these cases there is a danger that in one and the same study different perspectives will be confounded when comparing the results with those of other studies.

2.4 Differences in calculating direct expenditures

They key element of all calculations of cost effectiveness, whether from the perspective of the end user or society as a whole, are the direct expenditures involved in implementing the abatement options in question. Before going on to discuss other cost items in Section 2.5, in this section we first examine the choices affecting calculations of direct expenditures. First, in Section 2.4.1 we discuss these expenditures in a general sense. In the subsequent sections we then consider the following particular issues: **Error! Reference source not found.** (§ 2.4.2), **Error! Reference source not found.** (§ 2.4.3), **Error! Reference source not found.** (§ 2.4.4), oil prices (§ 2.4.5), and depreciation period or discount rate (§ 2.4.6). In Section 2.4.7 there follows a brief discussion.

Nota Bene: These sections are for illustrative purposes only and are not intended as a full review of all the possible choices affecting calculations of direct expenditures.

2.4.1 Direct expenditures

The direct expenditures of climate abatement options can be broken down into three cost categories (LNE, 2007):

- capital costs;
- operational costs;
- regulatory costs.

Capital costs refer to the sum total of one-off costs associated with the implementation of the abatement option. Although this generally means the purchase price or production costs of an

8 Indirectly, it is often possible to deduce what perspective has been adopted. If a high discount rate is used, or if calculations are based on market prices, as in CBO (2003), for example, it is likely that an end user approach has been employed. In the case of HMT (2003) things are more difficult, because although market prices are used, so, too is a low discount rate (3.5%).

investment, other one-off costs may also fall under this heading, such as the cost of retrofitting a technology and possible training costs. *Operating costs* are the current expenditures incurred in making the climate option or technology operational and keeping it up and running. One example would be the (extra) maintenance costs incurred in operating a hybrid car. Rather than additional costs, some abatement options may lead to savings or sometimes even financial gains. Thus, driving a more efficient car leads to savings on fuel costs. When calculating the costs of an abatement option, due allowance must also be made for these side-effects. This is achieved by employing the net operating costs, i.e. the gross operating costs minus the savings and/or financial gains. *Regulatory costs*, finally, refer to the costs incurred by the government regulator. Here we are concerned with the costs of policy estimation, implementation and enforcement, among other things. The additional costs accruing to groups targeted by the abatement option following its introduction but not contributing directly to achievement of the climate target also come under the heading of regulatory costs. Here we are concerned primarily with administrative costs such as the costs of data collection, preparation of progress reports and so on.

Which direct costs are to be considered as contributing to the costs of an abatement option depends on the perspective that has been adopted. If this is the government perspective, it is only the regulatory costs that are relevant, insofar as they indeed borne by government. From the end-user perspective it is the capital costs and operating costs that are important, while from society’s perspective all categories of cost should be included. From the end-user perspective the costs should be taken inclusive of taxes, while from society’s perspective they should not.

In their analysis, most studies take the same direct expenditures. Thus, all the studies on fuel-saving vehicle technologies consider extra vehicle costs and fuel savings. However, this still means that certain direct costs are left out of the equation, such as changes in maintenance costs relative to the baseline situation. At the same time, though, because there is still generally little experience with the various new technologies, there is also major uncertainty as to the magnitude of such costs and it is indeed not even clear whether maintenance costs will go up or down. This is the main consideration for assuming zero maintenance costs in calculations. More broadly, regulatory costs are seldom included in analyses of cost effectiveness.

2.4.2 Cost ex works versus end user prices

In most studies direct expenditures are calculated on the basis of retail prices, with taxes being deducted if the analysis is from society’s perspective. In IEEP *et al.* (2004) the producer mark-up is also deducted from the sales price (*cf.* CE, 2005b). The same approach was adopted by IIASA in the RAINS and GAINS models (IIASA, 1998; Klimont *et al.,* 2002; IIASA, 2005). On this issue TNO (2006) notes, however, that producer prices can be seen as a reward for entrepreneurial risk and should therefore be included as costs. The same reasoning is implicitly adopted by AEA (2001), which calculates the cost of fuel-saving vehicle technologies by including a mark-up of 20% for producers and 12% for dealers on top of the production costs of the technologies. Similarly, in calculating the costs of fuel saving technologies IEA (2006) proceeds from the retail price, including the producer mark-up but excluding taxes. In assessing the cost of modifications to vehicles to equip them for burning alternative fuels, CONCAWE (2007) also uses retail prices. Finally, it is this approach that is adopted in the treatment of profits in the ‘Guidelines for Social Cost-Benefit Analysis’ for environmental policy (CE, 2007).

2.4.3 Baseline scenario and baseline technology

Every cost effectiveness analysis should, in principle, compare the additional costs of a given technology or abatement option with those of a baseline technology or policy. This means making an explicit choice as to such a reference. The baseline scenario adopted in determining the costs of CO_2 abatement options is thus crucial for calculating the additional costs, for the aim of the exercise is to gain insight into how the costs would have developed if the technology or abatement option were not implemented. In the literature, various kinds of baseline scenario are distinguished (IPPC, 2007):

- The efficient baseline scenario, in which it is assumed that all relevant resources are efficiently used.
- The inefficient baseline scenario, in which certain market distortions are assumed, in the labour and/or energy market, for example.
- The 'business as usual' scenario, or technology, in which it is assumed that past trends will be continued in the future (autonomous trends) and that no new policies are introduced.

The calculated costs of a given climate abatement option will depend on the choice of baseline scenario or technology. Thus, the costs will be greater if an efficient baseline scenario is adopted rather than a business-as-usual scenario, as the costs arising in the latter case will often be largely compensated by energy savings. In studies on climate abatement options in the transport sector, a business-as-usual scenario is generally used.

One key issue in assessing the cost effectiveness of transport climate abatement options is the assumed trend in the oil price adopted in the baseline scenario. This variable is, firstly, of major influence on the magnitude of the benefits accruing as a result of fuel savings. In addition, there is a negative correlation between the costs of biofuels and oil price: the higher the latter, the cheaper biofuels become in relative terms, and thus the lower their additional cost. Because of the pivotal importance of the oil price when it comes to the costs of climate options, in most studies a range of values is taken (see, for example: CONCAWE, 2007; IEEP *et al*., 2004; TNO, 2006). In contrast, there are also studies in which calculations are based on a single oil price (for example: S&T Consultants, 2003).

2.4.4 *Ex ante* versus *post* cost estimates

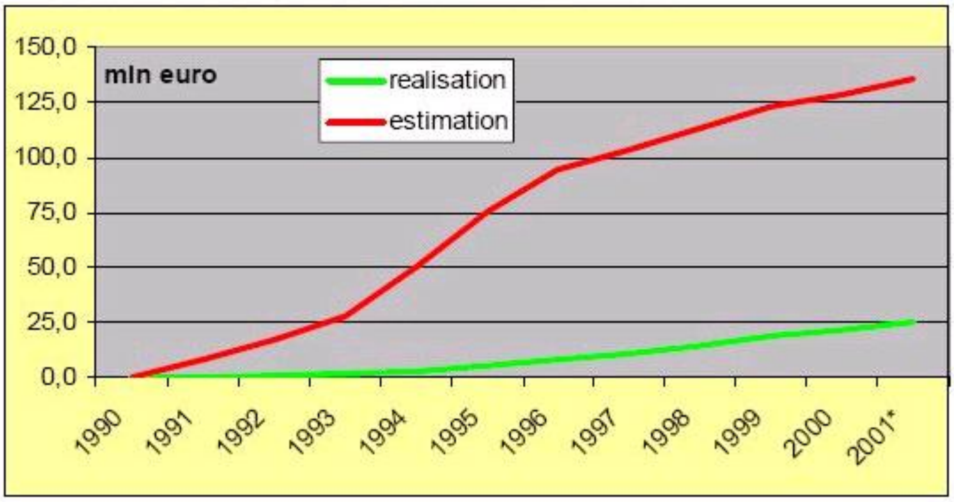
The costs of a given abatement option can be calculated prior to or after implementation: *ex ante* and *ex post*, respectively. When setting out future policy, on the basis of an Option Document, for example, *ex ante* cost estimates are generally required, unless the envisaged abatement options are already tried and tested.

