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Climate Policy Costing Methodologies

A comparative analysis for the transport sector

Final report

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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. To this end, experts in the Netherlands were consulted and the national and international literature reviewed. 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?

Conclusions

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 vehice 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 the Dutch environment ministry's 'Environmental Costing Methodology Manual', drawn up in 1994 and updated in 1998, it is recommended that the cost effectiveness of environmental measures be calculated on the basis of direct expenditures only. This is the approach adopted in many national and international studies. 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. In the Dutch context, the Option Document on Transport Emissions forms an exception here. Studies comparing different transport measures are thus generally based solely on analysis of direct expenditures. 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 lead to this kind of study yielding very different results:

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. It should be noted, though, that these are 'minority viewpoints':

- 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 vehice 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.

Recommendations

As set out above, we do not regard as problematical in itself the fact that differences in perspective, i.e. the end user, or society as a whole, leads to differences in the results of cost effectiveness analyses. Researchers and policymakers generally make it very clear that the two perspectives serve different purposes and consequently yield resuts that cannot be compared.

Nor do we have any concrete recommendations concerning the differences in the results of such analyses that arise through the different choices made in calculating direct expenditures. It holds for any individual cost effectiveness analysis that it is up to readers and users to determine whether the basic choices and assumptions made are sufficiently and convincingly underpinned. In the Dutch context, at any rate, we see no fundamental differences in the approaches adopted by researchers.

With regard to the distinction between cost effectiveness analysis based on an analysis of direct expenditures and a more comprehensive welfare-economic analysis, we have several recommendations, specifically for the Dutch context.

The ministry's 'Environmental Costing Methodology Manual' (1994, updated 1998) explicitly recommends that only direct expenditures be included in assessing the cost effectiveness of environmental policy measures. More particularly, it recommends, first, not to monetise additional externalities resulting from the measure in question and, second, not to include welfare effects due to behavioural change.

In the intervening period, however, numerous national and international studies have been published that recommend adopting a welfare-economic analysis that does include these kinds of effects. Particularly in the light of the recently published Dutch ‘Guidelines for Social Cost Benefit Analysis’, in which a welfare-economic analysis is likewise recommended, a new update of the ‘Environmental Costing Methodology Manual’ would appear to be warranted. A second motive for an update is that when the Manual was originally drawn up, environmental policy was focused more on prescribing specific technologies. In today’s environmental policy, in which economic instruments and incentives for behavioural change play such a key role, there is an even greater need for a more comprehensive welfare-economic analysis. We see the following changes to the Manual as crucial:

– The current recommendation not to consider welfare losses due to behavioural change as costs and to exclude these from the analysis should be revised and recommendation given to monetise such losses of welfare unless there are reasonable grounds for deeming analysis on this point unnecessary. The latter will be the case for regulations on many concrete energy-saving technologies. It should also be recommended that inclusion of welfare losses due to behavioural change in the reported cost data be explicity mentioned. In addition, the calculation methodology employed should be clearly and transparently explained to data users, making clear what has and has not been included, and the effects of these choices on the final results.
– Instead of the current recommendation not to monetise additional externalities, it should be recommended to do so, including guidelines for that purpose (based on the recent ‘Guidelines for Social Cost Benefit Analysis’).

There are currently various institutes that consider the financial valuation of externalities already sufficiently robust for use as a basis for policy calculations. Others consider these estimates still too uncertain, though. For the sake of policy consistency and in the light of the aforementioned Guidelines, we recommend that discussions be held about the desirability and feasibility of extending these Guidelines to include a list of recommended monetary values for key externalities, for use in both social cost-benefit analysis and cost-effectiveness analysis.

1 Introduction

The Dutch government has set itself ambitious targets, both nationally and internationally, for reducing concentrations of the greenhouse gases contributing to climate change. The targets are ambitious in the sense that securing them will involve quite substantial costs to society. It is therefore important they be achieved as cost-effectively as possible, not least with a view to retaining public support. More specifically, the goal must be to implement those measures that come at the lowest cost per avoided unit emission.

Calculating the costs of alternative policy measures is anything but straightforward, however, one reason being that the concept of 'cost' is subject to a variety of interpretations, each of which leads to a different assessment of the respective options. While one method may lead to the conclusion that a certain measure is prohibitively expensive, a second method may point to the same measure delivering benefits that outstrip the costs. In the realm of transport, particularly, the various approaches and methods often prove to lead to vastly differing conclusions. In many studies concerning the cost effectiveness of climate policy measures, transport often emerges as the one sector where money can in fact be made, while in other sectors major costs will be incurred.

To gain greater insight into the various methodologies and how they affect assessment of abatement measures, the Netherlands Environmental Assessment Agency (MNP) commissioned CE Delft to carry out a systematic analysis of the methods used nationally and internationally for calculating cost effectiveness.

Report structure
In Chapter 2 we review and discuss the principal choices that go to explain the differences in results between the different calculations of cost effectiveness of abatement measures in the transport sector. In our analysis we distinguish three clusters of influential choices. The first concerns the perspective of the methodology. Are the costs considered from the perspective of the end user, society as a whole, 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. Does the analysis include the welfare effects of behavioural changes or additional externalities, for example?

In Chapters 3 and 4 two case studies are discussed that illustrate the influence of various choices on the cost effectiveness of abatement measures. Chapter 3 looks at introduction of a mandatory speed limiter for light commercial vehicles. Chapter 4 examines European policy to reduce the CO_2 emissions of new passenger cars to 130 g/km in 2012. In Chapter 5 we present our conclusions and recommendations.

2 Analysis of costing methodologies

2.1 Introduction

In this chapter we review and discuss the principal choices that go to explain the differences in the results obtained by the various calculations of policy cost effectiveness in the transport sector. 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. For a full review of a conventional CEA the reader is referred to the Dutch environment ministry's 'Environmental Costing Methodology Manual' (VROM, 1994, 1998), the recent report 'Environmental policy costs: Concepts and calculation methods' published by the Flemish government (LNE, 2007 and background report VITO, 2003) and the 'Guidelines for Social Cost Benefit Analysis' (CE Delft, 2007)[1].

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 2.2. 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 2.3. 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 2.4.

Many of these choices are influenced by whether it is technologies or policy measures that are the subject of the cost effectiveness analysis. Traditionally, CEA was developed to evaluate the cost of technologies in relation to their impact, providing information on which technologies are cheapest for achieving set targets, a useful tool for policymakers. However, a CEA of technologies cannot provide a full picture of the total costs of a given policy measure. To start with, the policy itself brings with it a certain number of costs. In addition, it is often unclear whether a policy is being elaborated by *technical* or *organisational* means. While a kilometre charge indexed to vehicle category may mean costs being incurred for vehicle replacement, for instance, it may also cause a shift in demand for transport mobility. A CEA of policy options thus needs to be designed differently from a study of the cost effectiveness of technological measures.

It should be noted that in this study it is choices affecting the costs of climate measures that are reviewed. Cost effectiveness is obviously also influenced by the degree of success in securing the envisaged *benefits*: here, cuts in carbon emissions. In this report, however, choices affecting the calculated magnitude of carbon emission reductions have been ignored, such as whether or not to include

1 For ease of reference we have provided English titles for key Dutch-language documents mentioned in this report; in each case the original title is cited in the References section.

the rebound effect, decisions on emission scenarios, and correction for double counting of benefits when various measures and policies are combined.

2.2 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 measure 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 2.2.4.

2.2.1 The end user

From the perspective of the end user, the costs are calculated that are incurred by those directly affected by the environmental measure, i.e. from the perspective of companies, institutions or households. This perspective yields insight into how the costs of a given environmental policy 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 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 or discount rate for investments is generally higher than that from the perspective of society as a whole.

The end user perspective is adopted in numerous studies. In the Dutch context, it is first of all the perspective chosen in the 'Environmental Costing Methodology Manual' issued by the Dutch environment ministry (VROM) as a standard for defining and calculating the costs of environmental management (1994, 1998).[2] This was the methodology used in the government's 'Climate Policy Implementation Plan' (VROM, 1999)[3], in which the costs of policy measures on more fuel-efficient road vehicles, ditto driving behaviour and reduced transport mobility are defined at the end user level, and also in a broader study on the cost effectiveness of environmental measures by the National Institute for Public Health and the Environment (RIVM, 2000). Since then the VROM methodology has been used for a variety of purposes, including an assessment of the environmental costs accruing to individual target groups in the 'Environment and

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2 In 'Costs and benefits of environmental policy: Definitions and calculation methods' (VROM, 1998) it is stated under Section 1.3, 'Principles and scope', that: 'The methodology establishes the costs from the perspective of those affected by the environmental measures, i.e. from the perspective of companies, institutions or households.' For calculating the costs and benefits of energy conservation, the report also elaborates a perspective from society as a whole. In the relevant Section 5.6, though, this social (or national) perspective is explicitly distinguished from the 'Environmental Costing Methodology', which is definied as an end user approach (p. 50).

3 While employing the standard environmental costing methodology, this Implementation Plan also reports the national costs.

Nature Compendium', a joint publication by Netherlands Statistics (CBS) and the Netherlands Environmental Assessment Agency (MNP) (*cf*. RIVM, 2001). The methodology is also referred to in several recent guidelines for policymakers, such as VROM (2004).

In the transport and environment context, the end user perspective has also 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. In its guidelines on environmental impact assessment, the European Commission (EC, 2005) also deals with the subject of cost effectiveness, referring explicitly to the methodology developed by VROM. The context is assessment of the costs accruing to various parties from alternative projects to achieve a pre-defined goal. 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 narrow end user perspective.

2.2.2 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 the focus is on impact on overall social welfare, irrespective of distribution effects. From the perspective of a given society:
- In principle, external (environmental) 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 measures 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 policy measures, 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 amd road safety) are left out of the picture. Including these other items in calculating the cost effectiveness of policy measures is discussed in Section 2.4.

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

2.2.3 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 incured 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 servies, consultancy, monitoring, training and so on (ECN/RIVM, 2004). In the second approach it is not only the apparatus costs that are considered, but also government subsidies (CE, 2005; MNP, 2007). Here, we are concerned with subsidy effectiveness; see, for example: FEM, 2007.

2.2.4 Discussion

There are a number of studies in which the analysis is carried out from both the end user and social perspective, as in the 'Option Document on Transport Emissions' (RIVM/CE, 2004) and 'Option Document on Energy and Emissions 2010/2020' (ECN & MNP, 2006), 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 consequenly little debate as to which is the 'right' one (*cf.* IPCC, 2007).

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 vehice 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 externalties, 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 measures in the transport sector are less cost-effective than measures 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 n 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 (1,000-1,500)/3 = *minus* 167 Euro/tonne CO_2: the measure ** leads to 'earnings'. From society's perspective, though, the cost effectiveness is (1000-500)/3 = *plus* 167 Euro/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; GoC, 2002), Australia (Abare, 2006), the US (CBO, 2003; EPA, 2003), the UK (HMT, 2003) and Sweden (Regeringskansliet, 2005).[4] 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.

2.3 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 policy measures in question. Before going on to

[4] Indirectly, it is 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%).

discuss other cost items in Section 2.4, in this section we first examine the choices affecting calculations of direct expenditures. First, in paragraph 2.3.1, we discuss these expenditures in a general sense. In the subsequent sections we then consider the following particular issues: Costs ex works versus end user prices (paragraph 2.3.2), Baseline scenario (paragraph 2.3.3) and *Ex ante* versus *ex post* cost estimates (paragraph 2.3.4). In Section 2.3.5 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. Thus, we do not discuss the important choice of depreciation period, i.e. discount rate. For a discussion of these issues, see: VROM, 1998 and CE, 2007.

