Part I: Externalities and economic policies in road transport

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A B S T R A C T

Road transport imposes negative externalities on society. These externalities include environmental and road damage, accidents, congestion, and oil dependence. The cost of these externalities to society is in general not reflected in the current market prices in the road transport sector.

An efficient mobility model for the future must take into account the true costs of transport and its regulatory framework will need to create incentives for people to make sustainable transport choices. This paper discusses the use of economic instruments to correct road transport externalities, but gives relatively more weight to the problem of carbon emissions from road transport, as this is particularly challenging, given its global and long-term nature.

Economics offers two types of instruments for addressing the problem of transport externalities: command-and-control and incentive-based policies.

Command-and-control policies are government regulations which force consumers and producers to change their behaviour. They are the most widely used policy instruments. Examples include vehicle emission and fuel standards in the US as well as driving or parking restrictions in Singapore. The implementation cost of these instruments to the government is small. Although from an economic perspective these policies often fail to achieve an efficient market outcome, the presence of political constraints often make them the preferred option, in terms of feasibility and effectiveness.

Economic theory shows how policies, which affect consumption and production incentives, can be used to achieve the optimal outcome in the presence of externalities. Incentive-based policies function within a new or an altered market. We first examine incentive-based policies, which cap the aggregate amount of the externality, such as carbon emissions, by allocating permits or rights to the emitters. The emitters are then free to trade their permits amongst them. The permit allocation mechanism is important—although market efficiency would be satisfied by an auction, political influences usually favour a proportional allocation based on historic emissions. We discuss EU ETS as an example of a cap-and-trade system, however, no such policy for CO\textsubscript{2} emissions in road transport has been implemented anywhere in the world to date.

Fiscal instruments are, like command-and-control, widely used in road transport, because they are relatively cheap and simple to implement. They include the use of taxes and charges in order to bridge the gap between private and the social costs and, in principle, can lead to an efficient market solution. Registration, ownership, fuel, emissions, usage taxes, and parking and congestion charges have been implemented in many countries around the world. On the other side of the spectrum, subsidies can be given to those scrapping old cars and buying fuel-efficient vehicles. Some cities, such as London, have implemented congestion charges and many states in the United States have introduced high occupancy lanes. Other interesting possibilities include pay-as-you-drive insurance and other usage charges. However, the size and scope of taxes and subsidies are determined by governments, and because of their imperfect knowledge of the market the outcome is still likely to be inefficient.

Governments have many effective economic instruments to create a sustainable road transport model. These instruments can be used separately or together, but their implementation will be necessary in the nearest future.

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1. Introduction

Road transport plays an essential role in today’s world economy. For example, in 2004 the road share of passenger-kilometres was 89 per cent for the US and 85 per cent for the EU-25 (Eurostat, 2007, Table 5.24, p. 103). In 2003 the road share of tonne-kilometres was 33.4 per cent for the US and 72 per cent for the EU-25 (Eurostat, 2007, Table 5.1).

An efficient equilibrium is defined as a situation in which marginal social costs are equal to marginal social benefits. Externalities are a form of market failure, which means that the market is incapable of reaching an efficient equilibrium.

Although the presence of negative externalities in the transport sector was never in doubt, there has recently been some debate with respect to the presence of positive externalities. On the one hand, the idea of transport being associated with positive externalities seems to be mistaken. While positive externalities would materialise in the form of increased productivity and economic growth made possible by transport, to the extent that increased productivity and growth lead to increased individual income, these benefits are not external (European Commission, 1995). On the other hand, if transport (in the form of, for example, transport infrastructure investment) has an impact on agglomeration economies\(^1\) then it can certainly be thought of as causing positive externalities. Venables (2007) argues that an increase in employment resulting from a transport project is associated with two positive externalities. First, a transport project which increases employment increases the productivity not just of the new workers but also of the workers who were employed before the project went ahead and who now reap the benefits of a larger urban agglomeration. Second, higher urban productivity and associated higher wages (net of taxes) are, in equilibrium, matched by higher commuting costs and higher rents. The key point here is that this equality of higher commuting costs, higher rents and higher wages is achieved with wages net of taxes. The value of the extra product by the new workers is higher than the costs incurred, with the difference going to the government, in the form of income tax (Venables, 2007).

As fascinating as the debate on positive externalities from road transport may be, with the UK Department for Transport (UK DfT) currently revising their guidelines on how to appraise transport projects, the focus of the present review is on negative externalities.

As advanced above, in the presence of a negative externality the market is incapable of reaching an efficient equilibrium, when the total benefit of road transport is maximised (there is no way of increasing any road user’s benefit without reducing another’s).

From an economic theory point of view, this problem can be solved by implementing corrective instruments, such as for example Pigouvian taxes or cap-and-trade systems, which can achieve efficiency, or at least reduce the magnitude of the inefficiency or deadweight loss. In addition, there are a number of complementary policies, for instance, land use and transport planning, incentives to public transport and information campaigns, which can act in combination with corrective instruments to achieve more efficient outcomes. Finally, in some cases, it is simply a question of implementing a regulation and introducing a new technology. The example of catalytic converters has become standard in the road transport literature.

This paper concentrates on economic policies, stretching the concept to include command-and-control measures. Although the idea of corrective instruments to equate marginal social costs with

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\(^1\) Agglomeration economies refer to the benefits that economic agents obtain when locating near each other. For example, when firms are spatially clustered together they tend to face lower production costs and greater access to markets to buy from and sell to. The concept applies both to firms within the same industry and in different industries.
marginal social benefits is attractive in theory, there are three problems associated with the implementation of such instruments in the transport sector. The first problem is that the marginal external cost imposed by vehicles is not easy to measure. Even when the marginal physical damage can be assessed, monetisation is not always straightforward. Hence, typically, it is virtually impossible to implement first best policies. The second problem relates to the fact that any new instrument will usually be introduced in a system characterised by pre-existing corrective instruments and regulations. As a consequence, the interaction between new and existing instruments may lead to over-charging (thus, double-internalising the externality) and therefore to inefficient outcomes that reduce welfare (Zatti, 2004). The third problem is that even in the case where marginal external costs could be perfectly measured and first best policies implemented in the road transport sector, there would be no guarantee of an efficient outcome due to distortions in other (related) sectors in the economy, which are not priced according to marginal cost. For example, inefficiencies caused by adverse selection in the insurance sector will affect the efficiency of the road transport sector. Another important example of a related sector is the labour market, which does not typically operate perfectly, mainly due to the presence of distortionary labour taxes (Verhoef, 2000, p. 328).

Bearing that caveat in mind, this paper consists of a survey that concentrates on road transport economic policies in theory and in practice.

There is consensus in the literature that the most important negative externalities from road transport include accidents, road damage, environmental damage, congestion and oil dependence (Maibach et al., 2008; Newbery, 1990; Parry, Walls, & Harrington, 2007; Small & Verhoef, 2007). We now turn to briefly describing these externalities before moving on to the discussion on policies.

1.1. A brief summary of the main road transport externalities

1.1.1. Accidents

Accident externalities arise ‘whenever extra vehicles on the road increase the probability that other road-users will be involved in an accident’ (Newbery, 1990, p. 24). Although it is reasonable to assume that the more vehicles there are, the higher is the probability of accidents, there is evidence of compensating behaviour, as drivers tend to drive at lower speeds and pedestrians and cyclists tend to be more careful in heavier traffic (Newbery, 1990, p. 24; Parry et al., 2007, p. 381). The impact of one extra vehicle on the severity-adjusted accident risk is difficult to pin down (Brownfield et al., 2003). An extra vehicle increases the collision risk, but because vehicles would drive slower due to higher congestion, each collision would be less deadly. Although this may reduce the number and the severity of accidents, this compensating behaviour is itself costly, so it should not be considered a positive externality (Parry et al., 2007, p. 381). Unfortunately there is not much empirical evidence on the net effect.

External accident costs are the accident costs ‘not covered by risk oriented insurance premiums’ (Maibach et al., 2008, p. 36). If insurance premiums covered all accidents costs, the externality would then be fully internalised. In the US, for example, only about half of all accident costs are paid by private insurers (Blincoe et al., 2002).

Accident costs (regardless of whether they are internalised or not) include material damage, insurance administration, legal and court costs, police and fire services costs, medical costs, lost economic output, and the pain, grief and suffering imposed on the victims, and their friends and families (Maibach et al., 2008, p. 36; UK DfT, 2009a, p. 1).

Different countries adopt different methodologies to estimate accident costs, but they all, to a greater or lesser degree include the impacts listed above. The number of accidents and the number of casualties are therefore key quantitative indicators (UK DfT, 2009a, p. 1). In developed countries, these statistics are not usually problematic to collect. In developing countries, on the other hand, some accidents go unreported, or reported but not properly recorded (Pedersen et al., 2004, p. 53). The main complication, however, arises at the time of monetising these different accident costs. Although items such as material damages, various administrative costs, healthcare costs, and lost product are relatively straightforward to value, the pain, grief and suffering imposed on the victims, and their friends and families are not. Indeed, the most important and controversial component is the value of a life saved or the cost of a life lost.

Although there could be some instinctive rejection towards the idea of putting a value on human life, two points should be borne in mind. First, when estimating the value of a statistical life (VOSL), it is not the life of a specific human being in a given accident which is being valued, but rather, the value of the life of an average or representative victim before the accident takes place. Second, even when no such value is explicitly estimated, individuals make decisions and trade-offs in their daily activities, all of which involve higher or lower risks, and government departments choose some policies over others, thus, implicitly assuming a VOSL (Freeman, 2003, p. 11). The question is not how much an individual would be willing to pay to avoid his certain death or the certain death of another person (e.g., the ransom for a kidnapped relative) or how much compensation that individual would require to accept that death, as most people would be willing to pay everything they have and will ever have in order avoid certain death (Freeman, 2003, pp. 299–300, italics added by us). The VOSL is computed on the basis of willingness to pay to achieve a small reduction in the probability of death or willingness to accept compensation to accept a small increase in that probability over a certain period of time (Freeman, 2003, p. 300, italics added by us).

There is abundance of VOSL estimates in the literature, typically relating to one geographic area and time period. Examples include estimates for Chile (Rizzi & Ortuzar, 2003), Malaysia (Mohd Fauzi, Nor Ghani, Radin Umar, & Ahmad Hariza, 2004), Thailand (Vassanadumrongdee & Matsuoka, 2005), Sweden (Hultkrantz, Lindberg, & Andersson, 2006; Andersson, 2007).

4 Lost economic output is the output lost due to the death or injury of a person, which can in principle be estimated as the discounted present value of that person’s earnings over the remainder of his expected life. A representative victim is used for this calculation, as otherwise the elderly, already out of the work-force, and the disabled, would be less valued, and so would be young children, due to the discounting process before they enter the work-force, or women or people from ethnic minorities on lower wages. Other problems include whether that person’s productivity should be measured net of or inclusive of his own consumption, whether stay-at-home spouses’ productivity (completely outside the market) should be assumed equal to the productivity of the domestic service or to the opportunity cost of staying at home, which could be measured by the average earnings of workers (Freeman, 2003, pp. 302–303). It should be noted that lost economic output is just one element of the value of a statistical life (VOSL), discussed below, and does not constitute the whole VOSL, as the human capital approach would propose.

5 Accident costs are considerable: in the US, for example, they were estimated at 2.3 per cent of GDP in 2000. Property damage costs for all accident types (fatal, serious and slight) represented 26 per cent of all costs. Present and future medical costs due to injuries accounted for 14 per cent of the total costs (Blincoe et al., 2002). Accident costs are considerable: in the US, for example, they were estimated at 2.3 per cent of GDP in 2000. Property damage costs for all accident types (fatal, serious and slight) represented 26 per cent of all costs. Present and future medical costs due to injuries accounted for 14 per cent of the total costs (Blincoe et al., 2002).

6 In 1993, a Panel commissioned by the US National Oceanic and Atmospheric Administration with the task of assessing the reliability of contingent valuation methods concluded, amongst other things, that willingness-to-pay is a more reliable measure than willingness-to-accept compensation (Portney, 1994, p.5).

2 The problems are common to other areas of the economy as well.

3 At most, it might be possible to impose an ex-ante expected first-best policy.
1.1.3. Environmental damage

The environmental externalities from road transport include the impacts from emissions, noise and vibration, changes to landscape and townscape, impacts on biodiversity, heritage of historic resources, and water (UK DfT, 2004). From these externalities the two which have been best quantified and monetised are noise and emissions. The others tend to be assessed using qualitative techniques. These may include a description of the problem in question (such as for example, the threatened biodiversity or heritage), an assessment of its importance, and an overall scoring on a predetermined scale.