In the case of *ex ante* estimates, the additional costs are assessed with reference to future policy scenarios (see previous section). With *ex post* estimates, the aim is to assess how costs and technologies would have developed if the abatement option had *not* been implemented. In practice, it often proves difficult to do so with any great accuracy (*cf.* CE, 2005a).

A number of *ex post* cost estimates of options implemented in the past have shown that *ex ante* studies prior to implementation of these options generally overestimated the costs (Burtraw, 1996; SEI, 1999; Harrington, 2000; CE, 2006; IvM, 2006; TME, 2006; AEA, 2007). Thus, the study by TME (2006) shows that initial estimates (1985-1990) of the costs of introducing European standards on vehicle emissions and fuels overestimated these costs by a factor two. In such cases, the differences are due mainly to underestimation of economies of scale and unforeseen technical developments (learning curve), both leading to cheaper solutions (*cf.* Honig *et al*., 2001). In some studies it is also observed that cost studies may sometimes be undertaken for strategic reasons, in an attempt to thwart tougher environmental legislation, for example, which may exert 'upward pressure' on calculated costs.

As an example, Figure 5, taken from TME (2006: 9-10), reviews the *ex ante* estimates and *ex post* empirical values for introduction of three-way catalytic converters in diesel passenger cars and other light-duty vehicles. The *ex ante* estimates were based on investment costs of € 817 per vehicle plus additional fuel consumption, while *ex post* investment costs proved to be between €130 and 240 per car, with no extra fuel consumption.

Figure 5 Comparison of *ex ante* and *ex post* annual cost assessments of engine modifications for diesel passenger cars and light duty vehicles, 1990 - 2001, price level 2002



source: (estimation) TME, 1993 and (realisation) CBS, 2005

There are no concrete recommendations to be made on how *ex ante* costs should be estimated. What is clear, though, is that different approaches may lead to a very wide range of estimates of the cost effectiveness of abatement options, particularly when it comes to new technologies like more efficient vehicle engines.

2.4.5 Oil prices

Obviously, the costs of emission abatement are much easier recovered at an oil price of $150/barrel than $50/barrel. Fuel prices have fluctuated strongly over the last years. While for many years the fuel price has been estimated at an average of $20 per barrel, prices peaked above $140 in 2008, dropped to $30 again, and returned to about $70 per barrel at the time of writing (September 2009). It goes without saying that the assumed fuel price has a decisive influence on the determination of cost-effectiveness of fuel saving options. In other words, if an option is deemed cost effective at an expected fuel price of $100 per barrel, it may not be so at a price of $50.

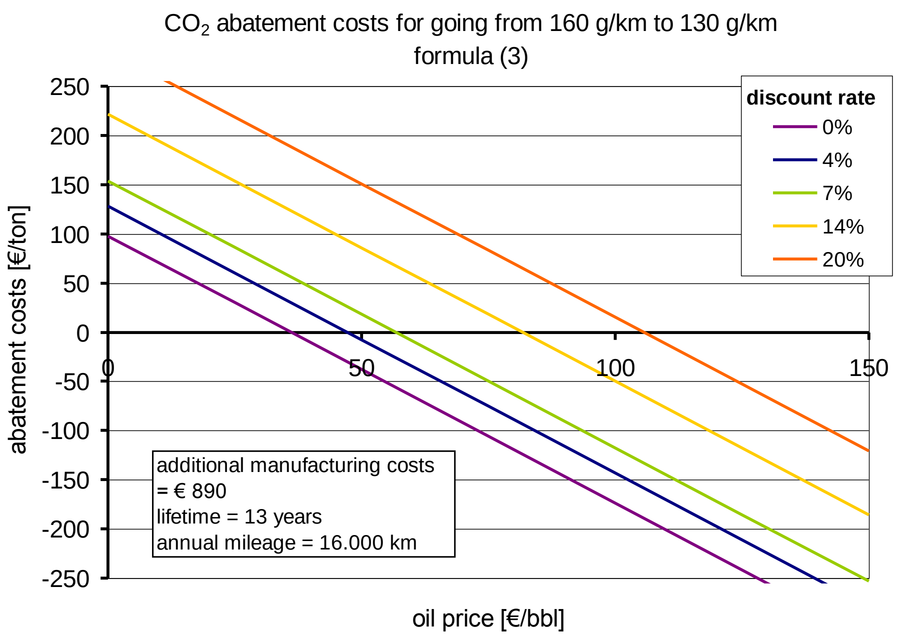
2.4.6 Depreciation period or discount rate

Governments often calculate with a social discount rate of about 4%. Consumers, however, are much more ‘short-sighted’ and face higher risks that expected savings do not materialize. They consequently demand depreciation periods of a few years and may use discount rates above 20% (see IEEP (2006) for an extensive discussion of the subject and references). Firms face additional risks of adopting new technologies that are not captured by the social discount rate because their assets are less diversified than those of society as a whole (Sutherland, 1991; Rivers and Jaccard, 2005). DeCanio (1993) showed that firms typically establish internal hurdle rates for energy efficiency investments that are higher than the cost of capital to the firm (OECD/IEA, 2007). Furthermore, there is a great deal of uncertainty around the potential outcomes of adopting new technologies. Early investors may be skeptical about the prospects of a technology and demand a premium on return in order to cover the risks of the investment (MMA, 2008). Apart from this uncertainty regarding the *reliability* of new technologies, there is uncertainty regarding economic trends (e.g. fuel prices) and governmental policy, and uncertainty regarding sector and company trends (Sorrell *et al.*, 2000).

2.4.7 Discussion

We have discussed several choices that go some way to explaining why different cost effectiveness analyses can sometimes yield such widely differing results. The choices made with respect to these issues are found to have a major impact on estimates of direct expenditures as illustrated in Figure 6. Depending on the assumed oil price or discount rate the marginal abatement costs for meeting a 130 g/km sales averaged target for passenger cars in 2012 is estimated between *minus* 300 and *plus* €300 per ton.

Figure 6 Marginal abatement costs for meeting a 130 g/km sales averaged target for passenger cars in 2012, based on results from (IEEP 2008), as function of assumed oil price and discount rate



2.5 Direct expenditures versus welfare-economic analysis

In a comprehensive, welfare-economic approach, the overall cost of an emissions abatement option is given by the balance of all the welfare effects of the option excluding the actual intended effect of the option (in our case, on the climate). These welfare effects may comprise direct expenditures ('out-of-pocket' expenses), such as payments for capital goods and maintenance work associated with the option, as well as savings on fuel expenditures. The welfare effects may also be indirect, though, or not be expressed financially in any way. Examples include environmental side-effects, such as reduced particulate emissions, reduced road congestion, or the broader welfare effects of behavioural change.