2.3.1 Direct expenditures

The direct expenditures of climate policy measures 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 implementation of the policy measure. Although this generally means the purchase price or production costs of an 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 measure 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 climate measures 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 a climate measure, 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 climate measure 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 a policy measure 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 maintanence costs will go up or down. This is the main consideration for assuming zero maintanence costs in calculations. More broadly, regulatory costs are seldom included in analyses of cost effectiveness.

2.3.2 Costs 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.* (2005) 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; IIASA, 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 implictly 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).

2.3.3 Baseline scenario and baseline technology

Every cost effectiveness analysis should, in principle, compare the additonal costs of a given technology or policy 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 measures 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 policy 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 policy 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 measures in the transport sector, a business-as-usual scenario is generally used.

One key issue in assessing the cost effectiveness of transport climate measures 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 measures, in most studies a range of values is taken (see, for example: CONCAWE, 2007; IEEP *et al*., 2005; TNO, 2006). In contrast, there are also studies in which calculations are based on a single oil price (for example: S&T Consultants, 2003).

2.3.4 *Ex ante* versus *ex post* cost estimates

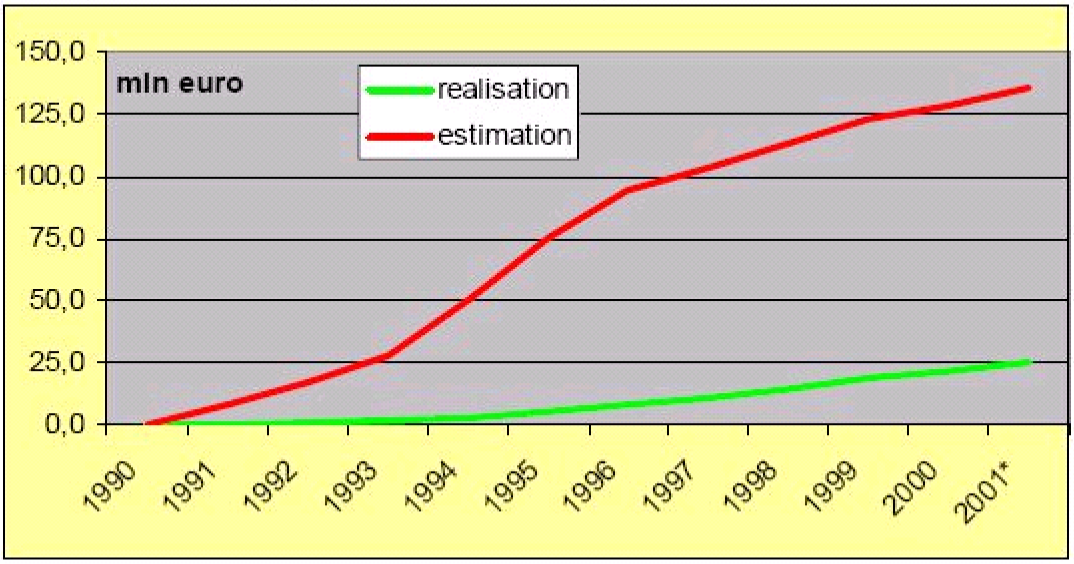
The costs of a given policy measure can be calculated prior to or after implementation: *ex ante* and *ex post*, respectively. When setting out future policy, on the basis of an Option Document, for example, *ex ante* cost estimates are generally required, unless the envisaged measures are already tried 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 policy 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 measures implemented in the past have shown that *ex ante* studies prior to implementation of these measures generally overestimated the costs (Burtraw, 1996; Stockholm Environmental Institute, 1999; Harrington, 2000; CE, 2006; IvM, 2006; TME, 2006). 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 (RIVM, 2000a). 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 1, 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 additonal fuel consumption, while *ex post* investment costs proved to be between € 130 and 240 per car, with no extra fuel consumption.

Figure 1 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 policy measures, particularly when it comes to new technologies like more efficient vehicle engines.

2.3.5 Discussion

We have discussed three choices that go some way to explaining why different cost effectiveness analyses can sometimes yield such widely differing results. Nevertheless, we see in these kinds of choices no reason for a broad debate among researchers, in the Dutch context at any rate. In any cost effectiveness analysis, the particular choices and assumptions made will always need to be closely argued. How specific decisions on cost or technology projections and other such issues are to be made is not something that can be laid down in advance. In the Netherlands we see no fundamental differences among the approaches adopted by researchers. What should be noted, though, is that because of the potentially varying assumptions employed in different studies, those using the data have a responsibility not to naively compare data from different sources if it is unclear whether they have been calculated according to the same methodology.

2.4 Direct expenditures versus welfare-economic analysis

In a comprehensive, welfare-economic approach, the overall cost of an emission abatement measure is given by the balance of all the welfare effects of the measure excluding the actual intended effect of the policy (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 measure, 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, 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 '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. A second motive was to achieve compatibility with then-standard practice at Netherlands Statistics (CBS) and the Netherlands Bureau for Economic Policy Analysis (CPB) in other fields. Thirdly, it may be noted that the 'Manual' was developed primarily for assessing tangibe technologies, where indirect and unpriced effects are limited. The approach adopted in the Dutch 'Manual' has been adopted elsewhere, for example in the guidelines of the European Environmental Agency (EEA, 1999). In the Netherlands' 'Climate Policy Implementation Plan' (VROM, 1999) and the 'Environmental and Nature Compendium', a joint publication by CBS and the Netherlands Environmental Assessment Agency (MNP), too, the analysis is restricted to direct expenditures (*cf.* RIVM, 2001).[5] 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, EPA, 2003; CBO, 2003; VITO, 2003; Verhoef *et al*., 2004; TML, 2006b; LNE, 2007). An SCBA in which the external benefits of climate measures are discussed but remain unvalued can be considered an extended variant of cost effectiveness analysis. In such studies the various welfare effects of climate measures are examined but

5 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.

no attempt made to translate the associated cuts in greenhouse gas emissions into monetary units.

For an extensive description of all the possible welfare effects of environmental policy, readers conversant with Dutch are referred to the 'SCBA Guidelines' for environmental policy (CE, 2007). Below, we discuss the more controversial effects involved in assessing the cost effectiveness of policy measures in the transport sector.

2.4.1 Direct effects: welfare losses and gains of behavioural change

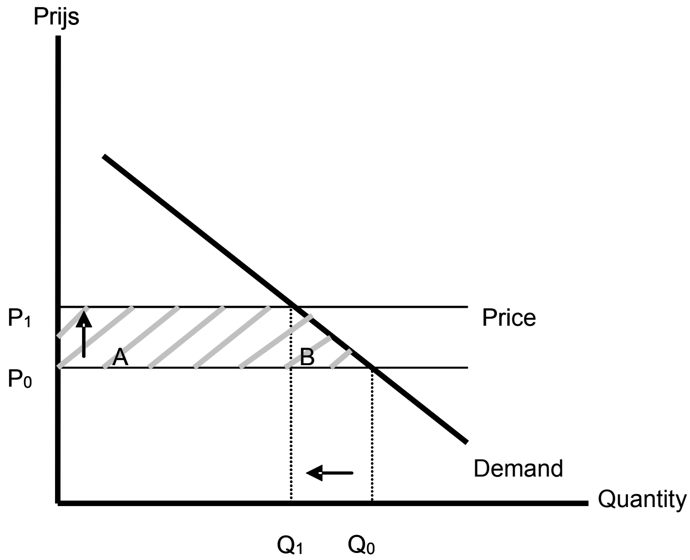
Besides leading to direct expenditures, a new policy measure may also induce behavioural change. For example, rather than incurring the extra costs implied by the measure, some people will opt to discontinue a particular form of environmentally damaging behaviour, as when a cleaner car is more expensive. With some measures, 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 measure 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). In assessing the factors determining people's choice of transport mode during rush hour traffic, AVV (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, figure 2 shows the loss of welfare resulting from road pricing, i.e. a kilometre-based charge (description based partly on CE, 2007; CPB, 2000). 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 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.[6] 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 2 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, 2004; CPB & V&W, 2004).[7]

Calculating the loss of welfare (loss of consumer surplus) associated with enforced behavioural change is a controversial issue, however. There are those

6 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.

7 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.

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 (paragraph 2.3.4).

There is, secondly, the principled argument that losses of welfare resulting from a person being unable to perform certain activitities 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 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. If these revenues are used to lower another charge or tax, the welfare losses occurring due to this other tax (a similar triangle B as in Figure 2) may be compensated (this is discussed in more detail in paragraph 2.4.3.)

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 in Figure 2.

2.4.2 Direct effects: 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 or 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, measures 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 measures 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 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 status considerations in car purchases, people generally rank them far less important in their own case than for others (Johansson-Stenman & Martinsson, 2006).

2.4.3 Indirect effects

Indirect effects are the effects of a policy measure on parties other than those directly affected. In many cases the indirect effects will often consist in 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 measure 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.

Traditionally, of course, CEAs encompass only primary (direct) costs, defined as costs incurred in an immediate and real sense by those implementing the environmental measures, with indirect effects (secondary environmental costs) not being included.[8]

[8] In ECN/RIVM (2004) it is noted that in assessing the relative performance of measures in different sectors it should be borne in mind that indirect effects may be of major relevance for the growth and structure of domestic production, through potential deterioration of international business competitiveness, for example, possibly leading to a decline in domestic production and transfer of activities abroad.

It is only logical that there is less focus on indirect effects in cost effectiveness analysis than in social cost-benefit analysis. After all, a CEA is usually concerned with measures that have (far) less of a society-wide impact, on labour and capital markets, for example, than the kind of 'mega-projects' for which social cost-benefit analysis is carried out. In their supplement to the 'Guidelines on the Overall Effects of Infrastructure', though, Oosterhaven *et al*. (2004: 10-11) state that even in the case of SCBA, a study of indirect effects will not usually be necessary, unless substantial indirect effects are anticipated:

'As the direct effects generally (far) outweigh the additional indirect effects, a partial CBA without indirect effects is in many cases sufficient to identify the principal benefits accruing at the national level. Only if the project is likely to have major indirect benefits or if such effects alter the sign of the cost-benefit balance will there be a need for an integral CBA that includes a thorough analysis of additional indirect effects. Further analysis of indirect effects may also be necessary if there is a desire to gain a better understanding of distributional effects.'

The possibility of indirect effects occurring should always be left open, however, and in any CEA there should be critical assessment of the relevance of such effects.

One indirect effect we would like to highight here: the indirect effects of pricing measures 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 measures 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 wil 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 measures, 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. Opinions on this issue is divided, though. Based, among other sources, on a study by de Mooij (1999), CPB (2005b: 15) concludes that road traffic pricing does not 'cut both ways'. In its calculations, though, CPB assumes that of tax revenues are redistributed via a reduction of Vehicle Circulation Tax rather than income tax. The motive for this is an assumption that it is only redistribution to the same group that pays the tax that will be deemed equitable. Because CPB takes the price elasticity of VCT to be minimal, the distortionary effect of this particular tax is only limited and recycling of revenues via VCT will therefore yield scarcely any welfare gains.

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 measures 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 (TML, 2006b), which factors in the difference in efficiencies between taxes. In the other studies reviewed, however, little if any consideration is given to indirect effects.

2.4.4 Monetisation of other externalities

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 accounted for in 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.[9] Typical examples of externalities in the transport context besides climate-damaging emissions include noise nuisance, road safety impacts and emissions of NO_x and particulates.

A transport policy measure 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 measure.

There is a very extensive literature on the the monetisation of transport externalities like 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

9 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.

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 Technology Environment, 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/).