Noise harms human health and interferes with people’s daily activities. According to the World Health Organization (WHO), noise from road transport affects the health of almost one third of people in Europe (WHO, 2007). The main impacts from noise include ‘pain and hearing fatigue, hearing impairment, annoyance, interferences with social behaviour (aggressiveness, protest and helplessness), interference with speech communication, sleep disturbance and all its consequences on a long and short term basis, cardiovascular effects, hormonal responses (stress hormones) and their possible consequences on human metabolism (nutrition) and immune system, performance at work and school’ (WHO, 2007). Protection from noise pollution is costly: households may have to double-glaze their windows and local governments often install noise barriers around motorways. Noise can be measured in decibels and noise costs are typically estimated with hedonic property price models (Parry et al., 2007).

Most of the energy consumed by road transport comes from fossil fuels. This causes emissions, not just from fossil fuel combustion but also from the evaporation of petrol during production, storage and distribution, and evaporative emissions from the gas tank and carburettor of petrol-engined vehicles. Here, however, we concentrate on emissions due to fossil fuel combustion, which is the main source of road transport emissions. Emissions from road transport have negative impacts at local, regional and global level.

At local and regional level, there are a vast range of air pollutants, which cause a variety of effects on the environment and health. The main emissions from road transport with effects at local level are nitrogen oxides, hydrocarbons and carbon monoxide, accounting for 58 per cent, 50 per cent and 75 per cent respectively of all such emissions in the EU (European Parliament Fact Sheets, 2006). Other pollutants include sulphur dioxide and particulate matter. Their impacts are summarised below.

1.1.3.1. Nitrogen oxides (NOx).

NOx emissions are formed in the combustion of fossil fuel. They include nitrogen dioxide and nitric oxide. Nitrogen dioxide can negatively impact the human respiratory system and reduce lung function. NOx also contribute to the formation of ozone, which is a harmful secondary pollutant in the lower atmosphere. High levels of ozone increase susceptibility to respiratory disease and irritate the eyes, nose throat and respiratory system, particularly in urban areas (UK DfT, 2007a, p. 51).

1.1.3.2. Hydrocarbons (HC).

HC are the result of incomplete combustion of fossil fuels and cause eye and throat irritation and...
1.1.3.3. Carbon monoxide (CO). CO is also produced by the incomplete combustion of fossil fuels and interferes with the absorption of oxygen. After being inhaled, CO molecules can enter the bloodstream, where they inhibit the delivery of oxygen throughout the body. This in turn leads to dizziness, headaches, and fatigue, and can affect fertility and general levels of health. CO interferes with respiratory bio-chemistry and can affect the central nervous and cardiovascular systems (UK DT, 2007a, p. 50). Other pollutants can exacerbate its effects (Banister, 1998, p. 4). The fitting of catalytic converters to all new vehicles virtually around the world in the late 1980s and throughout the 1990s has substantially reduced emissions of CO.

1.1.3.4. Sulphur dioxide (SO₂). SO₂ affects the lining of the nose, throat and airways of the lung, in particular, among those who suffer from asthma and chronic lung disease (UK DT, 2007a, p. 51). It causes respiratory illness, in particular bronchitis, and it also contributes to acid rain (Banister, 1998, 4).

1.1.4. Particulate matter (PM). PM is a generic term used to describe a complex group of air pollutants that vary in size and composition. The two main types of interest here are PM₁₀ and PM₂.₅, which are particles with aerodynamic diameters of up to 10 microns and 2.5 microns, respectively.⁸ PM has been linked to numerous adverse health effects including increased hospital admissions and emergency room visits, respiratory symptoms, exacerbation of chronic respiratory and cardiovascular diseases, decreased lung function, and premature mortality (Bell, Samet, & Dominici, 2003, p. 8).

Particulates are the main pollutants causing deaths in Europe today, with an estimate of 348,000 premature deaths in Europe due to exposure to anthropogenic PM₂.₅ for the year 2000 (European Environment Agency, EEA, 2005a, p. 98) or loss of statistical life expectancy of approximately 9 months (EEA, 2005b, p. 47). In some areas of Europe, such as the Benelux area, in northern Italy and in parts of Poland and Hungary, the average loss of life expectancy from particulates is up to two years (EEA, 2005a, p. 98).

Impacts on health, crops, buildings, etc from local and regional pollutants can be physically measured and monetised.⁹ Once the link between emissions and concentration of a pollutant has been established, the next step involves the estimation of a dose-response function, which relates pollutant concentration to the physical impact on the receptor (for example, population in a town). Finally, the impact resulting from exposure can be valued using stated and/or revealed preference methods and/or combinations of both.

It should be noted however, that emission rates per km of air pollutants with local and regional impacts has been decreasing in virtually all countries, due to the introduction of ever more stringent regulations on new vehicle models.

At global level, the main pollutant is carbon dioxide (CO₂), the anthropogenic greenhouse gas (GHG) which most contributes to global warming, and the unavoidable product of fuel combustion. In the period 1990–2002 the growth rate of energy consumption in the transport sector was the highest among all the end-use sectors, and road vehicles account for 77 per cent of that energy use (Kahn Ribeiro et al., 2007, p. 328). With 95 per cent of transport energy coming from oil-based fuels and only small differences in the carbon content of the different fuels, CO₂ emissions from the different transport sub-sectors are roughly proportional to their energy use (Kahn Ribeiro et al., 2007, p. 328). Indeed CO₂ emissions are closely related to the quantity of fuel consumed, and thus, as we shall see in the sections that follow, has important implications for policy instrument design.

Valuing the impact of global warming is undoubtedly a challenging task. A number of models have been developed¹⁰ with estimates varying according to the impacts (and some times adaptation) taken into account, the assumptions made, and the discount rate used.

1.1.4. Congestion

Traffic congestion is a road condition characterised by slow speeds. It takes place when the demand for road space is greater than road capacity. The impacts are well-known: longer and unreliable travel times and ultimately, negative economic effects as a result of an inefficient distribution and delivery of goods, services and resources.

In his seminal paper on the economics of congestion, Walters (1961) established the isomorphism between travel time on a given length of highway as a function of traffic flow (speed-flow relationship), on one hand, and average cost as a function of flow, on the other hand. That was Walters’ fundamental original contribution. An adequate conversion from time into cost, combined with a demand curve, makes the analysis of the whole picture from an economic point of view possible. Fig. 1 presents the standard diagram of the economics of traffic congestion on a link.

Higher traffic flows lead to lower average speeds and higher travel times and costs per km. Additional traffic imposes an external cost on all other road users. Under congested conditions, particularly in urban areas, and in the absence of efficient road pricing, traffic will be undercharged and hence, excessive.

In Fig. 1, road users are assumed identical apart from their marginal willingness to pay for a trip, given by the inverse demand curve D, which represents marginal private and social benefits. Marginal private (MPB) and marginal social benefit (MSB) are assumed identical (i.e., distributional considerations are ignored). The efficient equilibrium is at point H, where the marginal social

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⁸ One micron is one micrometer or one millionth of a meter (1 mm = 1000 μm, where μm: microns).

⁹ Spadaro and Rahl (2001), for example, argue that the impacts on health account for more than 90 per cent of the cost of automotive air pollution.

¹⁰ These models are reviewed in Schneider and Lane (2005), Hope (2005) and Stern (2006).
cost (MSC) is equal to the MSB. The market equilibrium however is at point C, where the average social cost (ASC) is equal to the MSB. At this point drivers are paying for their ASC but not for their MSC, leaving thus a congestion externality or marginal congestion cost (MCC), which can be measured by the segment MC. The inefficiency of incorrect pricing of scarce road resources or deadweight loss is then measured by the area HMC.

It should be noted that the MCC at the efficient level of traffic $q^*$ is positive and equal to the segment HE. In other words, zero congestion would not be an optimal situation, but rather, would reflect a waste of resources. Estimates of congestion costs are mainly based on assumptions on the value of time and value of reliability.\(^{11}\) These have been subject to extensive empirical and theoretical research. Some important recent contributions include Brownstone and Small (2005), Calfee and Winston (1998), De Borger and Fosgerau (2008), Fosgerau (2006), Hensher (2006), Hensher (2008), Hensher and Goodwin (2004), Lam and Small (2001), Mackie et al. (2003), Ramjerdi and Dillen (2007), Steinietz and Brownstone (2005), Tseng and Verhoef (2008), and, Wardman (2001).

There is substantial heterogeneity in values of time and reliability across the population and these depend on a number of factors. Some widely agreed upon determinants include income and trip purpose. Although the correlations are far from perfect, in general we expect that a higher income leads to a higher value of both working and non-working time; and the more important the trip purpose (work as opposed to shopping, and their corresponding delay penalties) the higher the value of time savings. The value of reliability also varies with income and trip purpose, although gender differences, reflecting childcare commitments, also have an influence.

Elasticities of the value of time with respect to income are also important. Mackie et al. (2003) conduct a meta-analysis of value of time studies and conclude that reasonable assumptions would be 1 for the working time and 0.8 for the non-working time elasticities, respectively (p. 11) and the UK DfT has adopted these values (UK DfT, 2009b, p. 5).

### 1.1.5. Oil dependence

Oil dependence is a problem for oil importing countries. These countries are vulnerable to volatile oil prices and oil price shocks. This has the potential of threatening national security and the economy of a country, especially if this country does not have enough market power to influence world prices.

Roughly 42 per cent of world oil is produced by OPEC countries, and from this, 68 per cent comes from Iran, Iraq, Kuwait, Saudi Arabia and United Arab Emirates (Parry & Darmstadter, 2003, p. 3). These countries, together with Venezuela, and two non-OPEC countries, Mexico and Norway, are willing to produce below their capacity in order to keep their influence over world prices (Parry & Darmstadter, 2003, p. 5). OPEC is an organization that influences production and ultimately prices (Kaufmann, Bradford, Belanger, McLaughlin, & Miki, 2008) via capacity utilisation, production quotas, and the degree to which OPEC production exceeds these quotas (Kaufmann, 2004, p. 81).

Virtually every activity in developed countries relies on oil directly and/or indirectly, and as a consequence, world oil prices impact GDP, other goods and services' prices, and employment, to name just a few economic indicators (Cleveland & Kaufmann, 2003, p. 488), as well national security. Reducing vulnerability to oil price volatility requires a reduction in oil use, regardless of where it is produced (Cleveland & Kaufmann, 2003, p. 488).

The question of how to measure the costs of oil dependence has attracted some attention, especially in the US (Greene & Ahmad, 2005; Leiby, 2007; Parry & Darmstadter, 2003). These costs include the transfer of wealth from the importing country to the oil producing country, a decrease in the maximum product the importing country would be capable of if oil prices were competitive (as opposed to monopolistic or oligopolistic) and any macro-economic adjustment costs driven by price shocks, which impact prices and wages and as a consequence, employment of resources, such as capital and labour (Leiby, 2007). Some consequences of oil dependence, such as its effect on foreign policy and national security, are much more difficult to quantify (Leiby, 2007).

### 1.2. Command-and-control vs. incentive based policies

Having summarised the most important road transport externalities the next question is whether anything ought to be done about them and why. As explained above, the market is incapable of reaching an efficient equilibrium in the presence of externalities. This calls for some action from the government in order to correct the market failure, at least in part, and achieve, or at least get close to, an efficient equilibrium.

From an economic point of view, policies to reduce negative externalities can be divided in two groups: command-and-control (CAC) and incentive based (IB). IB mechanisms can be further categorised into price controls and quantity controls.

A CAC policy is essentially a regulation, or a “command”, which needs to be “controlled” or enforced. When there is an externality, the regulator (or government) can impose a maximum level of the activity causing it, or a restriction over the behaviour of economic agents, or characteristics of products. CAC measures give rise to predictable outcomes and are relatively easy to implement, enforce, and understand (Button, 1990). However, they are inflexible and they do not provide incentives to go beyond the mandatory standard (Button, 1990). Every time a change is envisaged, the measure needs to be revised, thus leading to an additional bureaucratic burden.

IB policies, on the other hand, provide economic incentives to the targeted agents and act to directly alter private utility or private benefit from a given behavioural response. Consequently, they are crucial instruments to induce behavioural change.

Price control IB policies put a price on a good or an activity, such as emissions or congestion. These can take the form of taxes (or fees or charges) or subsidies, both providing incentives to reduce the level of the externality. A foregone subsidy can have a similar impact to that of a tax.

A tax to internalise an externality, also known as a Pigouvian or corrective tax\(^{12}\) (or fee or charge) is a tax set equal to the marginal external cost of the activity at the efficient equilibrium. Thus, the corrective tax brings the marginal private cost up to the level of marginal social cost.

A subsidy, on the other hand, encourages the agent to reduce the level of activity by compensating him for his loss. Thus, a car manufacturer may be given a subsidy to produce cleaner cars or a road user may be given a subsidy as long as he stops travelling by car during the rush-hour.\(^{13}\) However, it may be difficult to measure baseline emissions to estimate emissions reductions under a subsidy.