A comprehensive welfare-economic analysis seeks to include as many welfare effects as possible in the calculations, including indirect and unpriced effects. Because the latter costs and benefits are valued using economic models and monetisation techniques, these are surrounded by more uncertainty as well as controversy than direct expenditures. In the Dutch 'Environmental Costing Methodology Manual' (VROM 1994, 1998) this was one of the motives for restricting the analysis to direct expenditures and ignoring indirect and unpriced effects. The approach adopted in the Dutch

'Manual' has been adopted elsewhere, for example in the guidelines of the European Environmental Agency (EEA, 1999).[9] More recently, though, there has been growing interest in welfare-economic analysis and, with it, major growth in prominence of Social Cost-Benefit Analysis (SCBA) (see, for example, the Dutch 'Guidelines on Cost-Benefit Analysis of Infrastructure Projects': CPB & NEI, 2000, CBO, 2003; VITO, 2003; Verhoef *et al*., 2004; Proost *et al.,* 2006; LNE, 2007). An SCBA in which the external benefits of climate abatement options are discussed but remain unvalued can be considered an extended variant of cost effectiveness analysis. In such studies the various welfare effects of climate abatement options are examined but no attempt made to translate the associated cuts in greenhouse gas emissions into monetary units.

In the following sections, we discuss some of the most important issues.

2.5.1 Direct effects: welfare losses and gains of behavioural change

A purely technological abatement option, which comes at zero costs as well, does not affect people's consumption patterns, daily behaviour or lifestyles. One could think of a more efficient engine which does not affect the other characteristics of the car. Such an abatement option would not affect people's preference satisfaction (except those related to environmental protection). However, most new abatement options or policy instruments, besides leading to direct expenditures, may also induce behavioural change. For example, rather than incurring the extra costs implied by the option, some people will opt to discontinue a particular form of environmentally damaging behaviour, as when a cleaner car is more expensive. With some policy instruments, such as road pricing, behavioural change is indeed precisely what is intended, for such schemes are designed to reduce overall fleet mileage. Those driving less as a result of the policy instrument also suffer a loss of welfare, however, being unable to do something they would have preferred to do. Although this loss of welfare is real, it is not evidenced in any tangible flow of money. The only thing to emerge from an examination of concrete monetary flows will be that these people spend less on fuel. The same holds for policies designed to 'downsize' the vehicle fleet, i.e. reduce average vehicle size. The demand for large cars shows that people derive welfare from such a purchase. Limiting the freedom of choice in this respect would therefore mean a loss of welfare for those who would otherwise have bought a bigger car. In short: welfare encompasses more than purely monetary flows.

In an extensive study of the determinants of people's choice of transport mode, Slotegraaf *et al.* (1997) report that in addition to travel costs, issues like flexibility, comfort, a feeling of control, social standards and status considerations also play a role. These findings have been confirmed in a number of other studies (see, for example: Anable & Gattersleben, 2005; Ory & Moktherian, 2005; Steg, 2005; Stradling *et al.,* 1999; see IEEP, 2006). In assessing the factors determining people's choice of transport mode during rush hour traffic, Steg *et al.* (2001) even established that it is exclusively non-financial factors that are of influence.

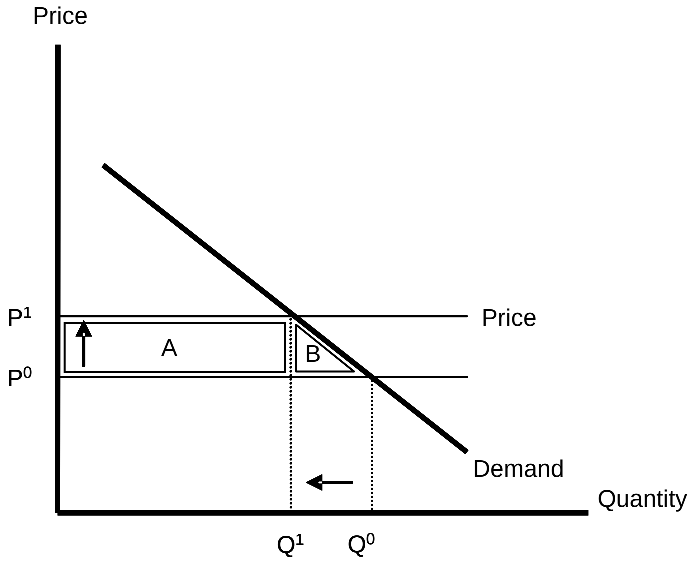
In a welfare-economic analysis, all these various losses of welfare are included in the calculations. As an illustration, **Error! Reference source not found.** shows the loss of welfare resulting from road pricing, i.e. a kilometre-based charge. The x-axis of the graph represents demand for 'automobility' and the y-axis its price per kilometre. In the situation with no charge, the marginal private costs are equal to P^0, while demand for automobility is given by Q^0. Introduction of road pricing, raising the kilometre price from P^0 to P^1, will lead to a reduction in automobility from Q^0 to Q^1 kilometres. This reduced automobility is suffered by the 'quitters': those opting for a different mode of transport or giving up a certain amount of mobility altogether. Although for this group expenditures decline, the loss of automobility is experienced as costs, so that on balance they undergo a loss of welfare given by the triangle B*.* It is a loss of welfare, because this group opts for an alternative that affords them less utility. If we take the demand curve to be more or less linear between the price levels P^0 and P^1, the

[9] In calculating its cost effectiveness curves for acidifying pollutants, IIASA also explicitly restricts its analysis to direct expenditures (IIASA, 1998: 25). Although at a recent IIASA workshop (IVL, 2006) it was acknowledged that non-technical measures, including behavioural change, are starting to gain in importance, the difficulty of including the costs of such measures in a model like RAINS in such a way that they can be compared satisfactorily with the costs of technical measures was also stressed.

loss of welfare can be estimated as *half* the kilometre charge times the number of kilometres less that are driven.

All the other road users pay a higher price per kilometre than before. Their loss of welfare is represented by the rectangle A.[10] In this case, though, we are concerned with a distributional effect, viz. the transfer of income from road users to the government collecting the charge revenues.

Figure 7 Welfare effects of a cost price increase (charge)



This kind of welfare loss may also accrue from a cost price increase resulting from extra investments, as when vehicle prices rise owing to mandatory technical provisions, for example. In that case rectangle A represents the investment costs plus operating costs. There may also be additional welfare effects due to reduced demand equivalent to triangle B (see, for example: CBO, 2003).

Finally, a loss of welfare will occur when behaviour is directly regulated, as when a new speed limit is introduced. In a number of studies this loss of welfare is factored in by assigning a monetary value to the resultant increase in travel time (RIVM/CE, 2004; CPB & V&W, 2004).[11]

Calculating the loss of welfare (loss of consumer surplus) associated with enforced behavioural change is a controversial issue, however. There are those who hold that calculations based on existing preferences *over*estimate this loss of welfare, because no allowance is made for habituation and adjustment to the new, regulated situation – with hindsight, the behavioural change proves to be not as 'tough' as it looked in the prior estimate, or the associated costs prove lower than anticipated. In addition, preferences may well change in the new situation. After initial introduction of a price incentive, for instance, after a while a lower incentive may prove sufficient to achieve the same behavioural result. In a certain sense, the situation is analogous to the aforementioned, empirically proven differences between *ex ante* and *ex post* estimates of the costs of technologies like catalytic converters, where the latter estimates are generally far lower than the former (§ **Error! Reference source not found.**).