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 measures. 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 insignifant 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 a measure costing € 1,000 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 option is to add (or subtract) the additional external costs to (or from) the other costs of the measure. The cost effectiveness is then 10 tonne CO_2 for € 900 (€ 1,000 minus € 100 NO_x reduction) = € 90 per tonne CO_2.
2 The second option 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 measure over the various effects, in proportion to the monetary value of the effect, for instance. In the example, the costs of the measure are then apportioned to the two effects in the split given by € 400 CO_2 reduction and € 100 NO_x reduction. For the cost effectiveness of the measure *as a CO_2 measure* this means 10 tonne CO_2 for € 800 (€ 1,000 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 reduction 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 measure with *negative* side-effects. Consider a measure costing € 500 that leads to a reduction 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 measure. 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 measure are apportioned to the two effects, in the split given by € 400 CO_2 reduction and € 400 *extra* NO_x. For the cost effectiveness of the measure *as a CO_2 measure* 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 measure. Given its substantial negative side-effects, though, it would be anything but sensible from the broader perspective of society as whole.

2.5 Worked example

To illustrate the distinction between the perspective of the end user and society, on the one hand, and between calculation of direct expenditures and a more comprehensive welfare-economic analysis, on the other, we here provide a simplified worked example. The question addressed is: What is the cost effectiveness of including transport in the European greenhouse gas emissions trading scheme (EU-ETS) or an additional 'climate charge' on motor fuel on top of today's excise duty, under the following illustrative assumptions?

1 The pre-tax fuel price is € 0.50 per litre.
2 Fuel duty amounts to € 1.00 per litre.
3 The climate charge or EU-ETS trading price is € 0.01 per litre (around € 4 per tonne CO_2).
4 There are *no* external costs due to issues like particulate emissions, traffic accidents or wear and tear of road surfaces.

Under these assumptions the cost effectiveness is shown in Table 1.

Table 1 Cost effectiveness

	Direct expenditures	Welfare economic analysis
End user	- 1.50 €/liter - 625 €/ton CO_2	0.005 €/liter 2 €/ton CO_2
Society	- 0.50 €/liter - 208 €/ton CO_2	1.00 €/liter 417 €/ton CO_2

From the perspective of the **end user** and a **welfare-economic analysis** the results are as one would expect: a marginal increase in costs is also experienced as a marginal loss of welfare. Only those motorists who derived little extra welfare from their fuel consumption anyway (private costs and benefits both roughly equal to € 1.50/litre) will now discontinue that consumption. Their loss of welfare is virtually zero.

From the perspective of **society** and a **welfare-economic analysis**, however, for every litre of fuel less that is consumed, the full € 1.00/litre tax is lost. This is because the number of litres of fuel less that is now consumed is precisely the amount that yielded € 1.50/litre welfare (for otherwise these litres would not have been sold without an increase in cost price), but for which the costs to society are no more than € 0.50/litre (the pre-tax price).

From the perspective of the **end user** and **direct expenditures** the end user *saves* the entire € 1.50/litre. Declining to buy a litre of fuel means nothing but gain, for the utility deriving from use of that fuel is not included in an analysis of direct expenditures.

From the perspective of **society** and **direct expenditures** society saves € 0.50/litre, for the taxes are obviously not saved.

Note once more that the figures used here are for illustrative purposes only and will change if the taxes are also deemed to serve as recompense for the external costs of particulate emissions, traffic accidents, wear and tear of road surfaces and other such issues.

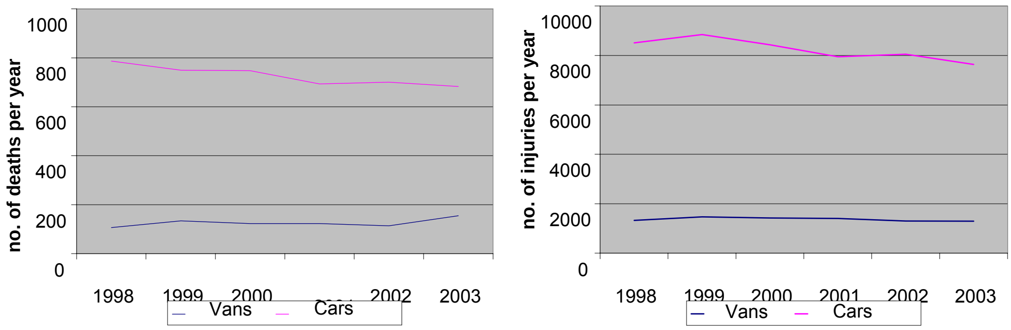
3 Case study 1: speed limiter on light commercial vehicles

3.1 Introduction

One possible measure for reducing road transport CO_2 emissions is mandatory fitting of speed limiters in light commercial vehicles (subsequently referred to as vans). At an average speed of 100 km/h, CO_2 emissions are over 15% lower than at 120 km/h (CE, 1998) and the potential contribution of this measure to CO_2 abatement was reason for including this option in the 'Option Document on Transpoort Emissions' (RIVM/CE, 2004).

The initial motive for the government to consider making speed limiters mandatory was not the potential CO_2 reduction that could thus be achieved, however, but the contribution of this measure to increased road safety (*cf.* AVV, 2004). The background to this is that the accident statistics for vans, in contrast to passenger cars, show no downward trend (Figure 3) and additional policy was therefore required.

Figure 3 Trends in deaths and injuries in Dutch traffic accidents involving cars and vans



Source: SWOV (2005)

Besides contributing to road safety and cutting CO_2 emissions, mandatory speed limiters in vans would also help reduce air pollutant emissions and traffic noise.

In this case study we have opted to take CO_2 emissions reduction as the stated policy objective. Our aim is then to determine the cost effectiveness with which this objective can be achieved by means of the cited measure, thereby identifying the factors of greatest influence on the calculated results. A major part of the analysis therefore consists of a series of sensitivity analyses, in which we examine the extent to which changes in methodology and assumptions affect the cost effectiveness ultimately found.

This case study is structured as follows: first we look in a little more detail at the measure for which the cost effectiveness is being assessed (Section 3.2). In Section 3.3 we then calculate the CO_2 emission reductions that can potentially be achieved by introducing a mandatory speed limiter for vans. The costs attending this measure are discussed in Section 3.4, and its cost effectiveness established in Section 3.5. In Section 3.6, finally, we carry out several sensitivity analyses to determine the extent to which various methods and assumptions affect results.

3.2 The measure

In this case study it is assumed that as of 2010 the European type approval requirements will include a provision that vans must be fitted with a speed limiter set to 100 km/h. The presumed aim of this move is to reduce the CO_2 emissions of these vehicles.

In assessing the cost effectiveness of the measure we take as our horizon the year 2010. All emission reductions and costs are thus calculated with reference to this year.

3.3 Reduction of CO_2 emissions

In order to calculate the CO_2 emission reductions to be achieved by making speed limiters mandatory for vans we must first of all know the emissions of these vehicles in the baseline situation (without mandatory speed limiters).

From the data reported in MNP (2006) we derive that the total annual mileage driven by vans on Dutch roads in 2010 will be 20.5 billion kilometres. Because of the very high share of diesel vans in this total mileage, in the following analysis we have chosen to ignore vans running on petrol and LPG, thus assuming that in 2010 vans will account for some 20.1 billion kilometres annually. However, it is only on roads with a speed limit of over 100 km/h that the speed limiter will effectively reduce speed, i.e. on motorways. The analysis is therefore further restricted to motorway mileage. According to the Dutch government's Transport Task Force (Taakgroep Verkeer, 2006) 20% of van mileage is on motorways, i.e. about 4.1 billion kilometres.

CE Delft (1998) report an average motorway speed of 112 km/h for vans. Using speed-indexed emission factors, presented in the same study, simple calculation shows that in 2010 the motorway CO_2 emissions of vans will be 1.4 Megatonne.

If vans are equipped with a speed limiter, the average speed of these vehicles on the motorway will decrease to about 92 km/h (CE, 1998). At this average speed the CO_2 emissions will be 1.1 Mt, giving a CO_2 emission reduction of 0.3 Mt.

3.4 Costs

As discussed in Chapter 2, climate measures may be associated with different kinds of costs. The following basic categories can be distinguished: direct expenditures, welfare costs, additional external costs and indirect costs. In this section we examine the extent to which a mandatory speed limiter in vans gives rise to costs in these various categories. In doing so, we shall endeavour to quantify these costs wherever we can. Where this is unfeasible, the costs will be discussed qualitatively.

3.4.1 Direct expenditures

Introduction of a mandatory speed limiters in vans will lead to three kinds of direct expenditure: capital costs, operating costs and regulatory costs.

Capital costs
The capital costs engendered by this measure consist entirely of the costs of purchase and installment of the speed limiter. These costs depend very much on when the device is installed in the vehicle: during manufacture, or at a later date (retrofit). In the first case the costs are about € 200, while in the second they may be as high as € 650 (based on CE, 1998). As the measure is to be introduced in 2010, in the vast bulk of the van fleet in 2010 the device will have to be retrofitted. We have therefore calculated with purchase costs of € 650. Besides production and installation costs and the manufacturer's profit margin, these costs also include Value Added Tax. This VAT must be included when calculating the costs to the end user when private ownership is concerned, but not when calculating the costs to society. As VAT is tax-deductible for firms, in this case too end user costs must be calculated exclusive of VAT. Although a small portion of the Dutch van fleet is privately owned, for simpliicity's sake we here assume that all vans are trade vehicles. Assuming a VAT rate of 19%, the social and private costs of a speed limiter are then roughly equal: about € 550.

In allocating the capital costs we have adopted the linear depreciation method, thereby assuming that the service life of a speed limiter is the same as that of a van, i.e. 13 years on average. Assuming an average vehicle age of 5.5 years in 2010 (based on CE, 2003), this gives a depreciation period of 7.5 years. This therefore means the capital costs are € 73 in 2010. Assuming around 1 million vans in that year (based on CE, 2003), this means the capital costs for both end users and society are € 73 million.

Operating costs
Because installation of speed limiters leads to a reduction in the average speed of vans, the average fuel consumption of these vehicles will likewise decline. The data reported in TNO (2001) indicate that the projected reduction in van speed will lead to about 20% less fuel consumption. The benefits of these fuel savings will depend very much on the fuel price. In this analysis we have assumed a diesel price, excluding VAT, of € 0.90 per litre (€ 0.53 excluding both excise duty and VAT). In the sensitivity analyses we shall assess the influence of changes in the diesel price on the benefits of fuel savings. Assuming an average fuel

efficiency of 11.5 litres per 100 km in vans, we calculate fuel savings benefits to end users of about € 85 million and to society of about € 50 million.

The reduction in average van speed following installation of speed limiters also reduces the costs of tyre wear. Limiting vehicle speed to 100 km/h reduces these costs by 20% (CE, 1998). Tyre costs are 0.9 Eurocent per kilometre, which means savings on tyre wear of up to 0.2 Eurocent/km. In 2010 the benefits of reduced tyre wear are thus € 6.9 million.

Finally, a lower average vehicle speed will also lower the cost of engine and gearbox maintenance. For lack of data on this issue, however, these costs have not been included here.

Regulatory costs
Regulatory costs encompass, first of all, the costs necessarily incurred by government for formulation, implementation and enforcement of the policy measure. In this case study these costs will be fairly limited and so have been ignored. The extra costs necessarily incurred by van owners to satisfy requirements but having no direct influence on reducing CO_2 emissions are also regulatory costs. These include costs such as those associated with establishing the precise nature of the obligation, which speed limiters meet the specified requirements and so on. In our analysis these costs have likewise been assumed to be negligible.

3.4.2 Welfare costs

Besides direct expenditures, introduction of mandatory speed limiters in vans will also affect people's welfare in non-financial ways. The most pronounced of these is the increased travel time of van drivers resulting from the average lower speed of their vehicles. In 2010 around 7.9 million hours will be lost on the account.

The value assigned to one hour of travel time depends very much on the purpose of the trip. The value of an hour business-related travel time is thus greater than that of an hour 'leisure' travel time. To correctly value the lost hours resulting from lower average van speed we must therefore first establish the purposes of van trips. As set out in CE (2003), 29% of van-kilometres are for private reasons, implying that 71% are business-related.