More importantly, Baumol and Oates (1988, chap. 14, pp. 211–234) highlight a number of differences between the impacts from taxes and subsidies at firm and industry level:

\(^{11}\) The cost curves in Fig. 1, if expressed in monetary units, are obtained by assuming a value of time.

\(^{12}\) These taxes are called Pigouvian in honour of Arthur Pigou, a Cambridge Neo-classical economist, who also developed the concept of externalities. They are also called corrective because they are aimed at correcting a market distortion.

\(^{13}\) In practice this could translate, for example, into a subsidy to public transport fares.
• It may be profitable for the firm to start off by emitting more than it would have otherwise in order to qualify for larger subsidies.
• There is less of an incentive to introduce new technologies under subsidies than under taxes.
• An unprofitable firm under taxes may become profitable under subsidies.
• While taxes will typically drive firms out of a competitive industry and this will usually result in a decrease in its production, a subsidy may attract new entrants, thus increasing the level of production of that industry.
• Under perfect competition, a subsidy will yield a reduction in emissions at firm level, but an increase in emissions at industry level, even beyond the level of emissions there would have been without any intervention.

Quantity control IB policies usually assume the form of cap-and-trade systems. They target a certain level of activity (for example, emissions), assign property rights (permits) to match the targeted total quantity and let consumers, firms and other entities trade these permits at an endogenously determined price. To the extent that quantity control instruments involve a trading mechanism, they also provide price incentives to the regulated parties (Hepburn, 2006). Cap-and-trade systems were theoretically articulated by Dales (1968), who built his ideas on the basis of work by Coase (1960).

The rights or permits initial allocation method is a decision which needs to be made early in the process. There are mainly two possible initial allocation methods: grandfathering and auctioning. With grandfathering, permits are allocated by the regulator for free to each polluter, according to their past (historic) emissions. With auctioning, polluters pay for the permits they wish to buy, according to the price that emerges from the auction.

Auctioning reduces barriers to entry in the case of industry, increases regulation stringency, prevents the possibility of windfall profits, and generates revenues that can be recycled for environmental purposes and/or cutting distortionary taxes. Grandfathering, on the other hand, is a more widely accepted option, as essentially trading parties are allocated permits for free, but does not apply the “polluter pays” principle in full.\textsuperscript{14}

The main advantage of IB (both price and quantity control) instruments over CAC ones is their cost effectiveness. Economic agents (producers or consumers) with low costs of abatement will find it relatively easier to reduce their level of externality than to buy permits or pay taxes, while those with high costs of abatement will prefer to buy permits or pay taxes. The cost of reducing the externality is thus minimised compared to the more direct regulatory approach of setting standards (Baumol & Oates, 1988, Chapters 6 and 8). Having said that, the magnitude of these cost savings varies greatly across specific cases (Newell & Stavins, 2003, p. 46). Tietenberg (1985, in Newell & Stavins, 2003) compares the costs of air pollution control across ten different examples and finds ratios ranging from 1.1 to 22.\textsuperscript{15}

Another advantage of IB instruments over CAC is that they lead to self-revelation of private information through the choice made by regulated consumers or producers; and they exploit the market’s capability of private information aggregation (Hepburn, 2006, p. 228). This makes IB instruments more efficient than CAC measures when the information basis is not complete and cannot be inferred by the regulator. The advantage of IB instruments over CAC is particularly marked when the regulated entities are not homogeneous\textsuperscript{16} and their optimal response is not uniform (Hepburn, 2006, pp. 228–229), or when there are time and spatial variations in problems (European Commission, 1995).

However, IB mechanisms can lead to market failures and blunt miss of objectives. This can happen, as explained above, when incentives are not correctly calibrated due to difficulties in determining the marginal external cost (and therefore of efficient level of activity), the system already having corrective instruments and regulations in place (whose effects will be added to the new IB instrument’s), and the presence of distortions in other (related) sectors in the economy. In these cases, the effectiveness of the instruments in meeting the targets may be impaired. Transaction and implementation costs may also be higher for IB mechanisms than for CAC, due to the need of advanced monitoring systems (European Commission, 1995). However, costs are likely to decrease as technology progresses.

Finally, CAC may be preferable to IB instruments when there are no informational asymmetries, there is little risk of government failure, and the regulated entities are required to respond in the same way. An instance when CAC would be advisable is when the optimal level of an externality, for example a certain polluting activity, is zero. The activity in this case should simply be banned (Hepburn, 2006, p. 229). This happens when after reducing emissions by 100 per cent the marginal social benefit is higher than the marginal social cost. Obvious cases of this are bans on the use of dichlorodiphenyltrichloroethane (DDT) as a pesticide and asbestos in construction, which were implemented by many countries in the 1980s. The lead phase-out from petrol, discussed in Section 2, is another example.

1.3. Incentive based: quantity vs. price control under asymmetric information

A cap-and-trade system guarantees a quantity of externality or activity producing the externality, regardless of the abatement costs. A tax, on the assumption that polluters (either consumers or producers) are cost minimisers, guarantees that the marginal abatement cost will be equated to the tax, regardless of the resulting quantity of the externality or activity producing the externality (Baumol & Oates, Chapter 5).

Under perfect information a system of Pigouvian taxes and a system of marketable permits are equivalent and yield the same efficient outcome. In general ‘it is neither easier nor harder to name the right prices than the right quantities because in principle exactly the same information is needed to correctly specify either’ (Weitzman, 1974, p. 478).\textsuperscript{17} If the optimal number of permits is issued by the regulator, their price will be bid up on the free market to precisely the level of the Pigouvian tax. At that point, it makes no difference to the polluter whether he pays a tax equal to his marginal external cost to the authorities, or whether he pays that same amount to buy permits that will allow him to produce the same level of externality and marginal external cost (Baumol & Oates, 1988, chap. 5).

\textsuperscript{14} At first sight, an initial free allocation of permits might lead to the conclusion that the “polluter pays” principle is not applied at all. However, one must consider the opportunity costs that come with the use of permits: by using a permit, the permit holder is giving up the earning he would make by selling it at the market price. The incentive, however, is smaller than in the case of auctions, where polluters must pay upfront for the permits.

\textsuperscript{15} Newell and Stavins (2003) attempt to fill the niche in research on the relation-ship between the nature and magnitude of the heterogeneity in abatement costs and the potential cost savings from IB instruments. They develop an analytical framework linking cost savings from IB instruments relative to CAC and different sources of heterogeneity in abatement costs, including the slope of the marginal cost curve, baseline emissions and firm size.

\textsuperscript{16} Permit trading provisions may, however, solve the cost heterogeneity problem.

\textsuperscript{17} It should be noted, however, that in the case of road transport most pollutants with local or regional impacts, exhibit a marginal damage curve which is approximately flat. In such cases, the optimal tax will be the same, regardless of the shape of the demand (or marginal social benefit) curve. Under these assumptions estimating the efficient tax requires less information than estimating the efficient quantity.
In an imperfect information setting, the conclusions are very different. Lack of enough relevant information on the part of the regulator is a crucial factor that can lead to regulatory failure under price as well as under quantity control mechanisms.

The source of uncertainty can lie in the marginal abatement cost curve or in the marginal social benefit curve. When there is perfect information regarding marginal abatement costs but lack of information regarding marginal social benefits, the policy, either taxes or permits, will not be optimal. The cost of the error, however, will be the same under either policy (Weitzman, 1974, in Baumol & Oates, 1988) and so in the presence of uncertainty with respect to marginal social benefits it makes no difference to choose one or the other. The reason for this is that the resulting permit price, after the regulator has determined what he thinks is the optimal quantity, and the resulting quantity, after the regulator has determined what he thinks is the optimal tax, depend exclusively on marginal abatement costs and are independent from marginal social benefits.

When there is imperfect information regarding marginal abatement costs, the outcome will be inefficient. In this case, the cost of the error under a permit or tax system may be different.

In general, both permits and taxes will yield inefficient outcomes. When the assumed marginal abatement costs are lower than the actual ones, the reduction of the externality under a cap-and-trade system will not be enough, and the reduction of the externality under a tax will be excessive. The opposite holds when the assumed marginal abatement costs are higher than the actual ones (Baumol & Oates, Chapter 5).

The inefficiency arising from lack of information on marginal abatement costs under permits and taxes depends on the relative slopes of the marginal abatement cost and marginal social benefit functions. An important result from Weitzman (1974), digested in Baumol and Oates (Chapter 5) is that:

- When the marginal social benefit and the marginal abatement cost curves are linear and the absolute values of their slopes are equal, the magnitude of the distortion caused by cap-and-trade and a tax will be identical (p. 67).
- The steeper the slope of the marginal social benefit curve, the less severe the magnitude of the distortion caused by a cap-and-trade system will be, compared to that of a tax (p. 64).
- The steeper the slope of the marginal abatement cost curve, the more severe the magnitude of the distortion caused by a cap-and-trade system will be, compared to that of a tax (p. 66).

Finally, when the slope of the marginal benefit and marginal cost curves are both uncertain, quantity control will in general be preferred, since ‘prices can be a disastrous choice of instrument far more often than quantities can’ (Weitzman, 1974, p. 486).

Taxation leads to price stability, while permits may lead to price fluctuations. The volatility associated with permits will add up to the intrinsic uncertainty over returns in technology development, and may reduce investment incentives by risk-averse firms the intrinsic uncertainty over returns in technology development, fluctuations. The volatility associated with permits will add up to more often than quantities can (Weitzman, 1974, p. 486). Finally, a multiple instrument package can also be implemented (Hepburn, 2006, p. 231). Such a package can include permits, taxes, subsidies and regulations. The danger in this case, in terms of efficiency, is the double-charging or double-penalising of an externality, as its reduction in that case would be excessive.

Aldy, Ley, and Parry (2008) discuss the design of CO2 taxes at the domestic and international level and the choice of taxes versus permits. They find a strong case for taxes on uncertainty, fiscal, and distributional grounds, especially if permits are grandfathered. First, taxes provide certainty with respect to the price for emissions, and therefore the marginal abatement costs, while permits fix quantity but not price, which fluctuates with market conditions, and hence, the uncertainty is larger. Second, unlike permits distributed for free, taxes offer revenues to the government, helping a healthier fiscal balance. These revenues can then be used to reduce distortionary taxes in the economy, thus increasing efficiency.

Third, governments can use revenues from CO2 taxes or auctioned permits to benefit lower income groups, thus reducing any initial regressive effects. They point out that the main disadvantage of taxes is that if they were implemented at an international level, individual countries could reduce other taxes, even taxes on sources of other GHGs, to offset the burden. To solve this problem they propose a CO2 tax which takes into account pre-existing energy taxes or subsidies.

1.4. Concluding remarks

The most important externalities caused by road transport are accidents, road damage, environmental damage, congestion and oil dependence. In the presence of externalities the market is incapable of achieving an efficient equilibrium. This problem can in theory be solved by implementing corrective taxes or introducing cap-and-trade systems. These are equivalent instruments under perfect information but errors can be made under imperfect information. The cost of the error is identical if there is lack of information regarding marginal social benefits or if there is lack of information regarding marginal abatement costs and the marginal social benefit and the marginal abatement cost curves are linear with equal slopes in absolute value. If there is lack of information regarding the marginal abatement cost and the slopes of the cost and benefit curves are different, the cost of the error varies depending on which curve is steeper. There are also a number of complementary policies, such as for example, land use and transport planning, which can act in combination with corrective instruments to reduce the level of externalities. Command-and-control measures and new technologies also have a very important role to play.

2. Command-and-control policies

As explained in Section 1.2, CAC policies are not efficient from an economic point of view: even in a context of perfect information when the regulation or standard is set at the optimal level, the target is not achieved at minimum cost, and, worse yet, the social costs could exceed the potential benefits (thus replacing market failure with government failure). Despite being inefficient policy instruments, they are the most widely used ones to regulate environmental and other externalities. In many cases, however, the
choice of instrument is justified by the severity of the problem, with the extreme case of lethal fumes being the prototype example. The transport sector is no exception, and throughout the world there is a plethora of examples of CAC policies targeting different transport externalities. In this section, some of these examples are briefly described and discussed, mainly to illustrate how the regulations were set and to give an idea of their success or failure.

2.1. Fuel standards

These are standards that countries impose on motor-vehicle fuels. The most notable example is the ban on lead in petrol, which has been implemented virtually all over the world as of 2009. Lead had been used as an additive since the 1920s. It was a pollutant of great concern, mainly due to its effects on children’s brains and was therefore phased out and finally banned in most countries.