There is, secondly, the principled argument that losses of welfare resulting from a person being unable to perform certain activities deemed less desirable from society's perspective should not be included in the costs of climate policy. A simple example of this argument is provided in the calculation by RIVM/CE (2004: 149) of the cost effectiveness of EU legislation on mandatory speed limiters (100

[10] This makes no allowance for any welfare *gains* accruing to those who continue to drive, in the form of reduced travel time (less congestion) or increased freedom of movement, for example. *Cf.* CPB, 2000.
[11] Note that travel time losses are not the only kind of welfare cost involved; restriction of the freedom to drive as fast as one would like is in itself also a cost item, although its magnitude is difficult to assess.

km/h) in light goods vehicles. In this calculation, allowance is made for the costs arising from extra salary costs due to longer travel times. At the same time, though, RIVM/CE state that some of the total time lost can be attributed to light commercial vehicles exceeding the speed limit: “As the kilometres driven over the speed limit yield ‘illegal’ time gains, this portion of time losses has not been included in the cost calculations”. In principle, this kind of argument can be extended to cover behaviour that, while not actually forbidden by law, is deemed socially undesirable from the perspective of halting climate change. This argument is debatable, however, from the angle of welfare economics, which rests on the premise that all preferences hold equal weight (consumer sovereignty), even those that are collectively regarded as less desirable.

A third issue is possible compensation of welfare losses by recycling charge revenues back to users. After all, if these revenues are used to lower another charge or tax, the loss of consumer surplus which resulted from the other tax or charge is reduced (see § **Error! Reference source not found.**).

In most studies, direct welfare effects are not included (see, for example: Decicco & Ross, 1996; IIASA, 1998, 2005; IEEP *et al*., 2005; Johansson & Aahman, 2002; TNO, 2006). Certain other studies do include (some of) these welfare effects, though. Standard & Poor’s DRI & K.U. Leuven (1999) and ZEW (2006), for example, use the economic transport model TREMOVE to calculate changes in consumer and producer surplus. RIVM/CE (2004), in contrast, put a figure on the welfare effects using the ‘rule of half’, whereby the welfare effect is estimated as *half* the price rise times the volume reduction, as illustrated above in Figure 10.

Finally, it should be noted that combinations of policy instruments may reduce the welfare costs which would arise due to a single instrument. For example, the welfare losses due to road charging diminish if the quality and availability of alternatives, i.e. public transport, improves. Therefore, policy instruments are often combined such as in the case of transport policy in the city of London.

2.5.2 Direct effect: positional goods

A separate issue is the discussion around so-called positional goods. Various authors argue that consumption is driven not only by absolute needs, but also by *relative* needs, or in other words the human need to distinguish oneself from others or, alternatively, to identify with them (Mill, 1848; Veblen, 1898; Easterling, 1974; Hirsch, 1976; Mishan, 1981; Frank, 2005; Grinblatt *et al*., 2005). It is above all conspicuous goods that are suitable for the purpose of distinguishing oneself, and numerous authors cite car ownership as a characteristic example (see, for instance: Verhoef & van Wee, 2000; Steg, 2005; Carlsson *et al.,* 2006; Litman, 2007). From this perspective, a person’s choice of car is to a large extent determined by the choices made by others. Conversely, though, it also means that one’s own choice affects other people’s welfare. In the drive for positional distinction, the status derived by one person from his or her new (bigger or flashier) car goes at the expense of the status derived by others from their vehicle. In a sense, then, we here have a zero-sum game. From society’s perspective, the purchase of a ‘superior’ car merely entails a redistribution of welfare rather than creation of new welfare. Because those buying such a vehicle are seeking to improve their own status, but not necessarily to undermine that of others. Verhoef & van Wee (2000) categorise the negative effects of positional consumption as negative externalities. Because externalities signal a form of market failure, these authors argue for an ‘exclusiveness tax’. This kind of tax on high-status goods was already proposed by John Stuart Mill in his ‘Principles of Political Economy’ (1848, Bk V, Ch. VI):

‘A great portion of the expenses of the higher and middle classes in most countries, and the greatest in this, is not incurred for the sake or the pleasure afforded by the things on which the money is spent, but from regard to opinion, and an idea that certain expenses are expected from them, as an appendage or station; and I cannot but think that expenditure of this sort is a most desirable subject for taxation. If taxation discourages it some good is done, and if not, no harm; for, in so far as taxes are levied on things which are desired and possessed from motives or this description, nobody is the worse for them.’

Because positional consumption is a zero-sum game, policy instruments to downsize the vehicle fleet as a whole lead to no more than a limited loss of welfare. This holds whether the fleet average is made more fuel-efficient (and smaller) or large, inefficient models are banned. In the first case people have just as much opportunity to distinguish themselves, as status derives only from the *difference* from the fleet-average model. As the average becomes more modest, people will not need such a large car to give them their perceived status. In the second case too, though, the welfare effects are limited, because positional consumption is essentially a zero-sum game. A similar line of reasoning can be pursued in relation to the safety aspect. In a fleet of small cars, an SUV, with its relatively greater height, commands a better view of the road and, by merit of its relatively greater weight, increased safety. Here again, though, the benefits enjoyed by the owner are at the expense of other road users.

At the same time, of course, it cannot be denied that a large car often provides greater comfort than a smaller model in absolute terms, too. It is consequently difficult to assess the extent to which the greater willingness to pay for larger vehicles is driven by the expectation of relative or absolute benefits. Apart from Carlsson *et al.* (2006), we know of no studies in which these effects have been quantified (*cf.* IPCC, 2007). For the sake of completeness we report the empirical study by Kooreman & Haan (2006), who found that a new registration plate increased the sales value of a car by 4%, without there being any additional intrinsic value.

More broadly, we know of no studies on the cost effectiveness of climate abatement options nor policy documents reporting on this topic that include the aspect of positional consumption. Although the degree to which positional preferences determine the willingness to pay for cars is unknown, it is important to note that such preferences are not restricted to the top segment of the market. In the bottom and middle segments, too, the perception of private cars is very much governed by the existence of more expensive models. It is worth noting, finally, that when it comes to the role of status considerations in car purchases, people generally rank them far less important in their own case than for others (Johansson-Stenman & Martinsson, 2006).

2.5.3 Indirect effects

Indirect effects are the effects of a policy instrument on parties other than those directly affected. In many cases the indirect effects will often consist of a redistribution of the direct welfare effect, but they may also involve an extra (positive or negative) welfare effect. Indirect effects are only deemed to be a welfare effect if:

1. the effects of the policy instruments impinge (partly) on foreign countries;
2. distortions arise in associated markets (such as the labour or capital market) affected by the technology or policy.

One familiar example of an indirect effect are the (potential) employment effects of environmental policies. Another class of indirect effects that is sometimes included in CEAs are impacts occurring in the supply chain. In most cases, however, these affect only the denominator of the cost effectiveness ratio and are not expressed in terms of costs.

One indirect effect we would like to highlight here: the indirect effects of pricing instruments on the distortional effect of other taxes, such as the tax on labour. Taxes distort markets if they affect production and consumption decisions. Such distortion arises because the consumer price does not correspond with (is higher than) the price received by the supplier. In such cases the market mechanism fails to match supply and demand in such a way that the marginal value of the good equals the marginal costs. The market equilibrium for taxes is therefore suboptimal, leading to a loss of consumer and producer surplus.

Climate policy instruments may either intensify or reduce this distortionary effect of taxes. The former is the case if they require an increase in taxes, for subsidies, say. If tax rates go up to cover subsidisation of fuel-efficient cars, for example, there is an additional loss of welfare. The distortionary effect of taxes will be reduced, on the other hand, if the revenues of an energy tax are used to cut income tax rates, say (*cf.* IPCC, 2007). In the literature this latter phenomenon is known as the ‘relative Double Dividend’: redistribution of environmental taxes by means of lower tax rates is better for the economy than lump-sum redistribution, as the latter does nothing to reduce the distortionary effect of the tax.