Index figures for valuing the time spent on journeys for different motives are published annually by the Dutch Transport Research Centre (AVV, 2006). For road journeys of a business nature, an hour of travel time will be worth approx. € 31 per person in 2010, that of an hour for private motives about € 10. These per-person valuations are for an average vehicle occupancy of 1.11 and 1.5, respectively. Using these occupancy figures and the respective average values for travel time, a calculation can be made of the aggregate cost of the 7.9 million extra travel hours resulting from introduction of a mandatory speed limiter for vans. These costs will equal € 229 million in 2010.

Besides the extra costs accruing from longer travel times, certain other welfare costs can also be distinguished. The measure will deprive people of the opportunity to drive especially fast, even though some people derive utility from such behaviour (Slotegraaf *et al*., 1997). For lack of data, however, this effect has been ignored.

3.4.3 Additional external costs

Although in this case study a mandatory speed limiter for vans is intended primarily as a means of cutting CO_2 emissions, the measure will also have an impact on other transport externalities, in particular air quality, road safety and congestion.

Air quality
The decrease in the average motorway speed of vans will also mean a decline in air pollutant emissions. Here we consider only the reduction in NO_x emissions. Using the speed-indexed emission factors in CE (1998) we calculate, in the same way as for the CO_2 emission cuts, that the speed limiter leads to a 1.4 kt reduction in NO_x emissions. Taking the shadow price for NO_x emissions in the non-urban environment reported in (CE, 2001), we concludee that the benefits of reduced NO_x emissions amount to € 190,000.

Road safety
It is generally appreciated that road safety is enhanced by lower vehicle speeds. More precisely, on motorways there is found to be a 4th-power relationship between vehicle speed and deaths in traffic accidents and a 3rd-power relationship between speed and the number of injured parties (AVV, 2004). According to AVV (2004) the number of traffic deaths and injuries occurring annually on Dutch roads will decline by 2 and 100, respectively, if van speeds are limited to 100 km/h. CE (1998), in contrast, calculates that this reduction in vehicle speed will lead to 25 fewer deaths and 148 fewer injuries. Superficial inspection of the respective results reveals no clear reason for the major discrepancy between these two sets of figures. For this reason we here calculate with the AVV (2004) data, while in Section 3.6 we carry out a sensitivity analysis using the CE data (1998).

To assign a value to the number of traffic deaths requires monetary valuation of a ‘statistical human life’.[10] The epithet ‘statistical’ indicates that this is a valuation of risk reduction, not of tangible human lives. After all, if tangible rather than statistical lives are involved, as with miners trapped underground, for example, society’s willingness to pay soars to virtually infinity. When it comes to reducing risks, however, in practice people prove to be prepared to pay only a finite sum for further risk reduction, based on comparison with the benefits of alternative uses of the funds. Based on people’s willingness to pay to reduce the risk of

[10] We are concerned here solely with the external costs, i.e. traffic deaths other than the drivers of the vehicles involved in an accident, for the risk of oneself perishing in traffic as a result of one’s own actions is already accounted for.

death in a range of situations, researchers have calculated the financial Value of a Statistical Life (VOSL).

The figures cited for the VOSL differ widely from study to study. De Blaeij *et al*. (2004), in a review of the VOSL values officially employed in seven countries, found a range between € 1.4 and € 2.6 million. According to the European Conference of Ministers of Transport (ECMT, 1998), the best VOSL estimate from a scientific perspective is € 2.4 ± 1 million. For policy studies ECMT recommends taking a conservative estimate of € 1.5 million, a value in line with the suggestion made in the Dutch CBA Guidelines for infrastructure projects (OEEI, 2000).[11] The European research programme UNITE cites an estimate of € 1.7 million for the Netherlands (UNITE, 1998). Basing itself on another study by de Blaeij (2003), the Dutch Institute for Road Safety Research (SWOV) recommends taking a value of € 2.2±0.3 million (2001 price level) (Weseman *et al.,* 2005). Here we shall follow this latter recommendation. To make due allowance for the influence of the uncertainty in the VOSL value, in Section 3.6 we carry out a sensitivity analysis using a value of € 1.5 million.

For the value of a statistical injury we have taken a value of € 227,500 (CE, 2004). We here assume that the private costs of suffering injury in a road traffic accident amount to 50% of the social costs (CE, 1998).

Proceeding from these various figures, the total social benefits of a reduced risk of road accidents as a result of speed limiters in vans is then about € 27 million, while the private benefits are approximately € 14 million.

Congestion

Road capacity is optimally utilised at an average speed of around 90 km/h (CE, 1998). Limiting the speed of vans will therefore probably lead to improved traffic flow on motorways and consequently less congestion. For lack of data this effect could not be quantified, however.

3.4.4 Indirect costs

Introduction of a mandatory speed limiter for vans may also have all kinds of indirect effects. Demand for speed limiters will rise sharply, for example, with considerable benefits for manufacturers of such devices. To what extent this indirect effect will mean increased welfare for Dutch society as a whole, and not merely a redistribution of welfare, depends on whether it occurs inside or outside the Netherlands, and whether there are distortions in associated markets like labour and energy (for a further discussion, see Chapter 2). To assess any indirect effects of the measure in question would mean running economic models, however, an exercise beyond the scope of the present study.

[11] Note that the value recommended by ECMT and OEEI is for the price level of 1997.

3.5 Cost effectiveness

The costs and effects (i.e. CO_2 emission reductions) of a mandatory speed limiter for light commercial vehicles are summarised in Table 2, thereby distinguishing between the perspective of end users and society as a whole.

Table 2 CO_2 emission reductions and associated costs (€ mln) of a mandatory speed limiter for vans

	End user	Society
Effect		
CO_2-emission reductie (Mt)	0.30	0.30
Direct expenditures		
Capital costs	73.00	73.00
Fuel costs	- 85.00	- 50.00
Tyre costs	- 6.90	- 6.90
Welfare costs		
Costs of extra travel time	229.00	229.00
Additional external costs		
NO_x emissions	-	- 0.19
Road safety	- 15.00	- 27.00
Total costs	**195.10**	**217.90**

As Table 2 shows, the costs of a mandatory speed limiter for vans are to a very substantial degree determined by the costs of extra travel time. Once again, this makes clear just how important it is to consider other welfare effects alongside direct effects when assessing the efficiency of a given policy measure. Besides the costs of extra travel time, the benefits of fuel cost savings are also an important factor. This effect is largely responsible for the difference between the private and social costs of the measure, moreover. Because reduced payment of excise duty is a benefit for the end user but not for society as a whole, the savings on fuel costs are greater at the individual than the social level.

According to these figures, the social cost effectiveness of a mandatory speed limiter for vans is € 725 per tonne CO_2. This makes the measure more cost-effective than was calculated in the 'Option Document on Transport Emissions' (RIVM/CE, 2004), where a cost effectiveness of € 900 per tonne CO_2 was calculated for 2010. This difference can be explained partly by the fact that in our case study more effects have been included in calculating cost effectiveness, in particular the benefits of less tyre wear and enhanced road safety. In addition, a lower fuel price was taken in the Option Document, which meant less benefits due to fuel savings. Another factor is the different valuation of extra travel time. In Section 3.6 we carry out a sensitivity analysis to assess the influence of this last factor on the social cost effectiveness.

3.6 Sensitivity analyses

Establishing the nature and magnitude of the various effects resulting from introduction of a mandatory speed limiter in vans is an exercise wrought with all kinds of uncertainties. To acquire a better grasp of these uncertainties, in this section we perform a series of sensitivity analyses.

Depreciation of capital costs
The depreciation period taken for the capital costs is of major influence on the overall costs and cost effectiveness of the policy measure discussed. In our main analysis we assumed the speed limiters would be retrofitted in vans with an average age of 5.5 years. As the average age of a van is 13 years, the depreciation period was 7.5 years. Let us now proceed from the fictional situation of the devices all being installed ex works and subsequently being written off over 13 years. While this situation may be fictional for 2010, it is quite realistic for 2020, say, and thus of interest to consider.

Besides the fact that the capital costs can be written off over a longer period in this situation, the investment costs are also lower, for it is far less expensive to install a speed limiter during vehicle manufacture than to retrofit it. Together, these two effects cause the social and private costs to fall to € 14 million (compared with € 73 million for retrofit and 7.5 years depreciation).

The social cost effectiveness of a mandatory speed limiter in vans now becomes € 530/tonne.

Variations in fuel prices
Fuel prices are always subject to major fluctuation and it is no wonder that future fuel prices are shrouded in uncertainty. To include these uncertainties in the analysis, we here carry out two sensitivity analyses: with a 25% higher and a 25% lower pre-tax fuel price, thereby assuming that excise duty remains unchanged.

With a 25% higher fuel price, the benefits of fuel savings accruing to the end user rise by € 13 million to € 98 million, while the social benefits rise by € 13 to € 63 million. With a 25% lower fuel price, the social benefits decline by € 13 million to € 37 million, while the private benefits fall by € 13 million to € 72 million.

The fluctuations in fuel price also obviously have an impact on the social cost effectiveness of the measure. With a 25% rise in pre-tax price, the cost effectiveness becomes € 683/tonne, while with a 25% higher fuel price it is € 770/tonne.

Valuation of travel time
As we have seen, the costs of additional travel time form a major component of the overall costs of a mandatory speed limiter in vans. It therefore makes sense to examine how the valuation of travel time affects the overall costs. We do so by carrying out a sensitivity analysis in which we take the value of travel time used in RIVM/CE (2004). In that study, one hour additional travel time was assigned a

value of € 27, regardless of the journey's purpose. In our analysis this would add € 33 million to the costs of extra travel time, pushing the figure up to € 262 million. The social cost effectiveness then becomes € 836/tonne.

Greater reduction in road accidents

In the literature we found widely varying data on the influence of van speed limiters on road safety. In the main analysis we calculated with a reduction of two in the annual number of fatalities and 100 in the annual number of injuries. According to CE Delft (1998), however, limiting the speed of vans to 100 km/h leads to 25 fewer fatalities and 148 fewer injuries in the year 2010. With these input values, the social benefits of enhanced road safety increase by € 62 million to € 89 million, while the private benefits rise by € 57 million to € 72 million. This change in the social benefits of enhanced road safety also has a major influence on the social cost effectiveness of the measure, which is now € 520/tonne.

A lower value of a statistical life

There is a great deal of debate in the literature about the value to be accorded to a statistical human life (see Section 3.4.3). A VOSL value that is frequently adopted in policy studies is € 1.5 million, as recommended by the ECMT (1998). This value is considerably lower than that used in the present study: € 2.2 million. If a value of € 1.5 million is taken, the social and private benefits of enhanced road safety decrease by € 1 million to € 26 and € 14 million, respectively. The influence of this change in the value of a statistical life on the cost effectiveness of a mandatory speed limiter in vans is marginal: € 730 rather than € 725/tonne.