In the US, for example, the Environmental Protection Agency (US EPA) started a lead phase-down program in 1973, with the objective of reducing the lead content in petrol from 1.7 grams per gallon (0.45 grams per litre) down to 0.5 grams per gallon (0.13 grams per litre) by 1980 in large refineries and by 1982 in small refineries. The standard allowed refineries to average their total (both leaded and unleaded) output to reach the 0.5 standard. In 1982, the US EPA loosened the standard to 11 grams per gallon (0.29 grams per litre) but eliminated the provision that allowed averaging between unleaded and leaded petrol. In July 1985 the standard was tightened again to 0.5 grams per gallon (0.13 grams per litre) and finally, in January 1986, to 0.10 grams per gallon (0.03 grams per litre) (US EPA, 1985). Although this was mainly a CAC policy, it was also complemented with a form of credit trading, an early ancestor of cap-and-trade. By 1995, unleaded petrol sales accounted for 99 percent of the petrol market in the US (US DOT, 2008b).

Meanwhile, in the UK the maximum amount of lead allowed in petrol was reduced from 0.45 grams per litre to 0.40 in 1981 and then again to 0.15 grams per litre in December 1985. In 1986 unleaded petrol was first sold in the UK and a final ban was imposed on leaded petrol at the end of 1999 (UK DfT, 2007a, p. 51).

Most developed countries followed similar paths in the 1980s and 1990s. On top of that, the United Nations Environment Program (UNEP) led a campaign to eliminate leaded petrol completely everywhere in the world and as of 2009 very few countries still use it.

Fuel standards are widely used in many (mainly developed) countries and thanks to them, the emissions of benzene, sulphur dioxide, and other harmful pollutants have been reduced.

Much less common are regulations on CO₂ emissions from fuel. One exception is the Low Carbon Fuel Standard, introduced by the state of California in 2007. The Low Carbon Fuel Standard requires fuel providers to reduce GHG emissions of the fuel they sell. The programme intends to achieve a 10 percent reduction in the carbon intensity of transport fuels by 2020 (Farrell & Sperling, 2007).

2.2. Vehicle standards

Vehicle standards are CAC policies which typically regulate vehicle safety, tailpipe emissions and fuel efficiency. In general, different countries set their own vehicle standards, although the EU sets standards for all its members.

Typically, safety standards set front and side impact tests which vehicles have to pass before they are introduced into the market. There are also regulations regarding compulsory fitting of head-restraints and seat-belts, and on the minimum depth of tyre treads, brakes, and annual safety checks (Acutt & Dodgson, 1997, p. 19). In general, safety standards have become more stringent over time and newer vehicles tend to be safer than older ones.

One prominent example of a CAC policy relating to tailpipe emissions is that of catalytic converters. These were first introduced in new cars in the US in 1975 in order to reduce the toxicity of emissions from internal combustion engines. The converters also facilitated the compliance with the Clean Air Act of 1970, according to which all new vehicles sold in the US from 1975 onwards had to meet US EPA standards on HC, NOx and CO emissions (US EPA website b).

The EU made catalytic converters mandatory in new cars with the Council Directive 91/441/EEC of 26 June 1991, on measures to be taken against air pollution by emissions from motor vehicles (Council of the European Communities, 1991). The main systems of vehicle emission standards are those from the US and the EU. Having said that, most developed countries (and some developing ones), including Australia, China, Japan, Switzerland, South Korea and Taiwan have implemented some type of fuel economy or CO₂ standard (Clerides & Zachariadis, 2008, p. 2658), although in many instances they closely follow (or import) the standards in place in the US or Europe (Faiz, Weaver, & Walsh, 1996).

In general, when a country or group of countries adopts a standard, this standard applies to new vehicles sold, rather than to those already on the road. The US was the first country in the world to set standards for vehicle emissions. Under the North American Free Trade Agreement these standards were later also adopted by Canada and Mexico (Faiz et al., 1996, p. 1).

There are a number of landmarks that have shaped the development of vehicle emission standards and fuel economy around the world. One important such landmark was the introduction of the Corporate Average Fuel Economy (CAFE) in the US, enacted by Congress in 1975 and still in place as of 2009. The purpose of CAFE is to reduce energy consumption by increasing the fuel economy of cars and vans. Regulating CAFE is the responsibility of the US National Highway Traffic Safety Administration (US NHTSA) and the US EPA. The US NHTSA sets fuel economy standards for cars and vans sold in the US, and the US EPA calculates the average fuel economy for each manufacturer (US NHTSA website a). Typically, a manufacturer can meet the standard by producing fewer large vehicles and more small vehicles or by improving the mileage of all of the vehicles it produces.

20 Apart from Bosnia and Herzegovina, which plan to phase-out leaded petrol by January 2010 and Serbia, which will probably do it around 2015–2020, no phase-out date is yet known for any of the following countries: Macedonia, Kazakhstan, Tajikistan, Turkmenistan, Afghanistan, Fiji, Micronesia, Myanmar, North Korea, Solomon Islands and Tonga (UNEP, 2007). Uzbekistan had plans to phase it out in 2008 but Taylor (2008) lists the country as not having done so. The rest of the world has completely phased leaded petrol out thanks to standards, regulations and bans, implemented on a country-by-country basis.

21 It should be noted that although catalytic converters remove HC, NOx and CO harmful to human health and the local air, the exhaust gases leaving the engine through the catalytic converter are two important GHGs: carbon dioxide, which remains in the atmosphere for a long time, and nitrous oxide.

22 California was the first US state to develop vehicle emission standards. Often, Californian standards were later adopted at federal level (Faiz et al., 1996, p.2).
CAFE standards in the US have typically been “technology-forcing” as opposed to “technology-following” (Faiz et al., 1996, p. 1). Technology-forcing standards are standards set at a level which, although feasible, remains to be demonstrated in practice, and in the case of the US, have pushed technological advances forward. Although consumers may regard CAFE as a good policy (all else being equal, they spend less on fuel to drive the same distance, or they spend the same and drive more), CAFE has been seen as an inefficient CAC policy for two reasons: (a) it encourages excess investment in fuel efficiency and distorts the mix of large and small vehicles (Godek, 1997, p. 495); and (b) it is less cost effective than fuel taxes at reducing petrol consumption because by lowering fuel costs per km driven it increases (rather than reduces) vehicle use (Austin & Dinan, 2005; Kleit, 2004; West & Williams, 2005; Parry, 2007, in Fischer, Harrington, & Parry, 2007). Referring to standards in general, although not necessarily to CAFE standards in particular, Gruenspecht (1982) shows that more stringent standards for new vehicles prolongs the retention of old, high emission ones and as a result aggregate emissions may increase in the short run (p. 328). CAFE standards could have been tightened periodically since they were implemented. In practice, however, CAFE standards for cars have remained the same since 1990, and are, as of 2009, still 27.5 miles per gallon (11.7 km per litre). CAFE standards for vans have indeed been tightened, albeit in very small increments. They are 23.1 miles per gallon (9.8 km per litre) and 23.5 miles per gallon (10 km per litre) for the years 2009 and 2010 respectively.

Portney, Parry, Gruenspecht, and Harrington (2003, p. 204) show that higher real petrol prices together with new standards in the US had a substantial combined impact on fuel economy. They point out that new car and new van fuel economy rose by over 30 and 35 per cent respectively between 1978 and 1982. At that point petrol prices decreased and yet fuel economy continued to increase, very likely as a result of the CAFE program. However, the increase in the share of vans, subject to less stringent standards, meant that the combined new vehicle average fuel economy declined 6 per cent between 1987 and 2002. This increase in the share of vans is a direct impact from CAFE (Fischer et al., 2007, p. 2; Godek, 1997, p. 496). As of 2007, vans accounted for half of new passenger vehicle sales in the US (Fischer et al., 2007, p. 2).24

On 19 May 2009 President Barack Obama announced new tighter CAFE standards, covering models for the period 2012–2016 and reaching an average of 35.5 miles per gallon (15 km per litre) by 2016 (The White House Office of the Press Secretary, 2009). Portney et al. (2003, p. 216) warn that, although tightening CAFE standards may reduce fuel consumption and CO2 emissions, they will also reduce the cost of driving and thus create a stronger incentive to drive. The external cost of the “rebound effect” of additional driving, they argue, could be as large as the benefits from CAFE. Small and Van Dender (2007), on the other hand, point out that the rebound effect has recently halved in the US due to rising incomes and diminished significance of fuel costs.

Until the mid-80s, vehicle emission regulations in Europe were designed by the Economic Commission for Europe, to be later adopted and enforced by individual countries, and were typically much less stringent than US standards, as they had to be agreed by so many countries (Faiz et al., 1996, p. 8). The turning point in Europe came with the Consolidated Emissions Directive 91/441/EEC, which was adopted by the Ministers of the European Community in June 1991, made effective from 1 July 1992 for new models and from 31 December 1992 for all production (Faiz et al., 1996, p. 8). Since then emission standards have been adopted at a European level without the need for unanimous agreement from all countries.

Like in the US, EU standards increase their stringency with time. As of 2009, emissions of NOx, HC, CO and PM are regulated for cars, vans and lorries and for other vehicle types. While CAFE standards in the US regulate fuel economy and in doing so, CO2 emissions, the EU still lacked similar legislation until recently. Naively, Europe relied on voluntary changes and information campaigns to reduce CO2 emissions from the road transport sector.

In 1995, the EU heads of state and government set themselves the goal of reducing emissions of CO2 from new cars to 120 grams per kilometre by 2012. This is roughly equivalent to a fuel efficiency of 22 km per litre for diesel cars and 20 km per litre for petrol cars (EurActiv website). The strategy relied mainly on voluntary commitments from car manufacturers, improvements in consumer information and the option of individual countries implementing fiscal measures. In 1998, the European Automobile Manufacturers Association committed to reducing average emissions from new cars sold to 140 grams of CO2 per km by 2008 and, in 1999, the Japanese and Korean Automobile Manufacturers Associations also committed to reaching the same target by 2009 (European Commission, 2007). However, by the beginning of the year 2007, it became clear that while some progress had been made towards the target of 140 grams of CO2 per km by 2008/9, the Community objective of average emissions from the new car fleet of 120 grams of CO2 per km would not be met by 2012 in the absence of additional measures (European Commission, 2007). Later that year, the European Commission proposed mandatory reductions of emissions of CO2 to reach the objective of 130 grams of CO2 per km for the average new car fleet, through improvements in vehicle motor technology, and a further reduction of 10 grams of CO2 per km by other technological improvements (fuel-efficient tyres and air conditioning, traffic management and eco-driving26) and by an increased use of biofuels (European Commission, 2007).

In December 2008 the European Parliament voted to adopt the regulation (European Commission, 2008). The fleet average to be achieved by all cars registered in the EU is 130 grams per kilometre. Heavier cars are allowed higher emissions than lighter cars while preserving the overall fleet average and the requirements will be phased in, starting with 65 per cent of each manufacturer’s newly registered cars having to comply on average with the limit.

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23 Vans, or light trucks, in US terminology, include sport utility vehicles, minivans and pickups (Fischer et al., 2007, p.1). More specifically, they are defined as: ‘a 4-wheel vehicle which is designed for off-road operation (has 4-wheel drive or is more than 6000 pounds of gross vehicle weight and has physical features consistent with those of a truck); or which is designed to perform at least one of the living quarters; (3) transport property in an open bed; (4) permit greater cargo-capacity than that of a truck); or (5) permit greater cargo-carrying capacity than passenger-carrying volume; or (5) can be converted to an open bed vehicle by removal of rear seats to form a flat continuous floor with the use of simple tools’ (NHTSA website b).

24 This change in vehicle share may have had an impact on accident rates and their severity in the US. There are studies (Gayer, 2004; White, 2004 in Brozovic & Whitmoung Ando, 2009) showing that in multi-vehicle collisions drivers of vans, or light trucks, in US terminology, include sport utility vehicles, minivans and pickups (Fischer et al., 2007, p.1). More specifically, they are defined as: ‘a 4-wheel vehicle which is designed for off-road operation (has 4-wheel drive or is more than 6000 pounds of gross vehicle weight and has physical features consistent with those of a truck); or which is designed to perform at least one of the living quarters; (3) transport property in an open bed; (4) permit greater cargo-carrying capacity than passenger-carrying volume; or (5) can be converted to an open bed vehicle by removal of rear seats to form a flat continuous floor with the use of simple tools’ (NHTSA website b).

25 Interestingly, this did not result in higher fuel economy in the US than in Europe. For the year 2006, the average fuel economy for all cars in the US was 22.4 miles per gallon (9.52 km per litre) (US DOT, 2008b, Table 4.23). The average fuel economy for all cars in the UK, also for 2006, was 26.7 miles per gallon (11.3 km per litre) (UK DfT, 2008, Table 3.4). Note that the fuel economy listed in UK DfT (2008) is 32 miles per UK gallon, and 1 UK gallon = 1.2 US gallons.

26 Eco-driving is discussed in Part II of this volume.
increasing to 100 per cent from 2015 onwards (European Commission, 2008). In order to maintain the diversity of the car market the CO2 emission targets are defined according to their mass, so different vehicle sizes are subject to different targets. Also, manufacturers are allowed to form pools so that the average emissions of the pool as a whole do not exceed the target emissions for the pool (European Commission, 2007). As of August 2009, the European Commission is developing a new legislative proposal to reduce CO2 emissions from light commercial vehicles (vans and minibuses).