In other words, if behavioural changes are enforced by means of pricing instruments, the loss of welfare resulting from those changes can be partly, or even more than fully, compensated by the welfare gains ensuing from the reduction in other distortionary taxes. Opinion on this issue is divided, though. Based, among other sources, on a study by de De Mooij (1999), CPB (2005: 15) concludes that road traffic pricing does not offer both dividends at the same time. Nonetheless, it is argued by a number of authors in recent publications that reducing other distortionary taxes will lead to substantial gains in welfare, thereby considerably reducing the costs of transport climate policy. See, for example: Parry & Bento (2001), Parry *et al*., (2004), Parry (2006), Mayeres & Proost (2001), West & Williams (2004, 2005, 2007), Austin & Dinan (2005) and Kleit (2004). The impact of climate policy instruments on distortionary taxes are included by both ZEW (2006) and Standard & Poor's DRI & K.U. Leuven (1999). To this end, both these studies make use of the TREMOVE model (Proost *et al*., 2006), which factors in the difference in efficiencies between taxes. In the other studies reviewed, however, little if any consideration is given to indirect effects.

2.5.4 Transaction and information costs

Transactions costs include the costs of obtaining and interpreting information as well as any costs associated with implementing energy efficiency opportunities including the costs of negotiating, implementing and enforcing contracts. In the context of energy efficiency, the costs of obtaining and interpreting information can be particularly problematic where energy is a small part of the overall budget and items are purchased for attributes other than their energy characteristics (MMA, 2008).

Implementing energy efficiency solutions also include such 'hidden' transactions costs, e.g. time taken to arrange and supervise work, disruptions while work is occurring and so on. Such costs are easily ignored in analyses of the benefits from energy efficiency improvements and can lead to an exaggeration of the expected net benefits from implementing energy efficiency opportunities. On the other hand, indirect benefits in the form of reduced exposure to price volatility, enhanced supply security and 'feel good' or reputational benefits can easily be ignored too and may lead to underestimates of the benefits from implementing energy efficiency opportunities.

2.5.5 Monetisation of other externatlities

Policies designed to reduce the climate-damaging emissions of the transport sector also often have an impact on other transport externalities. External effects, or externalities, are effects on the welfare of others not taken into account by the party causing the effects: they were *external* to, i.e. outside, the framework adopted when deciding whether or not to take the action in question. Externalities thus contrast with internal effects, i.e. those arising as a result of market transactions. Although in this case, too, there may be unintended effects on the welfare of others, by way of price mechanisms some allowance is still made for them.[12] Typical examples of externalities in the transport context besides climate-damaging emissions include congestion, noise nuisance, road safety impacts and emissions of NO_x and particulates.

A transport policy instrument designed to reduce aggregate vehicle mileage, for example, will reduce not only CO_2 emissions but other externalities as well. The question is then, first, whether these side-effects can be translated sufficiently reliably into financial terms and, second, whether they should be factored in when estimating the costs of the abatement options.

[12] The fact that externalities can influence real markets does not mean the effects are thereby internalised. The noise nuisance caused by Schiphol Airport translates to relatively lower land and property prices in the surrounding area, which means that residents are to a certain extent financially compensated for that nuisance. Because the airport is not the owner of the land, however, but in many cases the government, Schiphol makes no allowance for this loss of value. The noise nuisance thus remains an external effect.

There is a very extensive literature on the monetisation of transport externalities like road congestion, pollutant emissions, noise and traffic injuries and deaths. Following a general discussion of the various methodologies for costing such effects, the recently published Dutch SCBA Guidelines for environmental policy (CE, 2007) go on to state that in the case of side-effects having an impact on existing policy areas, valuation should preferably be carried out on the basis of prevention costs. For side-effects in areas where there is no standing policy, calculations should be based on damages. The CBA Guidelines for infrastructure projects, dating from 2000, include a list of values for externalities, although these are not intended to serve as a concrete standard (OEEI, 2000: p.219, based on ECMT (1998), p.73, Table 9).

Based on a literature study, in 1999 and 2004 CE Delft presented estimates of prevention costs for a variety of effects, which were considered sufficiently robust by several institutes for use in quantitative calculations. See, for example, several studies by CPB, among them 'Economic assessment of the [Dutch government's] Mobility Policy Document' (2004).

Internationally, too, a growing number of CBAs and CEAs are being performed in which environmental effects are monetised. The UK Department for Transport, for example, in its 'Guidance on Value for Money', has published recommended values for transport CO_2, NO_x and PM_{10} emissions (DfT, 2006). In 2001 the European Commission initiated the Clean Air for Europe (CAFE) programme for technical analysis and policy development in support of its *Thematic Strategy on Air Pollution under the Sixth Environmental Action Programme*. In this programme environmental effects are also monetised (AEA, 2005). Finally, the ExternE project deserves mention, set up in 1991 by the European Commission in collaboration with the US Department of Energy, with the aim of assessing the external costs associated with various fuel cycles. This project has yielded a substantial body of literature providing monetary valuations of environmental emissions (http://www.externe.info/).

It should be noted that according to a recent study commissioned by the European Commission DG TREN (CE Delft, 2008) the external costs of transport are much more determined by congestion and accidents than by noise and air pollution.

There are others, however, who consider the monetisation of externalities still too wrought with uncertainty for inclusion in CEA and prefer to cite these effects *separately* and in non-monetised form when discussing policy instruments. This is the approach adopted in the 'Option Document on Energy and Emissions, 2010/2020' (ECN & MNP, 2006), although this was not the case in the 'Option Document on Transport Emissions (RIVM/CE, 2004).

Decicco & Ross (1996) disregard the (environmental) side-effects of climate policy, holding that these effects are insignificant compared with the direct effects of such policy. In other studies, too, such as CE (2005), ECMT (2006), IEA (2006) and S&T Consultants (2003), the side-effects of climate policy are ignored. Again, there are yet other studies where these effects are factored in. This holds for AEA (2001), Standard & Poor's DRI & K.U. Leuven (1999) and ZEW (2006), for example. In most cases it is the effects on air pollutant emissions that are then included.

The second question – to the extent that externalities can indeed be monetised – is how these are to be included. There are essentially two options, which may yield different results. As an illustration, consider an option costing € 1000 which leads to reductions of 10 tonne CO_2 and 10 kg NO_x. We assume monetary values of € 40 per tonne CO_2 and € 10 per kg NO_x.

1. The first possibility is to add (or subtract) the additional external costs to (or from) the other costs of the option. The cost effectiveness is then 10 tonne CO_2 for € 900 (€ 1000 minus € 100 NO_x abatement) = € 90 per tonne CO_2.
2. The second possibility is to regard the additional external costs as an intended effect in the service of a different policy objective and split the costs of the option over the various effects, in proportion to the monetary value of the effect, for instance. In the example, the costs of the option are then apportioned to the two effects in the split given by € 400 CO_2 abatement and € 100 NO_x abatement. For the cost effectiveness of the option *as a CO_2 option* this means 10 tonne CO_2 for € 800 (€ 1000 x 400/(400+100)) = € 80 per tonne CO_2.

In a recent study on CEA by the Flemish government the first option has apparently been adopted when there is no emission abatement target for the other environmental effects and the second option when there is such a target (VITO, 2003: 39). In the 'Option Document on Transport Emissions' (RIVM/CE 2004: 251-252) the second approach has been adopted. This was also the case in TNO (2006).