4 Case study 2: downsizing the vehicle fleet

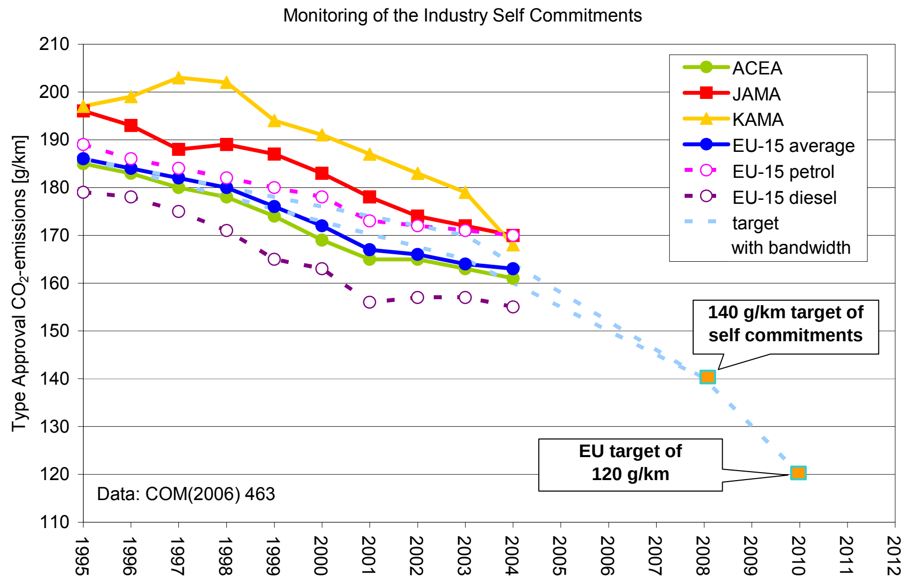
4.1 Introduction

In this case study, recent calculations of the cost effectiveness of scheduled European policy to reduce the CO_2 emission of new passenger cars to 130 g/km in 2012 are analysed and compared.

The aim of current EU policy on reducing passenger car CO_2 emissions is to reduce the average emissions (as measured in the so-called type approval test) of new cars to 120 g/km in 2010. The policy is based on three 'pillars'. The most important of these is an agreement between the EU and car manufacturers in which the three associations ACEA, JAMA and KAMA pledged to reduce the average CO_2 emission of newly sold cars from 186 g/km to 140 g/km between 1995 and 2008/9. The progress achieved to date under this agreement is shown in Figure 4. An important conclusion to have emerged from the monitoring of the agreement is that while significant emission cuts have been achieved by means of technical measures, the pace of reduction has been slowing down in recent years, although it should in fact have been increasing. As a result, it is becoming increasingly unlikely that the 'self-commitment' target of 140 g/km will in fact be achieved in 2008/9.

The other two pillars are consumer information (by means of fuel consumption labelling, as per Directive 1999/94/EC) and fiscal measures to induce consumers to buy more efficient vehicles, to be implemented by member states (COM(2005) 261).

Figure 4 Monitored trends in average CO_2 emissions of newly sold passenger cars under the agreement between the European Commission and ACEA, JAMA and KAMA



The European Commission recently reviewed CO_2 policy on cars and vans. In February 2007 a communication was issued (COM(2007) 19) setting out the main thrust of scheduled EU policy post-2008. The basic objective remains to ensure that the average CO_2 emission of newly sold vehicles is down to 120 g/km in 2012[12]. This is to be achieved in part by means of a CO_2 standard, under which the average CO_2 emission of these cars, as measured in the type approval test, may be no greater than 130 g/km in 2012. The other 10 g/km reduction is to come from a variety of elements from the so-called Integrated Approach:

- Minimum efficiency requirements for air-conditioning systems.
- Compulsory fitting of tyre pressure monitoring systems.
- Maximum tyre rolling resistance limits for cars and vans in the EU.
- Use of gear shift indicators.
- Measures to reduce the fuel consumption of vans.
- Increased use of biofuels, with requirements on environmental performance.

Exactly how the 130 g/km standard is to be implemented is still the subject of study[13]. The options being considered are various types of standard for individual vehicles and standards relating to the sales-average CO_2 emission per manufacturer.

Because CO_2 emissions are highly correlated with physical vehicle characteristics such as weight, engine rating, air resistance (degree of streamlining, front surface area) and the amount of on-board equipment, a

[12] More accurately, a level equal to the real-world emissions of a vehicle emitting 120 g/km in the type approval test.

[13] Being carried out by IEEP, CE Delft and TNO at the request of the European Commission (DG-Env.).

uniform CO_2 emission limit imposed at the vehicle level would lead to serious market distortions. Many cars would no longer be able to be marketed (e.g. sports cars, SUVs, MPVs). What would be feasible, by contrast, is to impose a uniform limit at the manufacturers' level relating to the average CO_2 emission of all vehicle sold. In that case it may make sense to combine the limit with a (closed) emissions trading scheme, under which manufacturers can trade emission credits among themselves (in g/km, based on the type approval test, for example) (IEEP 2005) so that in this way the desired aggregate emission reduction is apportioned across manufacturers and market segments in the most cost-effective manner.

The emission limit might also be differentiated according to a parameter representing a vehicle's utility, or use value, with bigger or 'flashier' vehicles being allowed to emit more, but manufacturers at the same time being challenged to produce as efficient vehicles as possible for a given utility. This emission limit could be applied to either individual vehicles or a manufacturer's sales average, again possibly combined with a trading scheme as described above. A third option would be to assign each manufacturer a reduction target formulated as a percentage of the sales-average CO_2 emission in a baseline year, possibly also combined with a closed trading scheme.

Estimates of reduction potentials and costs have been reported in (IEEP 2005), (TNO 2006), (TML 2006) and (ZEW 2006). These studies were carried out for the European Commission as groundwork for the new policy:

- Although TNO (2006) adopts the same methodology as IEEP (2005), because it uses additional data on costs and reduction potentials and an update of the analysis in IEEP (2005), it arrives at higher estimates of the cost of reducing passenger car CO_2 emissions beyond 140 g/km. TNO (2006) can thus be seen as superseding IEEP (2005).
- In TML (2006) and ZEW (2006) the results of TNO (2006) were used to calculate effects on the transport sector using TREMOVE and macro-economic impacts and effects on the automotive industry using PACE-T and FORCAR. Using TREMOVE, alternative calculations of CO_2 abatement costs were also made.

4.2 Results of available studies

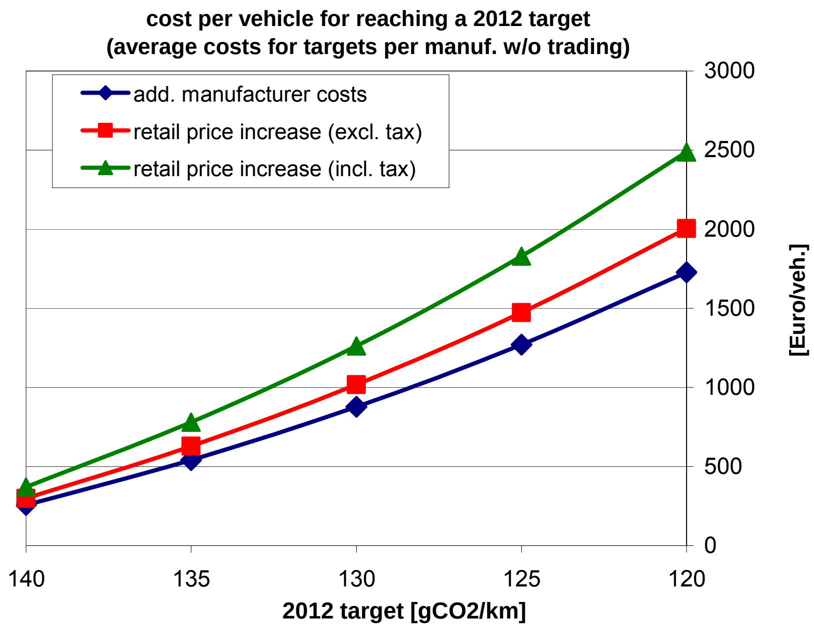
4.2.1 TNO 2006

As part of the groundwork for the EU's new policy on passenger car CO_2 emissions, TNO (2006) has presented estimates of the costs and benefits of various technical options that are already available for reducing the average CO_2 emissions (as measured in the type approval test) of newly sold cars from 140 g/km in 2008/9 to 120 g/km in 2012. The main thrust of this effort is to reduce these 'type approval emissions' by means of technical measures to vehicles (improved engine efficiency, improved powertrain efficiency, lower air and rolling resistance, lower weight).

In TNO (2006) it is calculated that to achieve 120 g/km in the type approval test will require the use of hybrid vehicles, in combination with action to improve engine efficiency and reduce vehicle energy requirements. The target of 130 g/km at the vehicle level already proposed by the European Commission is feasible with combinations of more conventional technical measures.

One key assumption in TNO (2006) is that the target pledged in the agreement between the European Commission and the automotive industry (140 g/km average in 2008/9) will indeed be achieved. Thus, the results are presented in the form of additional costs per vehicle and cost effectiveness (Euro per avoided tonne CO_2) for further reduction beyond 140 g/km to a level between 135 and 120 g/km in 2012.

Figure 5 Additonal costs per vehicle of reducing the arverage CO_2 emission (based on type approval testing) of new cars in the EU-15 from 140 g/km in 2008/9 to a target between 140 and 120 g/km in 2012.



Source: TNO, 2006.
N.B. The only costs shown in this figure are the additional costs of production (manufacturer costs) and new vehicle purchase (retail price increase).

Under the assumptions that purchase behaviour is uninfluenced, that autonomous trends with respect to the share of diesels[14] and increasing vehicle weight[15] remain unchanged, and that the full costs of meeting the mandatory CO_2 requirements are passed on to consumers, (TNO 2006) calculates that new cars coming onto the European market between 2008/9 and 2012 will be around € 2,500 more expensive (retail price) as a result of the technical measures required to achieve the desired CO_2 emission reduction from, on average, 140 to

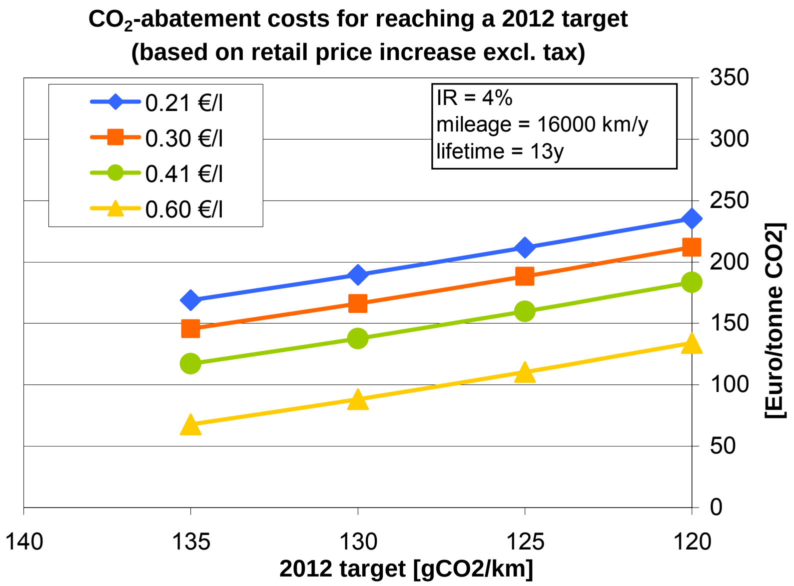
[14] The share of diesel vehicles in EU sales is assumed to rise from 50% in 2008 to 55% in 2012.

[15] The average autonomous weight increase (due to trends towards bigger, more luxurious and therefore heavier vehicles and to active and passive safety measures) is taken to be 1.5% per annum.

120 g/km. The consumer recuperates only part of this price rise by way of the associated fuel savings. The technical measures required to achieve 140 g/km in 2008/9 are cheaper and are, by contrast, recuperated by the individual consumer. The impact of standing EU policy up to 2010 may therefore be partly compensated by the increase in automobility resulting from lower costs. It may be added that enforcement of 140 g/km post-2008 also costs money, around € 400 per vehicle, because the effects of rising vehicle weight on CO_2 emissions must be compensated by efficiency-improving measures.

At an oil price of 50 €/bbl the abatement costs of achieving 120 g/km are around 180 €/tonne. At this price the step from 140 to 130 g/km costs about 140 €/tonne.