Clerides and Zachariadis (2008) explore the impacts of fuel standards (and also of fuel taxes) on new car fuel economy in Australia, Canada, the EU, Japan and the US, in the period 1975–2003. They find that fuel standards have contributed to fuel economy improvements in the US, Europe and Japan. They also find that new car fuel economy becomes less sensitive to fuel prices after the implementation of standards (p. 2671) and show that in Europe and Japan the impact of a fuel economy standard on new car fuel consumption has usually been more pronounced than that of an increase in fuel prices (p. 2671).

Although fuel economy standards have had impacts on fuel economy, Aldy et al. (2008, p. 496) argue that standards for the average fuel economy of vehicles in a manufacturer’s sales fleet do not encourage mitigation outside the automobile industry, or downstream sequestration activities, or reduction in road user mileage. Portney et al. (2003) conclude that fuel taxes (or carbon taxes) and tradable permits would make more efficient tools for reducing fuel consumption and GHG emissions. One way of making fuel standards slightly more efficient, they suggest, would be to provide some flexibility by making fuel economy credits transferable between car and van fleets and between different manufacturers (p. 216).

2.3. Other CAC policies in the road transport sector

Other CAC policies include restrictions on vehicle circulation, vehicle ownership, parking, and emissions in certain areas or on certain days. In this section we give some examples.

2.3.1. The low emission zone in London

The Low Emission Zone (LEZ) in London was implemented in February 2008. The zone covers most of Greater London, which can broadly be defined as the area inside the M25 motorway. Vehicles driving on the M25 without entering Greater London are not subject to the emission limits. The main objective of the scheme is to deter the most polluting diesel vehicles from driving inside Greater London.

The regulation includes diesel lorries, buses, coaches, large vans and minibuses. It also includes a number of other specialist vehicles, such as for example, breakdown and recovery vehicles, gritters and road sweepers. Cars, motorcycles and small vans are not included in the LEZ. The scheme operates 24 h a day, seven days a week, every day of the year (Transport for London website).

There are number of exemptions and discounts. Exempt vehicles include historic vehicles (built before 1973), military vehicles and vehicles which are mainly for off-road use, but which may be used on the road for limited purposes, such as tractors, cranes and farm machinery.

If a vehicle subject to the regulation neither meets the LEZ standards nor qualifies for an exemption or discount, it is subject to a daily charge. This is £100 for large vans and mini-buses and £200 for lorries.

The implementation is being phased. Although LEZ started in February 2008, there are different phases through to January 2012. Different vehicles will be affected over time as increasingly tougher emission standards apply.

2.3.2. Restrictions on vehicle circulation

Restrictions to circulation have been widely implemented in towns and cities throughout the world. It is very common to see pedestrianisation of streets, which are closed to traffic at all or some times of the day. It is also common to see streets where only public transport and taxis can circulate. This type of CAC policy is equitable, in the sense that it affects all drivers and does not differentiate by their willingness to pay for using the road (i.e. by their ability to pay).

Many (historic) towns in Europe have such types of areas. In general they do not harm the local economy, but rather, create a better environment for shoppers in the area. Parkhurst (2003) evaluates evidence from a pedestrianisation scheme in Oxford, UK, which was implemented in 1999 and is still in place today. He finds that as a result of the program there was a reduction of 17 per cent in the number of car trips to the centre but such reduction did not affect overall visitor numbers.

Another type of road space rationing has been to restrict certain licence plates from circulating. Such a policy can be found in cities like Athens, where cars with even (odd) number plates are allowed to drive on even (odd) days only. The problem with this type of policy is that even if the final number of vehicles using the road as a result of this type of policy was optimal from an economic point of view, there is no guarantee that the most efficient trips, with the highest marginal benefit (made by those drivers with the highest willingness to pay) would be the ones taking place (Verhoef, Nijkamp, & Rietveld, 1995, p. 141).

A similar scheme was implemented in Mexico City in 1989 to control air pollution from cars. The Hoy No Circula27 program bans most drivers from using their vehicles one weekday per week on the basis of the last digit of the vehicle’s license plate. The restrictions apply Monday to Friday, from 5:00 a.m. to 10:00 p.m. and affect most residential and commercial vehicles. Taxis, buses, and emergency vehicles (police cars, ambulances, and fire engines), as well as commercial vehicles operating with liquid propane gas, and commercial vehicles transporting perishable goods are all exempt. Davis (2008) measures the effect of the driving restrictions on air quality and finds no evidence of the restrictions having improved air quality. He does find evidence, however, of the restrictions having led to an increase in the total number of vehicles in circulation as well as a change in composition towards high-emissions vehicles. Eskeland and Feyzioglu (1997) find that many households bought an additional car to be able to drive on any day of the week and the amount of driving increased. The use of old cars also increased as well as weekend driving.

Another type of restriction on vehicle circulation is the Italian limited traffic zones, which have been implemented in a number of historic centres of medieval cities in Italy, such as for example, Rome. The Limited Traffic Zone (LTZ) in the historic centre of Rome is an area with restricted traffic. There are different permit types that allow different driving and parking arrangements. There are seven sectors inside the LTZ and different types of permits allow circulation with and without parking in different sectors at different times. Permits are restricted to a limited number of essential users. Those qualifying for permits include residents, some types of workers (such as doctors, nurses and teachers who work inside the LTZ), workers who have parking in the premises where they work, automobile mechanics, journalists, cars carrying children to school, school buses, goods vehicles, and disabled

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27 Hoy No Circula translates as ‘Today it does not circulate’.
people amongst others (ATAC website). Permits are valid for a year and there is a fee for each type of permit according to the sectors and the user, but this fee is fixed by the city council in advance, rather than determined at auction.

The scheme operates in the historic centre only, which has an area of 4.6 km² (1.8 mi²), representing only 0.36 per cent of the area of Rome, which is 1290 km² (500 mi²). The area is considerably smaller than the Charging Zone in London, which is discussed on Section 5. The hours of operation of the LTZ are Monday to Friday from 6:30 a.m. to 6:00 p.m., except public holidays, and on Saturdays from 2:00 p.m. to 6:00 p.m. (ATAC website).

The LTZ in Rome is a CAC policy for two reasons: (a) the permits are not tradable and (b) not everyone can qualify for a permit, which means that drivers who are willing to pay may not be allowed into the LTZ.

### 2.3.3. Restrictions on vehicle ownership

Another way of controlling vehicle use is through restrictions on vehicle ownership. The only example of a direct quantity control of this sort is the Vehicle Quota System (VQS), a policy implemented in Singapore in 1990 and still in place today. Prospective vehicle owners are required to purchase a Certificate of Entitlement (COE), which is a licence that lasts ten years, except for taxis, for which it lasts seven. The government sets a quota on COEs for different vehicle categories a year in advance, in April and May each year. Thus, the PQP payable for each month will be the average COE prices of March, April and May 2009. For example, the PQP in May 2009 is the average COE prices of March, April and May 2009. Therefore, the PQP payable for each month will be different.

Since there are bids this policy can be seen as an incentive based rather than a command-and-control one. However, the COEs are not tradeable, so the policy cannot be classified under the cap-and-trade umbrella. It falls into a grey area.

### 2.3.4. Parking restrictions

Parking restrictions can indirectly reduce traffic levels, and in doing so reduce most traffic externalities. Parking as an activity entails costs because parked vehicles use public space, which has an opportunity cost, as the land used for parking could be used for something else (Verhoef et al., 1995).

Verhoef et al. (1995) show that under strict assumptions (each individual drives the same distance, congestion is equally spread over the urban road network, the regulator has full control over all parking space available, every car is parked in a publicly managed parking space, and all cars are parked for the same length of time) quantitative restrictions on parking (in other words, a CAC policy) can achieve the optimal volume of traffic in terms of marginal congestion costs and marginal parking costs, just as they would with an IB policy. However, they point out that there would be no guarantee that the most valuable trips would be the ones that are realised. They also note that under quantity restrictions on parking, a market for parking permits could develop spontaneously, as drivers would try to secure parking for the most valuable trips.

Finally, reducing the number of parking spaces available, could result in more congestion, as more drivers would spend time looking for an available parking slot (Arnott & Inci, 2006; Arnott & Rowe, 1999).

### 2.4. Concluding remarks

CAC is not efficient from an economic point of view because even when the standard is set at the optimal level, it is not achieved at minimum cost. Despite that, CAC is the most widely used type of instrument in the road transport sector. Fuel and vehicle standards, which are used in virtually every country in the world, are the most prominent example. There are also some less spread regulations, which include parking restrictions, the Low Emission Zone in London, restrictions on vehicle circulation (pedestrianisation of streets, odd and even number plates allowed to circulate on alternate days, etc) and the restrictions on vehicle ownership in Singapore. The success of these measures varies, but in general they are all perceived as equitable by the motoring public, only because they do not involve payments, which tend to hit harder on the poorer.

### 3. Incentive based policies: quantity control

An IB policy based on quantity control is a system of marketable (transferable or tradeable) permits, credits, allowances or rights in which the regulator determines a cap or aggregate quantity of emissions, pollution or waste and leaves their allocation amongst polluters to be determined by the market (Baumol & Oates, 1988, p. 59). There are three steps involved in implementing and maintaining a system of this sort. First, the regulator sets the quantity or aggregate quota. Under a perfect information scenario, the regulator sets the quantity at the point at which marginal abatement costs are equal to marginal social damage. In reality, however, the regulator hardly has enough information to determine the efficient aggregate quantity of emissions, and sets a limit based on incomplete information. He is also often influenced by political factors...

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28 ATAC stands for Agenzia per i Trasporti Autoferrotranviari del Comune di Roma, which translates as Agency for Auto-rail-tram transport in the Municipality of Rome.

29 This result can probably be extended to other externalities.
such as lobby groups. Second, the regulator allocates the permits, which together do not exceed the cap. As already mentioned in Section 1, there are mainly two allocation methods: grandfathering and auctioning. These can be used in their pure form or they can be combined. With grandfathering, permits are allocated for free to each polluter, according to their past (historic) emissions. With auctioning, polluters pay for the permits they wish to buy, according to the price that emerges from the auction. An example of combination of both methods could be one in which 90 per cent of permits are distributed for free, according to past emissions, and 10 per cent are allocated through auctioning. Third, emitters trade their permits. Usually, those with lower marginal abatement costs sell permits and those with higher abatement costs buy permits. Permit trading amongst sources establishes the market-clearing price.

There are other allocation methods, although these have not been discussed very much in the literature. These include free distribution of permits, updating and benchmarking. Free distribution differs from grandfathering only insofar as the permits are not distributed on the basis of past (nor future) emissions or usage (Watters & Tight, 2007, p. 7), but on the basis of other criteria (e.g. equal per capita allocation). Updating also allocates permits for free, but on the basis of information (e.g. over level of activity) which is changed or updated over time (Watters & Tight, 2007, p. 6). According to this method, higher usage today will lead to more permits tomorrow. Benchmarking allocates permits for free to each polluter, according to each polluter’s emissions’ efficiency against a sector average.

It is important to emphasise that a cap-and-trade system is very different from the CAC approach discussed in Section 2. Under a CAC system, the regulator specifies a level of emissions or waste for each source and no trading is allowed afterwards.

Due to the novelty of the concept, penetration of cap-and-trade systems as a policy measure has been slow. In the 1970s a number of credit-based emission trading schemes, a very embryonic version of modern marketable permits, evolved in the US to deal with air quality.

In the 1980s the US EPA instituted two forms of trading: inter-refinery averaging during each quarter and banking for future use or sale (US EPA website a). The objective was to facilitate the phase-down of lead in petrol.

Inter-refinery averaging operated from November 1982 to December 1985, and allowed refineries to trade rights to add lead to petrol (Hahn & Hester, 1989, p. 381). If a refinery added less lead to petrol than was permitted, it could then sell lead rights to another refinery in an amount equal to the difference between the actual and allowed quantity (Hahn & Hester, 1989, p. 382). The other refinery was then allowed to add more lead than the standard allowed.

Banking for future use or sale was introduced in 1985 (Hahn & Hester, 1989, p. 382). Essentially refineries were allowed to bank credits for use until the end of 1987, in effect extending the life of lead credits to that date (US EPA website a).

Hansjürgens (2005, pp. 5–6) summarises the history of cap-and-trade systems in the US, following the US Clean Air Act of 1990. These included the Regional Clean Air Incentives Market in Southern California, trading sulphur oxides and nitrogen oxides (1994), the Acid Rain Program, trading sulphur dioxide (1995), the Northeast NOx Budget Trading Program, a multi-jurisdictional partnership between the federal and nine state governments, trading nitrogen oxides (1999), later expanded to include nineteen states and the District of Columbia (2004).