Nevertheless, the first option appears to yield more meaningful results. This becomes evident if we take the case of a climate abatement option with *negative* side-effects. Consider an option costing € 500 that leads to an abatement of 10 tonne CO_2, but also causes an additional emission of 40 kg NO_x. Assume once more that CO_2 is valued at € 40 per tonne and NO_x at € 10 per kg.

1. The first option is to add or subtract the additional external costs to or from the other costs of the option. The cost effectiveness is then 10 tonne CO_2 for € 900 (€ 500 plus € 400 extra NO_x) = € 90 per tonne CO_2.
2. In the second option the costs of the option are apportioned to the two effects, in the split given by € 400 CO_2 abatement and € 400 *extra* NO_x. For the cost effectiveness of the option *as a CO_2 option* this means 10 tonne CO_2 for € 250 (€ 500 x 400/(400+400)) = € 25 per tonne CO_2.

In this case the second option yields a cost effectiveness that is lower than the monetised value of CO_2, suggesting that this is a sensible option. Given its substantial negative side-effects, though, it would be anything but sensible from the broader perspective of society as whole.

3 Marginal abatement cost curves

3.1 Introduction

If the cost effectiveness is assessed of a wide variety of abatement options, the results can be integrated into a so-called marginal abatement cost curve (MAC curve or MACC), in which the options are arranged in order of their cost effectiveness.

In this chapter, we first discuss the various types of options available to abate emissions (section 3.2). Second, we discuss the general shape and set up of MACCs (section 3.3) and for what purposes MACCs are used (section 3.4). Section 3.5 discusses bottom-up and top-down approaches to the determination of MACCs, while section 3.6 discusses the pros and cons of these two approaches. Section 3.7 discusses some general limitations to the application of MACCs.

3.2 Types of abatement options

Basically, three types of options are available to abate emissions:[13]

- *Technical options*, which reduce emissions without altering the economic activity itself. Examples are more fuel-efficient engines, alternative fuels, low rolling resistance tyres or carbon storage.
- *Operational or behavioural options*, which change the activity but do not change the output or result of the activity. One could think of increasing the load factor of trucks: the same volume is transported, but with fewer trucks. Or one can think of fuel-efficient driving.
- *Demand or volume options*, implying the reduction of an economic activity. These options do not only include an absolute reduction in an activity, but also the switch-over to another activity. One could think of the modal switch from car to train, or vehicle downsizing from a sport utility vehicle to a small fuel efficient car, for example.

It is important to note that although a reduction in demand may seem the last option to look for in climate policy, i.e. when all technical and operational options have been exhausted, this view is unjustified. Demand options are *not* necessarily more expensive than technical or operational options. After all, in some cases, it may be cheaper to refrain from a certain economic activity (a specific cargo transport) than to reduce the accompanying emissions by technological options. This might be the case when a certain activity has only a very low (marginal) added value. However, demand options are generally difficult to identify and quantify, which might explain why they are generally omitted from cost-effectiveness analyses.

Moreover, it should be noted that the options listed above can seldom be taken in isolation. For example, a more fuel-efficient engine – a *technical* option – may increase the purchasing costs of a car leading to lower car sales, a *demand* effect. A more fuel-efficient engine may also result in an increase in mileage and thus fuel use, a *rebound effect* (Greening *et al*., 2000; UKERC, 2007: 36). Likewise, alternative fuels may be more expensive leading to a lower overall demand for fuel. Consequently, calculating the cost effectiveness of abatement options is a complex issue.

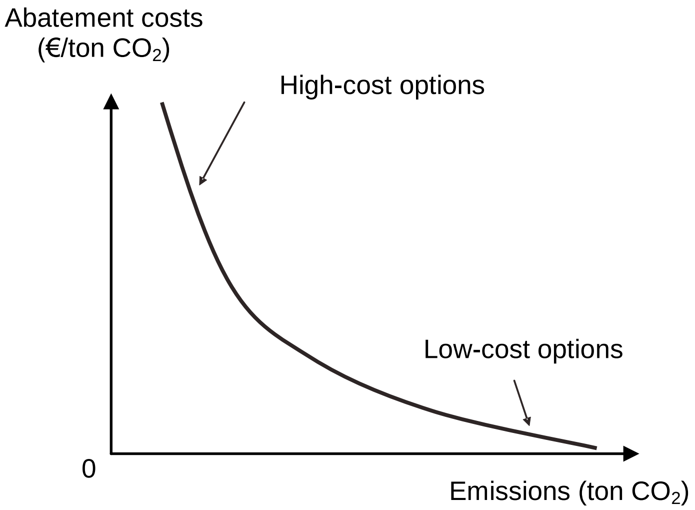
3.3 Marginal abatement cost curves

A marginal greenhouse gas abatement cost curve is a graphical presentation showing how the costs of *additional* greenhouse gas emission abatement increase the more emissions are already abated. Figure 8 illustrates the general shape of a MACC. The x-axis represents the emissions that remain after a certain level of abatement. The y-axis represents the marginal costs of the options, i.e. the

[13] Please note that these are all options which *directly* abate emissions. Indirect 'measures', such as policy instruments, are generally not included in MACCs. They could be included though if they are decomposed into the abatement options which are taken as a result of the policy instruments.

additional costs of additional abatement.[14] At the beginning, at business-as-usual emissions (the right side of the figure), emissions can be reduced against very low costs. However, the more emissions are reduced the higher the costs of additional emission abatement options become. Finally, when approaching zero emissions (the left side of the figure), the curve goes asymptotically to infinity.

Figure 8 Marginal costs of additional emissions abatement as a function of emissions

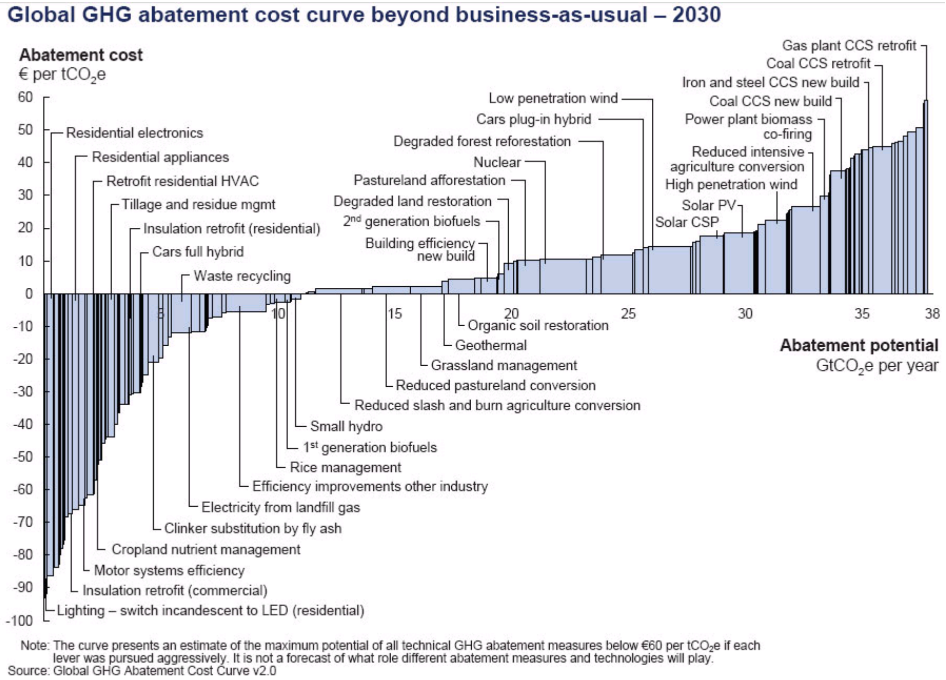


Discussion exists as to whether MACCs can intersect the x-axis, i.e. whether the first emissions can be abated against *negative* costs (IPCC, 2007: section 2.7.1.2). Some assume that many options exist which abate emissions by reducing fuel consumption and therefore save fuel costs. However, others believe that if abatement options exist with negative costs they would have already been implemented *without* policy. The fact that these options are not implemented without policy would imply that there are hidden costs, such as transaction costs (section 2.5.4) or welfare losses due to changes in behaviour (section 2.5.1). This topic has been discussed in the previous chapter.