Figure 6 CO_2 abatement costs (in €/tonne avoided CO_2-equivalents) as a function of target and oil price for achieving a 2012 target fo 140 - 120 g/km for new vehicles (based on type approval test) by technical measures to vehicles



Source: TNO, 2006a.
N.B. In calculating the abatement costs, fuel costs and CO_2 reductions were calculated using estimated real-world fuel consumption and, for CO_2 reduction, allowing for Well-to-Wheel CO_2 emissions down the fuel chain. The fuel costs shown correspond with oil prices of 25, 36, 50 and 74 €/bbl.

In estimating the cost of newly applied technologies, (TNO 2006) assumes sufficiently large production series (>100,000 a year), although without explicitly ‘tweaking’ this parameter to assess its effect. Additional costing uncertainties arise from the fact that there are no concrete data readily at hand for estimating learning curves, economies of scale or the impact of innovations with respect to both products and production methods. Nonetheless, it seems reasonable to assume that in the longer term (by 2020-2030, say) the costs of these technologies may be significantly lower than the current estimates of TNO (2006) for the period 2008/2012.

4.2.2 ZEW (2006), TML (2006) and EC(2007b) 60

The European Commission's Impact Assessment for the proposed new CO_2 policy for passenger cars is presented in EC(2007b) 60. It is based on the results of (ZEW 2006) and (TML 2006), which were calculated in part using inputs from TNO (2006). Although adjusted cost data were also used for these calculations, these have been left out of consideration here because our present focus is on a comparison of the results yielded by different methods based as far as possible on the same input data.

Table 3 Comparison of CO_2 abatement costs for different 2012 standards for the average CO_2 emission of newly sold cars, as calculated in (TNO 2006), (ZEW 2006), (TML 2006) and EC(2007b) 60

	135 g/km	130 g/km	125 g/km	120 g/km
TNO (2006)[a]	143	164	186	210
ZEW (2006), EC (2007a) 69[b]	50	85	109	132
TML (2006)[c]	95	150	189	228

[a]) Oil price 36 €/bbl; abatement costs at vehicle level calculated using formula (1) in § 4.3.5.

[b]) Oil price 31 €/bbl; abatement costs for private transport sector in period 2010 - 2020 calculated using formula (2) in § 4.3.5.

[c]) Oil price 31 €/bbl, abatement costs for private transport sector in 2020 calculated using formula (2) in § 4.3.5.

The CO_2 abatement costs for targets ranging from 135 and 120 g/km according to the various studies are reported in Table 3. In (ZEW 2006) and EC(2007b) 60 the abatement costs were calculated using TREMOVE by dividing the total additional social costs over the period 2010-2010 by the cumulative CO_2 reduction over the same period. This method leads to lower abatement costs than the figure adopted in TNO (2006), for the following reasons:

- Between 2008 and 2012 vehicles will be sold with an average CO_2 emission between 140 g/km and the standard for 2012. On average, these vehicles will have lower CO_2 abatement costs.
- In TREMOVE, annual mileage is a function of vehicle age. As new vehicles are driven more kilometres than old ones, most of the vehicles sold in the period 2010-2020 will have a greater average annual mileage during that period than that calculated over their entire service life. Because additional vehicle costs are annualised linearly, the CO_2 abatement costs are thus lower than if they were calculated over the entire service life.

In TML (2006) CO_2 abatement costs are calculated on the basis of additional social costs and the total CO_2 reduction in 2020. In that year a large portion of the vehicle fleet will consist of vehicles meeting the standard for 2012. These results are thus more in line with the approach adopted in (TNO 2006).

The formula used for calculating CO_2 abatement costs in TNO (2006) differs from that in EC(2007b) 60, ZEW (2006) and TML (2006). On this point, see § 4.3.5.

In addition, EC(2007b) 60, ZEW (2006) and TML (2006) base their calculations on additional vehicle costs a factor 1.16 lower than the results of TNO (2006), because EC(2007b) 60 assumes that the manufacturer passes on the costs of CO_2 abatement technology to customers with no mark-up at all.

4.3 Factors affecting calculated CO_2 abatement costs

4.3.1 Baseline

The calculations in TNO (2006), ZEW (006) and TML (2006) all proceed from a baseline that assumes the 140 g/km target laid down in the agreement with manufactrers will indeed be achieved in 2008. If this assumption proves unjustified, the costs of reaching a level of 120 or 130 g/km will therefore be higher. It is to be queried, though, whether these extra costs should be allocated to the new policy for 2012 or taken as 'repair costs' for standing EU policy.

One key assumption in the TNO (2006) calculations is the 1.5% per annum autonomous increase in vehicle weight. It is debatable whether this trend will continue through to 2012 or flatten out, for example because the scheduled tightening of 'crashworthiness' requirements will lead to minder weight-increasing measures than has been the case under the requirements in force over the past 10 years. An alternative scenario in TNO (2006) in which the annual rise in weight falls from 1.5% in 2004 to 0.5% in 2012 yieds a 19% reduction in the additional vehicle costs involved in going from 140 to 120 g/km. This cost reduction leads to 30% lower CO_2 abatement costs.

4.3.2 Technical versus policy measures

In TNO (2006), ZEW (2006) and TML (2006) the cost effectiveness of a range of technical and physical CO_2 abatement measures is estimated, with costs and reductions being calculated based on an assumed degree of penetration of the respective measures. In the factsheets accompanying the 'Option Document on Energy and Emissions, 2010-2020', on the other hand, to take an example, the analysis focuses mainly on policy measures for achieving more efficient vehicles and lower CO_2 emissions. The response to these measures generally involves a mix of physical reduction options and other behavioural measures. In the calculation of CO_2 reductions there is, for example, an estimate of the effects of the various policies on mobility demand (based on price elasticities, for example). This often makes it difficult to compare the results of cost effectiveness calculations concerning technical measures and policy measures.

4.3.3 System boundaries

In TNO (2006) cost effectiveness is calculated at the vehicle level, under the assumption that annual mileage is unaffected by changing fuel and vehicle costs. In ZEW (2006) and TML (2006), on the other hand, cost effectiveness is calculated for the private transport sector as a whole. This gives rise to much the same problems as touched upon in the previous subsection.

4.3.4 Time horizon

Both technical and policy measures generally take some time to achieve their full intended impact. If all new passenger cars coming onto the market from 2012 onwards satisfy the 130 g/km standard, for example, it will still take more than 10 years before over 90% of the fleet is up to this mark, i.e. before the full reduction potential is approximately achieved. One can then either opt to calculate the cost effectiveness at a time horizon when the measure has sufficiently penetrated, or calculate the abatement costs based on the cumulative costs and benefits over a longer period. The first approach is adopted in TML (2006), the second in ZEW (2006).

4.3.5 Calculation formulae for CO_2 abatement costs

Quantitative comparison of greenhouse gas abatement measures is based on the emission reduction potential of the measures in question and their aggregate costs, in Euro per tonne of avoided greenhouse gas emissions (expressed as CO_2-equivalents). The reduction potential is expressed in ktonne or Mtonne per annum and depends, first, on the geographic area in which the measure is to be implemented (e.g. the Netherlands, EU-15 or EU-25) and, second, on the penetration of the technology in the vehicle fleet.

Abatement costs can be calculated for an individual technology or for a complete array of measures, as part of scenario calculations. For the first option, TNO(2006) employs the following formula:

$$CO_2 \text{ abatement costs} = \frac{I - \text{NPV (lifetime fuel cost savings)}}{\text{lifetime } CO_2 \text{ emission reduction}} \qquad (1)$$

In this formula the total lifetime fuel cost savings associated with the technology are converted to a net present value (NPV), which is then deducted from the investment involved in acquiring the technology. In this calculationo of CO_2 abatement costs, 'social' costs are used and so investment costs and fuel costs are taken exclusive of taxes (as discussed elsewhere).

In other studies the following alternative and more generic formula is often used:

$$CO_2 \text{ abatement costs} = \frac{I^{an} + \Delta_{O\&M} - \Delta_{\text{fuel costs}} - \text{secondary benefits}}{\text{annual } CO_2 \text{ emission reduction}} \qquad (2)$$

where $\Delta_{O\&M}$ represents the additional annual operating and maintenance costs and $\Delta_{\text{fuel costs}}$ the annual savings on fuel costs (due to the vehicle becoming more fuel-efficient). The formula also includes monetised secondary benefits (e.g. cuts in air pollutant emissions through use of more efficient technology). In the case of transport technologies, additional O&M costs and secondary benefits are generally negligible, it may be added. I^{an} in the formula is the annuity of the total investment costs I:

$$I^{an} = I * \frac{(1+r)^l * r}{(1+r)^l - 1} \qquad (3)$$

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

The results of formulae (1) and (2) differ by up to several dozen percent, depending on the lifetime and interest rate assumed, as illustrated in Table 4.

Table 4 Comparison of CO_2 abatement costs according to formulae (1) and (2) for the example of a reduction in CO_2 emissions (type approval test) from 140 g/km in 2008 to 130 g/km in 2012 (based on results in TNO (2006))

Baseline type approval emission	[g/km]	140
Target type approval emission	[g/km]	130
TTW[a] CO_2 reduction	[%]	7.1
Social costs	[€]	711
Baseline TTW emission	[g/km]	167
WTW[1] CO_2 reduction	[g/km]	14
Fuel price (pre-tax)	[€/l]	0.30
Fuel saving	[l/km]	0.0048
Annual mileage	[km/a]	16,000
Fuel saving	[€/j]	-23
Lifetime	[a]	13
Discount rate	[%]	4
NPV of fuel saving	[€]	228
Abatement costs, formula 1	[€/tonne]	164
Annuity of investment	[€]	71
Abatement costs, formula 2	[€/tonne]	214
Difference	[%]	30

[a] TTW: Tank-to-Wheel; WTW: Well-to-Wheel

Instead of calculating at the vehicle level, as set out above, models describing the overall transport systeem or vehicle fleet can be used to perform scenario calculations on individual options or policy packages. Using the changes in total costs and total CO_2 emissions, the models can then be used to calculate abatement costs. In these models allowance can also be made for the impact of changes in fuel and vehicle costs on vehicle ownership and use and on utilisation of other modes of transport. In cases where costly CO_2 abatement measures lead to reduced transport mobility, the avoidance costs derived in this way will be lower than those calculated at the vehicle level under an assumption of unchanged behaviour vis-à-vis vehicle purchase and use. This is the approach taken in the calculations with the TREMOVE model reported in ZEW (2006) and TML (2006).

4.3.6 From type approval to real world, Well-to-Wheel and CO_2-equivalents

Calculations of fuel savings and CO_2 emission reductions should be based on real-world fuel consumption and not on the values measured in the type approval test. In TNO (2006) a factor of 1.195 is taken for converting fuel use and CO_2 emissions during type approval testing to practical values. This factor covers the effects of driving on various categories of road and associated driving patterns, driving style and air-conditioning use. With time, however, this factor will change. As many CO_2 abatement measures focus on the part-load efficiency of the powertrain, the discrepancy between the type-approval and real-world value will alter (in principle decreasing).

Other greenhouse gases besides CO_2 are also potentially important. This holds for vehicle emissions (Tank-to-Wheel, or TTW emissions) and also, particularly, for emissions occurring upstream in the fuel supply chain (Well-to-Tank, or WTT emissions). In the case of the former, we are talking specifially about CH_4 and N_2O. Despite the high global warming potential (GWP) of 23 for CH_4 and 296 for N_2O, the share of these gases in the total climate impact of vehicle exhausts is very limited, however. This is not the case for the WTT emissions associated with biofuels production and the extraction and distribution of natural gas, though. In studies such as (Concawe 2006) greenhouse gas emissions are calculated over the entire supply chain, i.e. as Well-to-Wheel (WTW) emissions, and expressed in CO_2-equivalents using standard GWP values. Emissions of coolants (HFCs) from mobile air conditioners also contribute to total transport greenhouse gas emissions.