The biggest scheme to have ever been implemented, however, is the EU Emission Trading Scheme (EU ETS). Before the EU ETS, some governments, such as the UK in 2002 and the Australian state of New South Wales in 2003, had already implemented carbon trading (Heburn, 2007, p. 377). The EU ETS is a permit-trading system implemented at European level to help achieve the Kyoto target of an 8 per cent emission reduction of GHG in the EU by 2012. The EU ETS only covers emissions of CO2. For the first two trading periods, 2005–2007 and 2008–2012, aggregated emission caps were imposed on the most energy-intensive sectors (cement, glass, ceramics, paper, steel and iron, and power) (European Commission, 2003, Annex 1), which were responsible for more than half of EU CO2 emissions from 2005 to 2007 (European Commission, 2005, p. 5). Permit allocation is made through grandfathering and combined with a very small percentage of auctioning (5 per cent and 10 per cent at most, for the first and second periods respectively). Allocation decisions are made at EU level, based on each country’s National Allocation Plan (NAP) (European Commission, 2000, p. 18). NAPs determine the total quantity of CO2 emissions that Member States grant to their companies, which can then be sold or bought by the companies themselves. This means each Member State must ex-ante decide how many allowances to allocate in total for a trading period and how many each plant covered by the Emissions Trading Scheme will receive.

Aviation will be included in the EU ETS from 2012 and as of 2009 there are already some discussions about the prospects of including road transport as well (UK DfT, 2007b). However, a cap-and-trade system to deal with the external costs briefly described in Section 1 has not been implemented for road transport anywhere yet.

There have been proposals and studies of marketable permits in road transport regarding different transport externalities (Verhoef, Nijkamp, & Rietveld, 1997), including air pollution (Raux, 2004) and CO2 (Albrecht, 2001; Raux & Marlot, 2005; Wadud, Nolan, & Graham, 2008; Watters, Tight, & Bristow, 2006). If a cap-and-trade system were to be implemented in the road transport sector, there would be a number of issues to consider. The Albrecht (2001), Grubb and Neuhoff (2006) and UK DfT (2007b) highlight a number of points that need to be assessed and decided upon the introduction of a cap-and-trade system for CO2 emissions (in the road transport sector). These are:

- Allocation of permits:
  - How to allocate (auctioning vs. grandfathering)
  - Whom to allocate (individual motorists, vehicle manufacturers, fuel producers)
- Area of applicability of the policy (regional, national, international)
- Duration of the permit (time scale)
- Decision over whether the emission trading scheme is a stand-alone scheme, or part of a broader emission trading scheme across sectors
- Interaction of the road transport emission trading scheme with other schemes, such as the EU ETS, and other policies, such as fuel taxation
- Credibility of the scheme continuation
- Costs and benefits for firms, road users and government
- Long-term marginal abatement costs (which in transport depend on intermodal shifts, penetration of highly fuel efficient vehicles, and behavioural change, amongst others)

Ellerman, Jacoby, and Zimmerman (2006) point out that the main challenges in the introduction of emission trading schemes in the transport sector come from the fact that emission sources are small, dispersed and mobile, hence difficult and costly to monitor, and from the fact that the transport sector is already highly regulated. This means that cap-and-trade systems need to be carefully coordinated with existing policies if they are to achieve the desired effects at the minimum cost, without imposing excessive burdens on the targeted agents.
A baseline-and-credit system could also be envisaged instead of a cap-and-trade system (Watters & Tight 2007, p. 8). In a baseline-and-credit system, a baseline consumption of the resource is established for each user. A period of time is then established for compliance with the baseline. If actual emissions are lower than the baseline, users are given credits. If they are higher, they must purchase credits. The problem is that the emission limit is user-specific rather than universal. Hence, total emissions could increase if new users entered the market (i.e. started using road transport). On the other hand, under a cap-and-trade system a limit of emissions for the whole sector (or for the whole country) is established. Permits are allocated and participants are allowed to trade these permits in a secondary market.

3.1. Grandfathering versus auctioning

The way in which permits are allocated to the relevant parties is important. The two ends of the spectrum are grandfathering and auctioning. As we explained above, grandfathering is an allocation method based on historic emissions or indicators: the number of permits allocated to an agent (a firm or a group of firms, a consumer or a group of consumers, etc) has some degree of proportionality to its historic emissions. Grandfathering is the method most commonly used in practice, due to its greater political acceptability, when compared to auctioning. Auctioning involves polluters bidding for emission permits, and is, from an economic point of view, the most efficient as well as the most equitable allocation system, as we will discuss below.

Any system of emission permits raises the prices of the goods and services affected directly or indirectly by the scheme. Firms needing emission permits will typically pass on their costs to consumers. Consumers needing permits themselves will also end up paying a higher final price for the good or service in question. However, the market price of permits is independent from the allocation method (grandfathering versus auctioning).

Grandfathering however allows companies to make windfall profits. This raises equity as well as implementation issues. To the extent that shareholders of the profiting firms are richer than average, this amounts to a transfer to the relatively rich. Equity considerations also concern the treatment of new entrants versus incumbents. Grandfathering discourages entry if new entrants, unlike the incumbents, have to buy all the permits they need and are not allowed a share of permits for free (Watters & Tight, 2007). Implementation problems under grandfathering relate to the difficulty in establishing historic emission levels, especially if the trading parties are individuals (Watters & Tight, p. 6), such as for example, drivers.

Auctioning creates revenues for the government, which can be redistributed in the form of, for example, tax cuts (Cramton & Kerr, 1999, p. 256). Hence, the use of auctioning has the potential to correct distributional impacts. Revenues from auctioning permits to fuel producers could also be channelled into investments in transport infrastructure, public transport, R&D of cleaner technologies, etc. However, it has been argued that individuals or entities with greater financial power are in a better position to buy as many permits as they require during the auctioning phase, thus threatening equity (although upper regulatory limits could be imposed to prevent this effect) (Watters & Tight, p. 6). Finally, auctioning provides more transparency and clearer long-term price signals compared to grandfathering. Periodic auctions could improve price stability and the management of uncertainty (Hepburn et al., 2006).

An important element to consider when evaluating emission trading proposals is the impact of the scheme on the competitiveness of the industry it regulates. In the long term, the chosen allocation method does not normally affect competitiveness. Grandfathering acts as a temporary subsidy to participants in the scheme. It does not normally affect marginal costs, and marginal costs determine competitiveness. Furthermore, evaluating different proposals on the basis of competitiveness is only possible in industries which are exposed to competition from countries outside the scheme (which might not apply to road transport, depending on whom the permits are allocated to), and which face significant cost increases. The final point regarding competitiveness is that auctioning has the advantage of making some border-tax adjustments feasible. These can only be based on real cost incurred as a result of the regulation under World Trade Organization rules. Auctioning helps align total costs (inclusive of opportunity costs) with real costs (Hepburn et al., 2006).

Auctioning is to be preferred to grandfathering also from a dynamic perspective in relation to the incentives it creates. Theoretically, auctioning enhances incentives for innovation. This is due to the fact that innovation ultimately leads to a decrease in demand for permits, and consequently a decrease in prices of permits. This reduces the scarcity rents which belong to the industry under a grandfathering scheme, but do not under auctioning (Cramton & Kerr, 2002). Hence, only the beneficial price-reduction effect is internalised by the industry under auctioning.

In addition, grandfathering may lead to perverse dynamic incentives in emissions reduction. This occurs when future rights are a function of current emissions (which always occurs under updating). In this case, regulation induces a higher amount of current emissions than optimal. Also, when allocation of permits to new plants is subject to more restrictive criteria than allocation of permits to the incumbent, incentives to invest in new plants will be reduced, and incentives for plant life-time extension will be enhanced in a distortionary manner (Hepburn et al., 2006).

The case for free allocation of permits is mainly due to the fact that it minimises problems of social and political acceptability (Raux & Marlot, 2005). Also, in some instances, free allocation may be administratively less costly. If an emission trading scheme were to be implemented amongst motorists, free allocation would allow at least a certain amount of fuel to be consumed at no additional cost (Raux & Marlot, 2005). Even taking into account the opportunity cost for a motorist of not selling the permit, this opportunity cost would be the lost revenue but would not be an “additional” cost on top of what he paid for fuel previously. Auctioning permits to such a large number of individuals would not be practically feasible. Grandfathering has already been shown to involve lobbying activities to get the preferred allocation (Hepburn, 2007), and this would cause a loss of resources. Also, a grandfathering allocation of permits among drivers would

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30 Obvious examples include an electricity company charging higher utility bills or a car manufacturer raising the sale price of the cars he produces. On the other hand, in the US, the power generation market in many states remains under cost-of-service regulation. This prevents the electricity utilities from passing on the opportunity cost of free permits to consumers, reduces the price impacts of allocation policies and causes suboptimal electricity conservation (Beamon, Leckey, & Martin, 2001; Burtraw, Palmer, Bhavevirkar, & Paul, 2001).

31 For example, if petrol consumption were included in an emission trading scheme drivers would pay for petrol and for the permit to consume that petrol. Drivers would perceive no change only in the case that permits replaced fuel duties and the final average amounts paid were roughly similar.

32 This is so unless grandfathering allows an unprofitable firm to survive.

33 This is true unless permits are auctioned many years in advance, in which case grandfathering and auctioning provide the same incentives (Kerr, 2000, in Cramton & Kerr, 2002). Also, empirically, there is little difference between R&D efficiency gains under auctioning and under grandfathering (Fisher et al., 2003).
involve administrative costs to collect the historic data necessary for the allocation. These administrative costs would be even higher under updating.

Finally, Parry (2003) analyses both carbon taxes and tradable carbon permits in a setting where there are pre-existing tax distortions in the labour market (which is very likely to be the case in virtually any economy) and finds that although both policies can cause substantial efficiency losses in the labour market (relative to abatement costs), these can be offset, or even more than offset, if revenues from carbon taxes or auctioned permits are used to reduce distortionary taxes. Thus, he concludes that there is a case for using carbon taxes or auctioned permits rather than grandfathered permits.

### 3.2. Target group

The question of whom the permits should be allocated to - fuel producers, vehicle manufacturers, or individual motorists - is not easy to answer, and we shall evaluate each option separately. Trading among fuel producers or car manufacturers is called “upstream” trading, while trading among individual motorists is called “downstream” trading.

If the target group were fuel producers, they would be required to hold permits to cover the total amount of emissions resulting from the fuel they sell (UK DfT, 2007b). Oil refineries in the EU are currently included in the EU ETS on the basis of their direct combustion emissions, i.e. CO₂ emitted during the fuel production process. They are therefore already familiar with the EU ETS. In the UK, regulating twenty fuel producers would cover 99 per cent of fuel sales (UK DfT, 2007b). This means that the administrative costs of this option would be relatively low. However, fuel producers have few tools at their disposal for reducing emissions, mainly efficiency increases in the production process and the alteration of carbon composition of petrol. The main effect of emission trading in the short run will be an increase in fuel prices, as refiners are likely to pass on at least part of the cost of emission trading permits to consumers. The likely short-term effect of the policy is a non-equitable regressive effect, similar to that of an additional tax on fuel (Watters & Tight, 2007). A tax would have the advantage of being less volatile, more easily adjustable, and requiring less monitoring. The exact burden share between refineries and final consumers of fuel would depend on supply constraints and the level of competition, amongst other factors (Millard-Ball, 2009).

By and large, the main short-term impact on emissions of upstream trading would be a reduction in fuel consumption due to an increase in fuel prices (UK DfT, 2007b). Short-term responses by motorists would be a reduction in the amount of travel and adoption of a more responsible travel behaviour (such as for example, eco-driving, using public transport, walking or cycling, which are discussed in Part II of this volume). In the long run, demand may lean more towards more fuel-efficient vehicles (UK DfT, 2007b). Demand for vehicle travel is relatively inelastic, especially in the short term. Consequently, if the trading system was part of a multi-sector system (like the EU ETS), the transport sector would tend to purchase credits rather than reduce emissions itself. Hence, upstream trading would miss out on the potential for numerous low-cost transport abatement measures (Millard-Ball, 2008).

Another weakness of an upstream emission trading scheme in the transport sector is that it would undermine the effectiveness of alternative measures to reduce emissions below the cap level. In fact, investment in public transport improvement and demand management policies, which lead directly to a reduction in emissions, indirectly produce emission increases elsewhere in the transport sector or in other sectors. This occurs through the price channel. As emissions are reduced through some measure that is not covered by the scheme, the permit price falls, and an increase in emissions will be observed elsewhere in the economy. The implication is that the benefits from programs that could further reduce emissions below the cap level are economic (reducing compliance costs elsewhere) rather than environmental (reducing aggregate emissions) (Millard-Ball, 2009).