In contrast to Figure 8, MACCs are often presented in graphs where the x-axis represents *abated* emissions instead of the emissions remaining *after* abatement. The curve is then mirrored and the point of zero emissions is omitted. Furthermore, MACCs may be presented with discrete abatement options instead of a smooth curve. In Figure 9 such a MACC is presented from the McKinsey report “Pathways to a Low Carbon Economy” (2007).

[14] The marginal cost curve differs from a normal cost curve in the fact that in the latter case the costs are *integrated*, i.e. at each point on the x-axis the total costs of all options are given which are required to achieve the emission abatement up to that point. The marginal cost curve gives the cost of the *additional* option only.

Figure 9 Marginal costs of additional emissions abatement as a function of emissions



3.4 For what purposes are MACCs being used?

MACCs are used for a variety of purposes.

First, policy makers wish to have insight into the whole range of abatement options and their costs so as to be able to identify the most cost-effective (cheapest) set of options available in society by means of which environmental targets can be reached. If these options have been identified, policy can be aimed at these specific options, for example by prescribing the best available technologies.

Second, policy makers wish to have insight into the level of emission abatement that can be achieved in society against a certain level of total or marginal costs, i.e. the emission abatement *potential* at certain costs. On the basis of an assessment of the costs, national or sectoral targets can then be set. If the costs of emission abatement are relatively low, more stringent, higher targets can be set.

Third, MACCs make it possible to determine the level of emission abatement that can be achieved in society by means of financial incentives such as emission or fuel charges. After all, it is to be expected that once a charge is introduced of say €10 per ton CO_2 all options are taken that cost less than €10 per ton (i.e. from an *end user perspective*!).

3.5 Bottom-up and top-down approaches

There are basically two approaches to determine MACCs: *bottom-up* and *top-down*.[15] In a bottom-up approach MACCs are composed by means of detailed inventories of individual technical options, their potential and their costs. This is also called the *engineer* approach.

In a top-down approach MACCs are not determined on the basis of knowledge about individual options, but knowledge of aggregate economic variables, which describe the economic system, such as demand for goods and services, and the supply from main economic sectors. Top-down models

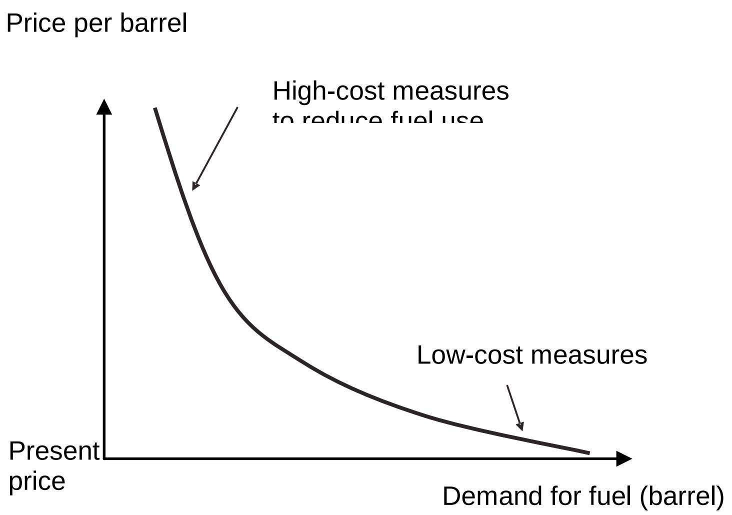
[15] The distinction between these approaches has been thoroughly discussed in chapter 8 of the contribution of Working Group III to the Second Assessment Report of the IPCC (1995).

examine a broad equilibrium framework. This framework addresses the feedback between the energy sector and other economic sectors, and between the macroeconomic impacts of climate policies on the national and global scale (IPCC, 2001: section 7.6.3).

The simplest example of a 'top-down' approach is the use of a (point) price elasticity for fuel demand. An elasticity of -0.6 means that if fuel prices rise by 10% fuel demand *decreases* by 6% (see also van Essen at al, 2009). See Figure 10 for the general shape of fuel demand as a function of fuel price based upon a single (point) price elasticity. Price elasticities give an indication of the costs of emission abatement. The reason is that consumers and producers have to choose between the costs of options to reduce fuel consumption and the costs of higher fuel prices, and emission abatement generally can be seen as an increase in the costs of fuel use. For example, at fuel prices of €2.00 per litre and a CO_2 content of 2.4 kg per litre of diesel, a tax of €100/ton CO_2 would result in a diesel price increase of €0.24/litre or 12%. If the price elasticity of fuel demand is -0.6, such a tax would result in about 7% CO_2 emission abatement. If a tax of €100/ton CO_2 would result in about 7% CO_2 emission abatement, this means that the marginal emission abatement costs to obtain an emission abatement of about 7% are €100/ton CO_2. At a 14% abatement, they are about €200/ton CO_2.

Please note that using price elasticities automatically implies an end-user perspective. As explained in section 2.3, an option which may seem cost effective from the point of view of the end user may not be so from a social perspective. For example, fuel savings include savings on fuel taxes, which are no savings from a societal perspective.

Figure 10 Fuel demand as a function of fuel price



3.6 Top-down *versus* bottom-up approaches

Principally, bottom-up and top-down approaches should yield the same MACCs. In practice, however, the differences in approach lead to quite different results. Each approach has its advantages and disadvantages.

First, top-down approaches are based upon empirical data relating to how economic systems, producers and consumers actually have responded in the past, i.e. *revealed* behaviour. Therefore, top-down approaches are less vulnerable to *wishful thinking* or too much technological optimism (or pessimism: see section 2.4.4). On the other hand, a criticism against top-down models has been precisely that because they are based upon historical data they are unable to handle the (radical) system changes that are required to tackle climate change.

A second advantage of top-down approaches is that they more easily incorporate various economic costs and benefits of policy. Examples of costs which bottom-up approaches handle with much more difficulty are transaction costs (section 2.5.4) and welfare losses due to behavioural changes (section 2.5.1). Such costs are automatically included in top-down approaches.

Consequently, bottom-up models usually tend to suggest that mitigation can yield financial and economic benefits, depending on the adoption of best-available technologies and the development of new technologies. Conversely, top-down studies have tended to suggest that mitigation policies have economic costs because markets are assumed to have adopted all efficient options already (IPCC, 2007: section 2.7.1.2).

A major disadvantage of top-down approaches, however, is the fact that insight is lost about the potential of individual technical options. It should be noted though that the traditional distinction between ‘bottom-up’ (engineering) and ‘top down’ (macro-economic) models is becoming increasingly blurred as ‘top down’ models incorporate increasing technology detail, while ‘bottom up’ models increasingly incorporate price effects and macro-economic feedbacks, as well as adoption barrier analysis, into their model structures (IPCC, 2007: section 2.7.1.2).