When calculating CO_2 emission reductions, due consideration should also be given to the fact that energy savings at the vehicle level also lead to reduced energy use and CO_2 emissions in the entire fuel supply chain. Table 5 shows indicative factors for converting direct, Tank-to-Wheel CO_2 vehicle emissions to aggregate, Well-to-Wheel emissions.

Table 5 Well-to-Wheel greenhouse gas emissions of supply chains for petrol and diesel (taken from TNO (2006), data based on Concawe, (2006))

	TTW		WTT			WTW
	CO_2-content (gCO_2/MJ_fuel)	Lower heating value (LHV) (MJ/l_fuel)	WTT energy consumption (MJ/MJ_fuel)	WTT CO_2-emission (gCO_2/MJ_fuel)	WTT CO_2-emission (gCO_2/gCO_2_TTW)	WTW CO_2-emission (gCO_2/gCO_2_TTW)
Petrol	73.40	32.20	0.14	12.50	0.170	1.170
Diesel	72.80	35.80	0.16	14.20	0.195	1.195

The question, though, is to what extent this should be included in calculating the cost effectiveness of national policies, for example. If the fuel is not produced domestically, the upstream emissions will occur abroad and cuts in these WTT emissions will not therefore count towards securing national CO_2 reduction targets (including the Kyoto target). In (TNO 2006) a conscious choice has been made to calculate using WTT emissions, because technical measures to vehicles are compared with CO_2 reduction through use of natural gas or biofuels, for example. It is unrealistic to assign zero CO_2 emissions to biofuels produced outside the EU.

4.3.7 Sensitivity to variations in input data

Although the concept of abatement costs appears to be an attractive variable for comparing the effectiveness of CO_2 abatement options, it should be used with due caution. In calculating these costs (see formulae (1) and (2)) one is confronted in the numerator with a difference between two numbers (investment and avoided fuel costs) that are often of roughly the same magnitude. The difference between two such numbers is very sensitive to minor variations in the magnitude of those numbers. If investments and avoided fuel costs differ by 10%, a 10% variation in one or other of these numbers will lead to a variation of 100% in the calculated abatement costs! In this case the cost calculation formula acts as an 'multiplier' for variations in the input data. In TNO (2006) it is demonstrated that this issue is highly relevant in calculating the abatement costs of measures to improve the fuel efficiency of passenger cars, particularly in scenarios with a low oil price. Estimating the costs of future technologies to an accuracy of more than about ±10% would appear unfeasible, it may be added, certainly in the case of vehicle technologies where innovation, learning curves and economies of scale play a major role.

4.3.8 Comparison of data from different sources

The various studies in which CO_2 abatement costs are calculated deal with the aforementioned and other methodological issues in different ways. In some cases calculations are made under different assumptions with respect to oil price, currency exchange rates and discount rate, for example. One important conclusion drawn in CE (2005a, 2006) and IVM (2006) is that these differences make it very difficult to compare the results of different studies, whether these studies concern CO_2 abatement in the transport sector alone or comparisons of CO_2 abatement costs across different sectors.

4.4 Interactions between CO_2 and other emissions

Fuel-efficient cars are not necessarily cleaner with respect to emissions of tailpipe air pollutants (CO, HC, NO_x, PM) than their inefficient counterparts. Likewise, small cars are not always cleaner than large and heavy vehicles. All vehicles using a given fuel must in principle meet the same emission standard (expressed in g/km). In practice, however, there are differences in the emissions of individual vehicles (using the same kind of fuel), due to:

- How far the vehicle remained within the standard in the type approval test
- The extent to which engine management and exhaust gas after-treatment are able to keep emissions low under practical conditions, too.

In practice the wider design margins adopted with large, heavy vehicles may well mean that the quality of the exhaust gas after-treatment system on large vehicles is so superior to that on small vehicles that large, fuel-inefficient vehicles are on balance cleaner.

Neither is a hybrid powertrain in itself any cleaner than a conventional one. Because of the higher engine load per kWh output, a diesel engine in a hybrid train will in fact produce more NO_x. Because the engine load is less dynamic, though, a hybrid train creates greater scope for further emission reduction. In the case of the Prius, this has been used to satisfy the Californian SULEV emission standards. This was a conscious choice on the part of Toyota, however, and not a 'free' side-effect of the hybrid powertrain used. Hybrid vehicles powered partly from the electricity mains ('plug-in hybrids'), giving them a greater action radius in the purely electric mode (i.e. with the combustion engine switched off) may give greater emission gains at the local level, which may be of relevance in the context of environmental zoning, for example.

Measures to improve vehicle fuel efficiency often seek to avoid partial loading of the engine. In principle, altering the engine load characteristics of an existing vehicle will have a major influence on air pollutant emissions. Some emissions will increase, while others will decline. In the case of new vehicles, however, implementation of CO_2 abatement measures is part and parcel of the overall design process, which already involves improvements to engine management and exhaust gas treatment systems with a view to satisfying the latest generation of emission standatrds. Measures relating to variable valve timing, for example, are employed to both ends. Any undesired side-effects of fuel-efficiency measures can be addressed by means of additional exhaust treatment (a particulate filter, say). In the final phase of development, moreover, once the hardware configuration has been finalised, calibration of the engine and control systems still provides significant scope for optimisation. On the one hand it is difficult to predict how possible effects of CO_2 abatement measures will affect the air pollutant performance of the overall design; this is even truer of real world emissions than of type approval emissions. On the other hand these effects are likely to be largely absorbed, as vehicles must satisfy the current emission standards anyway.

In the official Dutch emission factors (TNO's VERSIT+ model) for air pollutant emissions, no distinction is made with respect to categories of vehicle size. Any shift to smaller vehicles as a result of rising vehicle costs will therefore have no effect on registrered air pollutant emissions.

Tailpipe emissions standards differ for petrol and diesel vehicles. Euro 4 diesels produce roughly 5 times more NO_x and PM than Euro 4 petrol cars, but 15 to 20% less CO_2. Any trend towards a greater share of diesels, in the context of CO_2 abatement policy or as a market response to changing vehicle costs, will thus indeed have a secondary impact on air pollutant emissions. Under the

Euro 5 and (possible further) Euro 6 standards, the requirements for petrol and diesel vehicles will converge, however, so that with time this effect will become increasingly less important.

Tyres with low rolling resistance are a valuable CO_2 abatement measure. Efforts are also being made to create quieter tyres. It is unknown whether technical measures to reduce rolling resistance will have a positive or negative impact on tyre noise.

Considering all of the above, when evaluating CO_2 abatement measures for passenger cars it is defensible to assume no secondary effects on air pollutant emissions. This is also the position taken in (TNO 2006).

4.5 Welfare-economic analysis

In the following sections we carry out a welfare-economic analysis of the cost effectiveness of 'downsizing' the vehicle fleet, i.e. reducing average vehicle size. The numbers used in this analysis are intended primarily for illustrative purposes. Within the scope of this study it was not feasible to chart the complex consequences of mandatory CO_2 emission standards in any detail. This is because behavioural responses may take any number of forms and the magnitude of such responses (price elasticities) have been only superficially investigated to date.

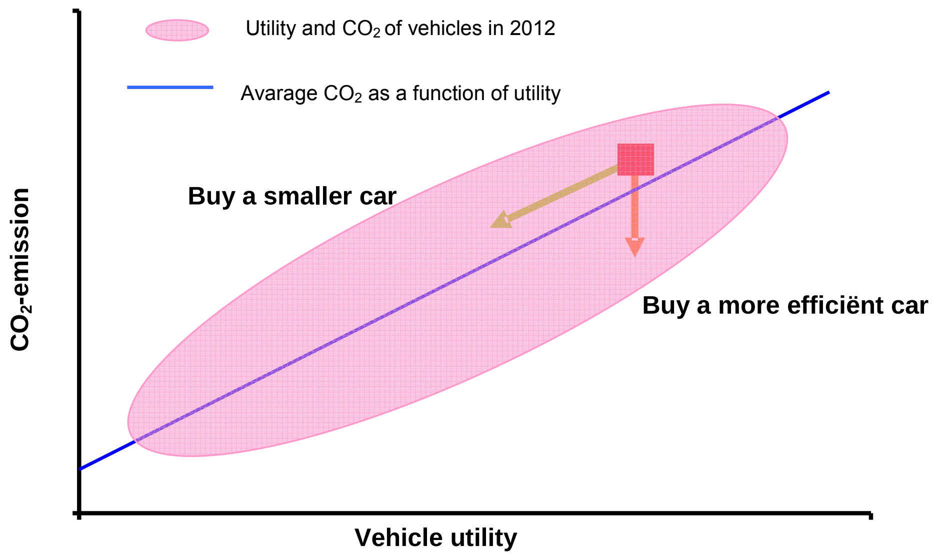
Introduction of mandatory vehicle emission standards generally leads to an increase in the costs of car ownership and use.[16] Consumers will endeavour to limit these additional costs and have various options for doing so. They may buy a more efficient model of car in the same category (size or market segment). Whether this will limit the additional costs, or even lead to cost savings, depends on the additional price of the technology employed to improve fuel efficiency (including any subsidies or tax benefit) and the extent of fuel cost savings (and thus on the improvement in fuel efficiency and the fuel price). Alternatively, consumers may buy a smaller car. Based on a purely financial considerations, this is always a cost-effective means of reducing CO_2 emissions, as the direct abatement costs are negative. In this case, though, welfare costs arise, because from the consumer's perspective a smaller car does not represent the same added value as a larger model. Finally, faced with the higher costs, some consumers will opt to no longer purchase or drive a car.

The most obvious average strategy is one embracing all three options, with some people buying a more efficient model in the same category, some buying a smaller model, and some declining to purchase altogether. Assuming this average response, and making due allowance for welfare costs, the economic optimum will be given by the situation whereby the marginal additional costs of an even more efficient vehicle begin to exceed the marginal welfare costs of an

[16] Assuming the same composition for the existing fleet and newly sold vehicles.

even smaller one (Figure 7). Both the welfare costs and the additional costs of technical measures will increase supralinearly with respect to the origin.

Figure 7 Vehicle utility and CO_2 emission of car models



As a quantitative illustration of the welfare effects, we here provide a simplified calculation based on an average vehicle.

4.5.1 Direct expenditures

The direct expenditures and CO_2 savings are based on Table 4 in Section 4.3.5.

Table 6 Direct expenditures and CO_2 savings

	End user	Society
Investment costs (€)	-846 [17]	-711
NPV of fuel savings (€)	912 [18]	228
Vehicle lifetime CO2 savings(tonne)	2.9	2.9

4.5.2 Welfare effects of behavioural change

The loss of welfare resulting from introduction of emission standards for passenger cars we shall approach as follows. The key question is how the split will be between consumers who continue to buy a car in the same category and put up with the higher cost price, those who buy a cheaper model, and those who opt out of the market altogether. Instead of reviewing the entire market, however, we here project the effects onto purchase of a single vehicle, with the buyer

17 Assuming a 19% average European tax on top of the basic car price.

18 At a fuel price including taxes of 1.20 €/litre.

being able to choose between a greater or smaller number of 'auto-utils' (representing vehicle utility value), expressed in Euros prior to the cost price increase. At the current price of one auto-util per Euro, the average car user buys a vehicle costing around € 20,000. As a result of the cost price increase of € 846, the price of an auto-util rises from one Euro to € 1.0423 (1 + 846/20,000). It now depends on the price elasticity how many fewer auto-utils the buyer purchases. At a price elasticity of minus one, this will be 20,000/1.0423 = 19,188 auto-utils. In practice this is the average for the entire vehice fleet, of course. Some people will continue to purchase the same number of auto-utils, some people fewer and some none at all. It is important to note that the price elasticity that needs to be used is thus not the price elasticity of total car sales as a function of retail price. This latter elasticity is lower, because it does not include the effects of market shifts from pricier to cheaper models, with the accompanying loss of 'auto-utils'.