Another emission trading scheme option in the road transport sector would target vehicle manufacturers. Under such an arrangement, manufacturers would be required to purchase permits for emissions imputed to their vehicles (UK DfT, 2007b; Millard-Ball, 2008). It would be difficult to retroactively cover all vehicles in circulation but it would certainly be possible to cover all new vehicles sold. The disadvantage of this approach is that it could potentially boost the second-hand market, increase second-hand car prices, prolong the life of fuel inefficient vehicles, and hit the new vehicle market.

If car manufacturers were to trade emission permits, these could be required to cover all expected future life-time emissions of the fleet (total carbon approach), or emissions in grams of CO₂ per km. Since there are few car manufacturers, the “administrative costs would be low” (German, 2007, p. 95).

On the other hand, if the cap were based on expected life-time emissions of the fleet and the estimates were not precise, the desired carbon reduction objective could be missed. Furthermore, if transport emissions were to be covered in the EU ETS, the adoption of future emissions as a standard would risk undermining its integrity because the EU ETS is based on current emissions. A solution could be the setting up of a separate regulation for car manufacturers, so that they would only be allowed to buy but not sell emission permits in the EU ETS (UK DfT, 2007b).

Tradeable certificates would increase the involvement of the car industry in the development of fuel-efficient vehicles (Millard-Ball, 2008) and create incentives to innovate. They would provide a direct financial reward to the reduction of emissions from vehicles (both in terms of type of car and type of fuel the car uses) and give car manufacturers that have invested in fuel-efficient cars a competitive advantage (Millard-Ball, 2008). The total carbon approach would create incentives to reduce emissions at all possible levels (German, 2007, p. 97). Beside the development of new low-emission technologies, and the reduction in average vehicle size, car manufacturers could respond with an increase in marketing efforts to sell low emission vehicles (Watters & Tight, 2007). At the same time, if the permit price were high, highly efficient cars would be made relatively cheaper and this would stimulate their market penetration.

Not only would regulation at the manufacturers’ level have low administration costs from an administrative point of view, as already pointed out above, but it would also have the advantage of avoiding politically-sensitive increases in fuel prices. However, it has been argued that fuel-economy standards and feebates may be equally effective in targeting the optimal fuel-efficiency of the vehicle fleet (Millard-Ball, 2008). One of the major weaknesses of a cap-and-trade scheme designed to target car manufacturers is that it has a negligible influence on vehicle usage. This happens because the number of permits and the consequent market price are determined prior to the car-use decision made by consumers.

Double-counting is yet another potential problem if upstream producers in another sector (e.g. fuel producers) are also included in the emission trading scheme (Millard-Ball, 2008).

An option suggested by Gallagher et al. (2007) would be to establish a market of permits that includes both vehicles manufacturers and fuel producers. This would increase the flexibility of...
the system, but has greater implementation challenges and shares most of the downsides of an upstream trading scheme discussed above.

**Downstream trading** refers to the inclusion of individual motorists into the cap-and-trade scheme. The inclusion could occur under different formats:

- Motorists could be required to use permits against the purchase of fuel. Permits could be allocated for free by the regulator or auctioned. Individuals could then trade their permits. Bank operators or petrol stations could act as intermediaries in the purchase and sale of permits (Raux & Marlot, 2005). This would make this option similar to allocation to fuel producers (UK DfT, 2007b). A problem with this option could be cross-border refuelling, unless and entire region adopted such a system.
- Motorists could be required to use permits against the purchase of a vehicle. The number of automobiles would be capped and vehicle ownership permits would be traded (Millard-Ball, 2008).
- Motorists could be required to use and trade parking permits or vehicle kilometres travelled allowances, or tradable driving day rights.
- Motorists could be endowed with personal carbon quotas or domestic tradeable quotas to cover their entire carbon footprints, including personal transport emissions (Millard-Ball, 2008).

Although downstream trading at individual motorist level could be extremely costly to implement and monitor it would also have a number of advantages. First of all, the maximum efficiency of an economic incentive instrument is achieved when it operates at the most decentralised level (Raux & Marlot, 2005). Second, a large number of consumers would yield a large and liquid market for allowances, making the system attractive to intermediaries in the purchase and sale of permits. Third, a free allocation of permits to motorists, or fuel producers or car manufacturers becoming net buyers of permits, would mean a larger number of players, which would facilitate finding a trading partner. With regards to the length of the validity of the permit, there are potential trade-offs. The longer is the permit duration, the lower is its tradability, and the higher are transaction costs. However, the longer is the validity of a permit, the greater is the uncertainty about future prices. Uncertainty, in turn, undermines the efficiency of the trading system.

If the road transport sector were included in the EU ETS drivers would probably become net buyers of permits, at least in the short-run. This would be likely to happen unless permit prices were so high that alternative technologies became attractive, or unless additional requirements were imposed. It should be stressed that motorists, fuel producers or car manufacturers becoming net buyers is a desirable characteristic of economic instruments and not an imperfection. Reductions would be made where the marginal abatement costs are lower, which would be in other sectors of the economy, rather than the road transport sector. If the regulator’s objective were to guarantee a certain level of emission reduction from road transport, then a separate scheme (perhaps linked in some way) to the EU ETS would need to be designed.

### 3.3. Other considerations

Amongst other important considerations in the design of trading schemes there are: the definition of the area of applicability of a trading scheme, the possibility of integration of a (road) transport trading scheme within other broader trading schemes, and the choice of time validity of the permit. For example, it may be advantageous to have a wider area covered by a trading scheme due to the added liquidity in the permits market. In fact, a wider area would mean a larger number of players, which would facilitate finding a trading partner. With regards to the length of the validity of the permit, there are potential trade-offs. The shorter is the permit duration, the lower is its tradability, and the higher are transaction costs. However, the longer is the validity of a permit, the greater is the uncertainty about future prices. Uncertainty, in turn, undermines the efficiency of the trading system.

If the road transport sector were included in the EU ETS drivers would probably become net buyers of permits, at least in the short-run. This would be likely to happen unless permit prices were so high that alternative technologies became attractive, or unless additional requirements were imposed. It should be stressed that motorists, fuel producers or car manufacturers becoming net buyers is a desirable characteristic of economic instruments and not an imperfection. Reductions would be made where the marginal abatement costs are lower, which would be in other sectors of the economy, rather than the road transport sector. If the regulator’s objective were to guarantee a certain level of emission reduction from road transport, then a separate scheme (perhaps linked in some way) to the EU ETS would need to be designed.

### 3.4. Concluding remarks

A cap-and-trade system is one in which the regulator determines a cap on emissions, pollution or waste and leaves their allocation amongst polluters to be determined by the market. Polluters can trade permits: those with lower marginal abatement costs sell permits and those with higher abatement costs buy permits. There are mainly two initial allocation methods: grandfathering and auctioning. With grandfathering, permits are allocated to each pollutant for free, according to their past (historic) emissions. With auctioning, polluters pay for the permits they wish to buy, according to the price that emerges from the auction. Although there have been and there continue to be proposals to introduce cap-and-trade to deal with different externalities (lately with CO2 emissions), no such system has been implemented in the road transport sector anywhere in the world yet.

Implementing tradeable permits in road transport would entail a number of decisions, including how to allocate (auctioning vs.

35 An upstream system for fuel refiners with a number of permits proportional to the carbon content of refined fuel would have similar advantages.

36 However, Parry (2007) argues that the benefits from auctioned (rather than grandfathered) permits (and also taxes) to reduce road transport emissions are larger than the costs when account is taken of (a) their impact on reducing other road transport externalities such as accidents, congestion, local air pollution and oil dependence and (b) interactions with the broader fiscal system such as the use of revenues to reduce distortio
grandfathering), whom to allocate (individual motorists, vehicle manufacturers, fuel producers), how many permits to allocate (actual cap on emissions), area of applicability of the policy (regional, national, international) and duration of the permits, to name just a few.

Although fuel and vehicle taxation is already in place, and adjusting it would be administratively less costly than introducing an entirely new system, such as cap-and-trade, there seems to be a trend around the world to favour or welcome the novelty of the concept of tradeable permits, and just because of that reason, cap-and-trade may have a role to play in the road transport sector in the short, medium and long-run.

4. Incentive based policies: fiscal policy instruments

Governments use taxes for two reasons: either to generate revenues and to redistribute part of the revenues to achieve a “fairer” distribution of resources in society and provide goods or services that the market by itself would not provide (such as for example, defence) or to correct market failures. The first type of taxes are distortive taxes, the second type are corrective taxes. What the two types of taxation have in common is that they act as incentives (or disincentives) on behaviour, by increasing the marginal cost of certain activities. What they differ on is that the first type of taxes is introduced in an otherwise efficient economy. Their introduction generates a deadweight loss, by ‘inserting a wedge between marginal cost and price’ (Cramton & Kerr, 2002, p. 339). The second type of taxes, on the other hand, is introduced in an otherwise inefficient economy, in order to correct the distortions and change behaviour so as to restore efficiency.

The road transport externalities discussed in Section 1 are examples of the type of inefficiency that can arise in an economy. As it was already advanced in that section, one possible instrument that can be used to correct the distortion is a corrective or Pigouvian tax, charge or fee. Such a corrective tax can help align marginal private costs with marginal social costs.

In practice, any economy will have inefficiencies, and any government will try to provide at least some of the services which are not typically provided by the market and pursue some equity goals. As a consequence, both distortive and corrective taxes will typically be present. Corrective taxes, besides tackling the market failure, generate revenues that the government can use to finance expenditures or (at least partially) to substitute for distortionary taxes.37

Taxes have been widely implemented in the road transport sector around the world, both in developed and developing countries. These have been introduced as either IB instruments to influence travellers' behaviour in such a way as to induce a given response that reduces or corrects market failures, or as revenue raising instruments. For example, the taxation of cars can be structured so as to limit ownership, promote adoption of cleaner vehicles, change age and class car structure within a market, and improve car maintenance (Hayashi, Kato, & Teodoró, 2001, p. 124), at the same time as generating revenues. The effectiveness of taxes as corrective instruments depends on the strength of the link between the externality they are targeting and the tax itself. If this link is weak, then drivers, polluters, or road users may respond in an inefficient way (Crawford & Smith, 1995, p. 40).

Parry and Small (2005) attempt to compute the optimal petrol tax in the US. If this were to internalise congestion, accidents and local and global air pollution. Parry (2008) conducts a similar exercise for diesel tax in the US, accounting for accidents, road damage, noise, energy security, and local and global pollution externalities. Although Parry and Small (2005, p. 1277) recognise that global warming is ‘the only component for which the fuel tax is (approximately) the right instrument’, they still estimate what the fuel tax should be if it were to internalise most road transport externalities, and assuming no other tax or charge were implemented simultaneously.

There are many types of taxes and charges in road transport. Several factors determine the ideal policy instrument to be selected (Timilsina & Dulal, 2008). These are the type of externality that the government wants to target, e.g. emissions versus congestion, global emissions versus local emissions; the flexibility the government has to meet the targets; the nature of information asymmetries; the specific characteristics of the location; and ultimately the (administrative and monitoring) costs and benefits associated with each instrument. For example, fuel consumption is closely related to the global warming impact of a vehicle. However, PM emissions and noise also strongly depend on location, weather conditions and congestion levels prevailing at the time the vehicle is driven. Hence, even though fuel taxes may be good instruments to target global warming, they may be less effective for targeting local air pollution (Crawford & Smith, 1995, p. 34). Proost, Delhaeye, Nijs, and Van Regemorter (2009) argue that (high) fuel taxes are imperfect instruments to internalise all transport externalities simultaneously.

Another important consideration concerns the distinction between first best and second best policies. While in a first best context one instrument can suffice to reach the optimal allocation, in a second best context,38 many instruments may need to be combined to obtain the best feasible allocation. This may involve the taxation of activities that are complementary to the taxation of fuel and vehicles, when these cannot account for the full social costs, and subsidisation of substitute activities. Hence, additional parking charges or subsidisation of public transport may be advisable (Crawford & Smith, 1995, p. 44). Second best situations are likely to be the rule rather than the exception in a context of transport externalities and corrective taxes (Verhoef, 2000, p. 308).