3.7 Limitations

In spite of their usefulness, there are of course limitations to the use of MACCs as well. GHG abatement costs do not cover all economic, societal and political motives that lead to a decision to apply or promote a certain option. This is illustrated, for example, by the fact that the EU policies on biofuels and on CO_2 emissions of passenger cars enforce the application of GHG abatement options which have significantly higher abatement costs than the value of CO_2 credits under the European emission trading scheme (ETS). Apparently the benefits of these options with regard to e.g. energy security and agricultural policy have a value that is not factored into the generally used definition of GHG abatement costs.

Furthermore, as has become apparent in the previous chapter, different points of departure in the calculation of cost effectiveness lead to very different results, which implies that one has to be careful with the conclusions drawn from cost-effectiveness analyses. Often, only the order of magnitude (is it -10, 0, 10, 25 or 100 €/ton?) can be considered significant or meaningful.

4 Conclusions and research needs

4.1 Conclusions

This study examined why studies to assess the cost effectiveness of policies addressing the climate impact of transport have yielded such widely different results to date. Our analysis of the costing methodologies in use shows there are three types of choice having a major influence on results. The first concerns the perspective adopted. Are costs being considered from the perspective of the end user, society or government? Secondly, there are a series of choices to be made in calculating direct expenditures, with respect to depreciation rates and prior estimates of investments, among other things. Finally, there is a basic choice as to whether only direct expenditures are to be included, or a comprehensive welfare-economic analysis carried out. Are the welfare effects of behavioural change or additional externalities to be included, for instance? The conclusions are the following:

4 Particularly in the transport sector, the cost effectiveness of an abatement measure can be very different when assessed from the perspective of the end user or that of society as a whole. This is first of all because measures designed to reduce vehicle fuel consumption also affect the flow of tax revenue from road users to government, and when it comes to transport, fuel duty and other taxes make up a substantial proportion of total costs. From the perspective of the end user, savings on these costs definitely count and should be included, while from the perspective of society as a whole they do not. Secondly, climate policy measures that reduce the aggregate annual mileage of the vehicle fleet also have a substantial impact on the overall welfare of society, because they also reduce other externalities (such as air pollution and noise), which should be included from society's perspective but not from the end user's. Although the choice of perspective adopted in analysing the transport sector has a major impact on results, the choice in itself is *unproblematical*. Generally speaking, researchers and policymakers clearly distinguish that the two perspectives serve different purposes and that results cannot therefore be compared. Consequently, many studies present results for both the end user's and society's perspective.

5 The pivotal items in any calculation of cost effectiveness, whether from the end user or social perspective, are the direct expenditures associated with implementing the measure in question, in other words the capital costs, operating costs (including costs due to changes in fuel use) and regulatory costs. In this study we have examined in more detail three choices that influence calculations of direct expenditures. Are calculations based on costs ex-works or on end user (i.e. retail) prices? What baseline scenario is used, with respect to fuel price trends, for example? And how are cost price trends for new technologies estimated? The choices made with respect to these issues are found to have a major impact on estimates of direct expenditures.

6 In many national and international studies the cost effectiveness of environmental measures are calculated on the basis of direct expenditures only. However, a growing number of reports are appearing, in both policy and research circles, in which a comprehensive welfare-economic analysis is recommended. In this kind of analysis it is not only direct expenditures that are regarded as costs, but also losses of welfare associated with enforced behavioural change, the indirect costs of the measure, and additional externalities, i.e. other than those the measure is designed to reduce. This kind of analysis has been carried out for a number of individual transport policy measures. Studies in which the cost effectiveness of a wide range of measures are compared from a broader, welfare-economic angle are rare, though. There may be two reasons for this. First, a welfare-economic analysis is more complex and thus time-consuming than an analysis of direct expenditures. This is obviously a problem if a large number of measures are to be assessed. Second, the costs and possible benefits that a welfare-economic analysis adds to an analysis of direct expenditures follow from derivative calculations and models and are consequently more open to debate. There are two extra 'cost items' in a welfare-economic analysis that may make the outcome very different from a calculation on direct expenditures only:

   c In the realm of transport, particularly, climate measures have a substantial impact on other externalities, too. Measures to cut vehicle fuel consumption reduce not only CO_2 emissions but also those of NO_x and particulates, for example. Measures to reduce aggregate annual mileage affect not only emissions but also noise, congestion and the number of road traffic injuries and deaths. As the majority of studies take most of the cited external effects to be broadly similar in terms of importance to society, whether or not the impact of a measure on these externalities is included in calculations of cost effectiveness is of major influence on results.

d Measures to reduce aggregate annual mileage or fuel consumption often mean an enforced change in behaviour: without the measure, people would have driven more kilometres or bought a different kind of car. If only direct expenditures are included, these kinds of measures would be all profit and no loss. After all, those choosing not to make a particular journey or buying a smaller car are left with more money in their pocket. In a welfare-economic analysis the conclusions look rather different, though. Not being able to do something that one would have preferred to do constitutes a loss of welfare. This loss can be expressed in monetary terms, with reference to a price incentive, for example. Such studies show that because of the already relatively high taxes on car ownership and use, additional cuts in transport volumes will be associated with high costs to society. An alternative perspective is to see the currently high costs of car ownership and use as a means of pricing negative transport externalities. In that case, to the extent that the negative externalities of transport are already priced and internalised, additional regulations can no longer bring about an increase in welfare, and may even lead to a loss. However, various arguments can be cited as to why this loss of welfare may well be less pronounced than appears at first sight from a welfare-economic analysis:
   - First, much of people's transport behaviour is conditioned. What was estimated beforehand (*ex ante*) to constitute a loss of welfare, proves subsequently (*ex post*) to be far less problematical (for consumer and researcher alike).
   - Second, the fact that people buy 'gas-guzzling' vehicles has to do with *relative* consumption. People derive personal welfare from having a bigger car than their neighbours. Policies that impinge on the entire vehicle fleet will leave relative consumption unaffected, however, thus causing less loss of welfare than originally anticipated.
   - Third, there is the objection that, as a matter of principle, an inability to engage in consumptive behaviour deemed socially undesirable should not be included as a cost item in calculating policy costs.
   - Fourth, in the case of transport pricing measures, the welfare effects can be partly offset by using the revenue to reduce other, distortionary taxes like income tax. There is a growing body of literature that argues on these grounds that pricing measures in the transport sector are particularly cost-effective.

Top-down approaches are generally better equipped to handle the welfare effects described above. A major disadvantage, however, is the fact that insight is lost about the potential of individual technical options.

4.2 Recommendations

One should be careful with the conclusions drawn from cost-effectiveness analyses. First, as has become apparent, different points of departure in the calculation of cost effectiveness lead to very different results, which implies that often only the order of magnitude (is it -10, 0, 10, 25 or 100 €/ton?) can be considered significant or meaningful. Second, it should be noted that GHG abatement costs do not cover all economic, societal and political motives that lead to a decision to apply or promote a certain option, such as distributional motives and energy security.

When results from cost-effectiveness analyses are applied or compared, it is therefore recommended always to look carefully at the explicit or implicit assumptions regarding oil prices, interest rates of chosen perspective (society, end user).

Finally, it should be noted that cost-effectiveness analysis is presently more suitable for short-term than long term effects. This is particularly the case when policy measures have far reaching consequences for spatial, social and economic structures. In the long term, the palette of options to reduce emissions widens, but becomes also more unpredictable. Such options include unexpected technological breakthroughs, ‘spatial adaptation’ to new transport modes and prices, and habituation. Policy restrictions to already available options for consumption and transportation may cost more welfare in the short term than in the long term, when preferences adapt to the new situation. It is recommended that more research is performed in this area.

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