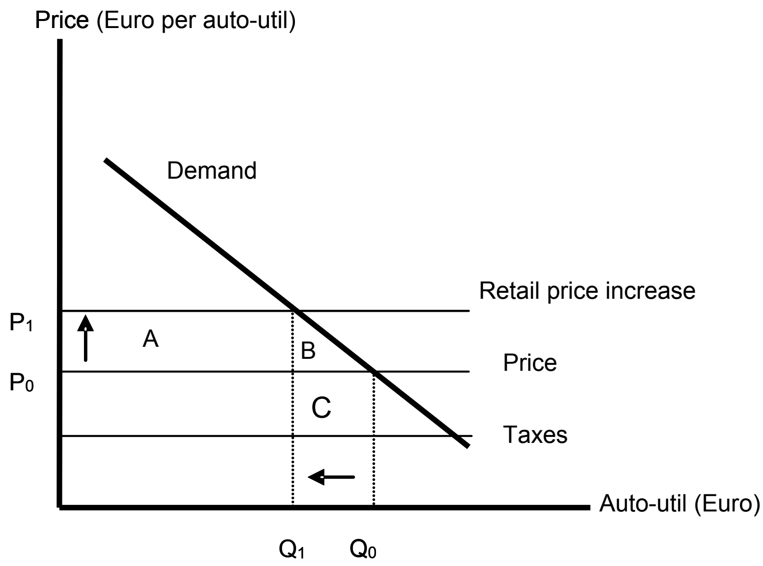
A wide range of price elasticities is to be found in the literature. In a study on the effects of abolishing the tax on car ownership in Virginia, Craft & Schmidt (2005) report a price elasticity of -1.2 for 'vehicle capital'. Studies on the price elasticity of total car sales as a function of retail price yield lower values, as expected. The meta-analyses by Graham & Glaister (2002: 31) and Goodwin *et al.* (2004: 286), for example, give long-term price elasticities of -0.90 and -0.49, respectively. In the other literature consulted we encountered similar values, viz.: Train, 1979; Lave & Train, 1979; Levinsohn, 1988; McCarthy, 1996; Bjørner, 1999. Based on the literature we see no reason for taking a price elasticity other than minus one for 'vehicle utility value'.

The welfare effects are as follows. The rise in the retail price of the number of auto-utils that the vehicle user purchases he puts up with. In Figure 8, this is represented by the rectangle A. By approximation, this rectangle has an area given by: $846 + \varepsilon\ 846^2/20{,}000$, where ε is the price elasticity. With a price elasticity of -1, the area is then approximately € 810. Given the uncertainties with respect to the investment costs themselves, the rectangle can also simply be considered equal to the investment costs of € 846.

- Part of this loss of welfare on the part of the end user consists of additional tax (Vehicle Purchase Tax and VAT, for example) over the now higher retail price as a result of the new emission standard. Taking the European average, with about 16% of the retail price consisting of tax, the additional government revenue over a single vehicle is then approximately given by $0.16 \times (846 + \varepsilon\ 846^2/20{,}000)$ or, by further approximation: 0.16 x 846 = € 135. From society's perspective, however, this tax revenue embodies a redistribution of welfare rather than a loss.
- To an extent, the retail price increase will also mean that the average consumer buys fewer auto-utils. This leads to a loss of consumer surplus, as described in Section 2.4.1, equal to the percentage price rise times the decrease in auto-utils times 0.5. In Figure 8 this is triangle B. By approximation, it has an area of: $-0.5 \times \varepsilon\ 846^2/20{,}000$. At a price elasticity of -1, this area represents about € 18. This term is thus relatively small.
- A more important welfare effect resulting from behavioural change arises, however, through the loss of tax income (VPT, VAT). In Figure 8 this is given

by triangle C. By approximation, the magnitude of this loss of social welfare is: - 0.16 x ε 846. With a price elasticity of -1, this gives a figure of around € 135. The first point to note is that we here have a true loss of welfare that is not compensated by the extra tax income associated with the higher costs of emission abatement technologies. From society's perspective, as mentioned, the tax revenues represent no more than redistribution. The second point to note is that profit and loss with respect to tax revenue are only equal if the price elasticity is minus 1.

Figure 8 Welfare effects of a retail price rise



The following simplified example may serve to illustrate the above welfare effect due to behavioural change. Assume that without an emission standard eleven people buy a car costing € 10,000. Tax comprises 50% of the purchase price, so total tax revenues are € 55,000. Following introduction of the emission standard, the cars become € 1,000 more expensive, including tax, and as a result one person decides to no longer buy a car. Total expenditures and tax revenues thus remain the same: € 110,000 and € 55,000, respectively. However, as there is one person no longer buying a car that was worth him or her € 10,000, but for which the cost to society (i.e. excluding tax) was € 5,000, there is a loss of social welfare due to behavioural change amounting to € 5,000.

Alternatively, all eleven people still buy a car, but now a model that would have cost € 9,091 without the emission standard, and € 10,000 with the standard in place. The total expenditures and tax revenues then again remain the same: € 110,000 and 55,000, respectively. All eleven people now decline to buy a € 909 'more luxurious' car, a purchase they would have made without the emission standard. The costs to society (i.e. excluding tax) of that extra 'luxury' was half this figure, however: € 455. There is therefore a loss of social welfare due to behavioural change of € 5,000.

4.5.3 Additional externalities

As argued in Section 4.4, we see no reason to include other externalities resulting from introduction of CO_2 emission standards for passenger cars.

4.5.4 Total costs

Table 7 summarises the various welfare effects per average car buyer from the perspective of the end user and society as a whole. The main differences between the two are, first of all, the tax component. The end user pays extra tax on the emission abatement technology but less tax because of fuel savings. Secondly, a welfare-economic analysis shows up the loss of welfare resulting from behavioural change, that is, the loss of welfare for those deciding to buy a smaller car, or not buy one at all. From the perspective of the end user this effect is limited, but because of the considerable tax paid on a new car, from society's perspective the impact is substantial. Its magnitude is directly proportional to the price elasticity, however: with a price elasticity of zero there will be no behavioural effects and consequently no loss of welfare due to behavioural change.

Table 7 Average welfare effects (in Euro) per car buyer of tightening CO_2 emission standard from 140 g/km in 2008 to 130 g/km in 2012

	End user	Society
Retail price (pre-tax)	-711	-711
Retail price (tax)	-135	0
NVP of fuel savings (pre-tax)	228	228
NVP of fuel savings (tax)	684	0
Loss of welfare due to behavioural change (pre-tax)	-18	-18
Loss of welfare due to behavioural change (tax)	0	- 135
Total	48	-636
Lifetime CO_2 savings	2.9 tonne	2.9 tonne
CO_2 cost effectiveness	17 €/tonne	-219 €/tonne

5 Interviews

5.1 Interviewees

In this study, besides conducting a literature survey we also held interviews with the following experts in the field of cost effectiveness analysis:

Peter Zwaneveld (Netherlands Bureau for Economic Policy Analysis, CPB);
Annemiek Verrips (CPB);
Paul Besseling (CPB);
Pieter Kroon (Netherlands Energy Research Centre, ECN);
Piet Boonekamp (ECN);
Ruud of den Wijngaart (Netherlands Environmental Assessment Agency, NMP);
Bert van Wee (Delft Technological University);
Kees Vijverberg (Netherlands Environment Ministry, VROM).

5.2 Results

The purpose of the interviews was first of all to acquire a deeper understanding of a number of substantive issues, such as the relevance of the debate around positional goods. The second aim was to gain an impression of whether the ongoing debate in the international literature is also taking place in the Dutch context. This does indeed appear to be the case. The majority of the researchers are of the opinion that to arrive at a realistic estimate of the cost effectiveness of climate measures requires a more comprehensive form of welfare-economic analysis than recommended in the environment ministry's 'Environmental Costing Methodology Manual'. In particular, the effects of measures designned to influence motorists' behaviour may be erroneously estimated if only changes in direct expenditures are included. The only issue that is more or less uncontroversial, however, is the inclusion and monetary valuation of time losses and gains. With respect to other externalities, although these are minority viewpoints, there are still doubts about whether these should be included, as in the case of loss of comfort arising as a result of environmental policies, for example. Most researchers regard the monetisation of the various externalities as sufficiently robust inputs for cost effectiveness analysis, although here again there is a minority viewpoint that holds that the monetary values are still too controversial for inclusion.

6 Conclusions and recommendations

6.1 Conclusions

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. To this end, experts in the Netherlands were consulted and the national and international literature reviewed. 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 vehice 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 the Dutch environment ministry's 'Environmental Costing Methodology Manual', drawn up in 1994 and updated in 1998, it is recommended that the cost effectiveness of environmental measures be calculated on the basis of direct expenditures only. This is the approach adopted in many national and international studies. 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. In the Dutch context, the Option Document on Transport Emissions forms an exception here. Studies comparing different transport measures are thus generally based solely on analysis of direct expenditures. 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 lead to this kind of study yielding very different results:
   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. It should be noted, though, that these are 'minority viewpoints':

- 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 vehice 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.

6.2 Recommendations

As set out above, we do not regard as problematical in itself the fact that differences in perspective, i.e. the end user, or society as a whole, leads to differences in the results of cost effectiveness analyses. Researchers and policymakers generally make it very clear that the two perspectives serve different purposes and consequently yield resuts that cannot be compared.

Nor do we have any concrete recommendations concerning the differences in the results of such analyses that arise through the different choices made in calculating direct expenditures. It holds for any individual cost effectiveness analysis that it is up to readers and users to determine whether the basic choices and assumptions made are sufficiently and convincingly underpinned. In the Dutch context, at any rate, we see no fundamental differences in the approaches adopted by researchers.

With regard to the distinction between cost effectiveness analysis based on an analysis of direct expenditures and a more comprehensive welfare-economic analysis, we have several recommendations, specifically for the Dutch context.

The ministry's 'Environmental Costing Methodology Manual' (1994, updated 1998) explicitly recommends that only direct expenditures be included in assessing the cost effectiveness of environmental policy measures. More particularly, it recommends, first, not to monetise additional externalities resulting from the measure in question and, second, not to include welfare effects due to behavioural change.

In the intervening period, however, numerous national and international studies have been published that recommend adopting a welfare-economic analysis that does include these kinds of effects. Particularly in the light of the recently published Dutch 'Guidelines for Social Cost Benefit Analysis', in which a welfare-economic analysis is likewise recommended, a new update of the 'Environmental Costing Methodology Manual' would appear to be warranted. A second motive for an update is that when the Manual was originally drawn up, environmental policy was focused more on prescribing specific technologies. In today's environmental policy, in which economic instruments and incentives for behavioural change play such a key role, there is an even greater need for a more comprehensive welfare-economic analysis. We see the following changes to the Manual as crucial:

- The current recommendation not to consider welfare losses due to behavioural change as costs and to exclude these from the analysis should be revised and recommendation given to monetise such losses of welfare unless there are reasonable grounds for deeming analysis on this point unnecessary. The latter will be the case for regulations on many concrete energy-saving technologies. It should also be recommended that inclusion of welfare losses due to behavioural change in the reported cost data be explicity mentioned. In addition, the calculation methodology employed should be clearly and transparently explained to data users, making clear what has and has not been included, and the effects of these choices on the final results.
- Instead of the current recommendation not to monetise additional externalities, it should be recommended to do so, including guidelines for that purpose (based on the recent 'Guidelines for Social Cost Benefit Analysis').

There are currently various institutes that consider the financial valuation of externalities already sufficiently robust for use as a basis for policy calculations. Others consider these estimates still too uncertain, though. For the sake of policy consistency and in the light of the aforementioned Guidelines, we recommend that discussions be held about the desirability and feasibility of extending these Guidelines to include a list of recommended monetary values for key externalities, for use in both social cost-benefit analysis and cost-effectiveness analysis.

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