4.1. Taxes on purchase and ownership of a vehicle

Many countries impose taxes on the purchase of vehicles, and/or a periodic (e.g. annual) licence fee on the ownership of a vehicle. Purchase taxes are direct tools to address CO₂ emissions generated during the manufacturing and disposal of vehicles. However, CO₂ emissions generated during purchase and disposal account for only a small share of life-time vehicle emissions, which purchase and ownership taxes do not directly tackle (Hayashi et al., 2001) in their basic un differentiated form. In fact, as such, the primary impact of a purchase tax is a reduction in the number of vehicles in circulation (Chia Tsui, & Whalley, 2001), and a longer tenure. A longer tenure implies increase in the usage rate of vehicles (De Jong, 1990, in Barter, 2005). This is a perverse incentive that induces over-consumption of older and more polluting models. In addition, Pritchard and DeBoer (1995, in Timilsina & Dulal, 2008) point out that if higher taxes on new vehicles are not matched by higher taxes on second-hand ones the policy may fail to bring about a reduction in car ownership. Higher taxes on new vehicles only can otherwise result in a perverse effect, shifting purchase behaviour towards older and more polluting vehicles. In general, purchase and ownership taxes also increase business costs of a country and harm its competitiveness (Barter, 2005). However, more sophisticated

37 This is the double dividend hypothesis, which is discussed in Section 4.5.

38 Second best refers to the optimal policy when the true optimum (the first best) is unavailable due to constraints on policy choice.
forms of purchase and ownership taxes can be envisaged that better target vehicle usage emissions and internalise (at least partly) environmental externalities. These are differentiated taxes, on the basis of fuel economy characteristics of vehicles, or other vehicle characteristics.

In the case of purchase taxes, these can be differentiated on the basis of fuel consumption per mile, type of fuel supported, horsepower, size of engine, vehicle weight, and retail price. Jonhstone and Karousakis (1999, p. 101) identify model year, mileage and the presence of fuel injection as the most significant factors that contribute to emissions of HC, NOx and CO. In the case of ownership taxes, relevant characteristics also include vehicle age. Such a differentiation provides price signals to consumers that act as incentives or disincentives on their purchase decisions (Evans, 2008) and can, for example, encourage a shift towards smaller cars (Acutt & Dodgson, 1997, p. 23). However, there is often a trade-off between the precision of the characteristics as a basis for emissions correction, and the easiness in adoption of such a basis. For example, engine size is straight-forward to tax. However, its correlation with emissions of conventional pollutants might be weak compared to a tax on emissions-per-km. Emissions-per-km are, however, less easily measured, since they depend on engine size, vehicle age, fuel cleanliness and pollution control equipment, as well as driving style (Fullerton & West, 2003 p. 5; Fullerton & Gan, 2005, p. 300).

Differentiated taxes are potentially effective tools and administratively relatively simple, since a tax system is already in place. Moderate costs can arise from modifications to the tax collection system that may be required if the tax structure is changed (Acutt & Dodgson, 1997). Although purchase and ownership taxes are low relative to the total cost of purchase, ownership and usage of a car, they can have a significant impact by influencing choice of car ownership at the margin (Knight, Vanden Branden, Potter, Enoch, & Ubbels, 2000). Differentiated vehicle tax systems have an impact both on the supply and on the demand of vehicles. They incentivise manufacturers to improve the fuel efficiency of vehicles, so as to reduce consumers’ tax burden and they provide incentives for consumers to privilege fuel economy in their purchase choice (Evans, 2008). However, a danger exists that imposing a tax on a certain characteristic will elicit a manufacturer’s response in vehicle’s design that eludes the tax and breaks down the observed relationship between characteristic and emissions. As a consequence, these policies are more effective in countries that do not have significant manufacturing capacity (Jonhstone & Karousakis, 1999).

An ideal differentiated tax to internalise emissions would be set equal to the present value of the social costs of emissions, the vehicle’s emissions per mile or km, and the life-time mileage choice (Jonhstone & Karousakis, 1999). However, this is difficult to implement in practice for several reasons. Firstly, assessing the social costs of emissions is a complex and costly task. Secondly, if the tax is calculated at the time of purchase, the life-time mileage choice is an ex-post decision, and the fixed tax remains an initial sunk cost. As a consequence, the tax does not present the driver with the correct subsequent incentives for mileage choice, pollution control equipment maintenance, and length of vehicle lifetime. For this reason, it might be advisable to combine a purchase tax with a fuel tax and/or a congestion charge, measures which act directly on usage decisions (Jonhstone & Karousakis, 1999; Sterner, 2003, in Evans, 2008).

When taxes are combined, one classic problem if the combination is not carefully designed is that of double charging. For example, a tax based on the size of engine is superfluous when a tax on fuel is already in place, since larger engines are associated with greater fuel consumption (Fullerton & West, 2003). If the vehicle tax was determined on a periodic basis, then periodic odometer readings could lead to efficient internalisation of usage choices, since intensity of car use could be taxed. However, individuals would have incentives to roll back their odometers (Fullerton & West, 2003, p. 53). This, however, is becoming increasingly harder, as electronic odometers are being introduced.

Whether the tax is levied on purchase or on an annual basis has consequences on incentives. If purchase of both used and new vehicles is taxed, then, as previously argued, the turnover of the vehicle stock is reduced, as the price of new vehicles increases. This slows down emissions reduction, as new vehicles are typically cleaner than older ones (Jonhstone & Karousakis, 1999). A tax proportional to age of the vehicle bought (or a subsidy on new vehicles) could counteract these perverse incentives and play a significant role in reducing emissions, by discouraging the purchase of older vehicles (Fullerton & West, 2003). A periodic tax makes consumers bear the costs of continued operations of a polluting vehicle, and can potentially foster maintenance activities that reduce emissions. In addition, a periodic tax allows the adjustment of the tax on the basis of time-variant characteristics that are linked to pollution, e.g. age. Age is highly correlated with emissions, and is easier to observe and tax than emissions per km. However, Fullerton and Gan (2005, pp. 303–304) find that a tax on age is less effective in reducing emissions, compared to a tax on fuel or on emissions-per-km.

The distributional effect of differentiated tax policies on purchase and ownership depends on the characteristics taxed, and on the characteristics of the households in the location where the policy is implemented. For example, Fullerton and West (2003) show that in California, a tax on vehicle size would be progressive, as household wealth is positively correlated with vehicle size. However, West (2004) shows that overall in the US a tax on size would be regressive, since higher income households’ preferences for size are non-homogeneous across vehicle age: ‘households with higher incomes that own 1980s-vintage vehicles prefer smaller engine sizes, while those with 1990s vehicles prefer larger engine sizes’ (p. 738). Thus, expenditure on the size-tax would account for a smaller share of household income in high income compared to low income households. A subsidy to encourage the purchase of new vehicles would also be regressive, since higher income households prefer newer vehicles (West, 2004, p. 738). Similarly, a tax based on vehicle age would also typically have a higher impact on lower income families (Fullerton & West, 2003).

4.1.1. Subsidies to efficient vehicles and feebates

Subsidies for the purchase of fuel efficient or alternative fuel vehicles are an important economic instrument to encourage the purchase of low (or zero) emission vehicles. Consumers may choose not to buy a cleaner (but more expensive) vehicle, especially if the annual distances driven provide no opportunity to capture the benefits from fuel savings over the life of the vehicle (McKinsey & Company, 2009, p. 19). In these cases, subsidies may be effective in tilting consumers’ preferences. However, they may also lead to an increase in the number of car ownership, offsetting the subsidies’ beneficial effect. The choice of subsidy policies needs to be carefully matched to technological information and information about the specific local characteristics. For example, if electric cars were to be subsidised, the way in which the electricity they consume is produced (e.g. coal vs. nuclear power stations) would need to be taken into account. Similarly, from a fuel consumption and CO2 emissions point of view, hybrid cars are a good choice in towns and cities, where speeds are low and congestion is a problem. On trunk roads, on the other hand, the advantage of hybrids over conventional petrol cars is much smaller.

An alternative to subsidies would be deductions from taxable income to make up for part or all of the vehicle cost, or tax credits
for the purchase of cleaner vehicles. In designing tax credit schemes, care must be taken not to inadvertently create perverse incentives. In the US, for instance, the CAFE credits given to hybrid vehicles are discounted so as to reduce the extent by which manufacturers can cross-subsidise fuel inefficient vehicles by producing some extremely efficient ones (US NHTSA website b). Another crucial observation is that the distributional impact of a subsidy is not the same as the distributional impact of a tax deduction or tax credit. In fact, the primary and (perhaps the only) beneficiaries of tax deductions and tax credits are those individuals who earn taxable income, i.e. the relatively rich in the population. In addition, if the chosen format is deductions from taxable income, the benefits will be higher the higher the income when taxation systems are progressive, hence the system would be regressive, and for this reason, not desirable from an equity point of view.

A particularly attractive form of subsidisation is that of feebates, a term used to describe a scheme which combines fees and rebates. This could take the form of for example, purchase taxes on fuel inefficient vehicles, and rebates on the price of fuel efficient ones. Feebates reduce the burden on the government's finances compared to simple subsidies. By financing the subsidisation of fuel efficient vehicles through taxation of less efficient ones, feebates have the potential to be budget neutral instruments (BenDor & Ford, 2006; Koopman, 1995).

Feesbates can shift both demand and supply of vehicles. In fact, not only do they alter the purchase price of a vehicle, but they also encourage manufacturers to develop and adopt low emission technologies on the vehicles offered, in order to best exploit the feebate system in place, and get a head start over competitors. The supply response to the feebate incentive occurs also when feebates are offered directly to consumers. Consumers' demand, however, does not shift when feebates target manufacturers on the basis of their sales mix within a model year. This suggests that consumers should be the privileged target for the application of such schemes.

Feesbates can provide an incentive to meet vehicle standards in a more efficient way than through the internal pricing strategy each vehicle manufacturer would undertake under normal competitive conditions. They can also work alongside congestion charges and fuel taxes as additional price signals (Greene, Patterson, Singh, & Li, 2005), by strengthening the life-time fuel saving linkage to the purchase decision (Evans, 2008), which may be weak due to ‘imperfections in the market for fuel economy’ (Kunert & Kuhfeld, 2007, p. 315). The shortcoming of feebates is that they target only the new vehicle fleet. Hence, the emissions reduction that they achieve is gradual, and their full potential can only be achieved within the span of one or two decades as the whole fleet renews (BenDor & Ford, 2006). For this reason they are considered long-term measures. The effect of feebates on emission reduction will also be mitigated by the rebound effect (Evans, 2008). This effect arises whenever consumers buy more fuel efficient cars, thus face a lower cost per km and travel longer distances in response.

One of the appealing features of feebates is their potential neutrality on government finances. This is a theoretical possibility that stems from the fact that there always exists a zero-point level (where no taxes or subsidies are imposed) and a feebate structure such that the expected revenues equal the expected costs of the policy. In practice, however, calculating the optimal structure presents an important challenge, due to the uncertainty regarding consumers’ response to feebates and the elasticity of substitution between different car models. This uncertainty makes it difficult to determine the optimal feebate rate (Gallagher et al., 2007). However, Ford (1995, in BenDor & Ford, 2006, p. 1200) demonstrates that despite the uncertainties over market shares, it is possible to maintain a reasonable balance and control the finances, provided that the subsidisation plan is flexible enough.

4.1.2. Scrappage incentives

Scrappage incentives, also known as voluntary accelerated vehicle retirement programs, are rebates granted to owners of old vehicles upon scrappage (Evans, 2008). These incentives directly act on the fleet of old vehicles in circulation, which are responsible for a (relatively) disproportionately higher share of emissions than newer vehicles (Evans, 2008). The immediate benefit that this policy aims at is an earlier scrapping of some vehicles and a consequent reduction in fuel consumption due both to the higher fuel efficiency of newer vehicles, and to a reduction in the number of vehicles if the old ones are not replaced (Acutt &Dodgson, 1997). However, Alberini, Edelstein, and McConnell (1994) point out that scrappage incentives may lead some individuals to keep their vehicles longer so as to reach the age at which they become eligible for the scrappage scheme. Similarly, De Palma and Kilani (2008) show that scrapping premia may perversely delay the replacement of vehicles due to the fact that an old car will be more valuable with the scrapping scheme than without. The costs imposed by scrappage schemes on the government consist of the subsidies and administration costs. The schemes are generally considered relatively easy to implement (Hsu & Sperling, 1994, pp. 90–98). However, many scrappage programs have been financed by private companies as a way to achieve imposed regulatory standards or to obtain emissions offsets (Alberini, Harrington, & McConnell, 1998).

Assessing the benefits of a scrappage scheme is not an easy task, since there are a number of uncertainties over car use and emissions (Hsu & Sperling, 1994). Another problem associated with the evaluation of the cost effectiveness of scrappage schemes is that of moral hazard (only the worst cars get scrapped, which are hardly used anyway), which makes it difficult to adequately ‘reward actual reductions in the number of vehicles (Hahn, 1995, p. 239). The problem can be worse if it is difficult to estimate the remaining life-time of a particular vehicle (Hahn, 1995, p. 239). Vehicles that are scrapped under a scrappage scheme are likely to be those in worst condition and almost at the end of their life (Alberini et al., 1994). Old vehicles that are not replaced upon scrappage are likely not to have been in active use prior to being scrapped. Hence, their contribution to emissions and consequent social costs would have probably been lower than the scrappage bonus.

This may explain why the literature is not unanimous in its assessment of the benefits of scrappage schemes on emissions. BenDor and Ford (2006) predict that, even if all vehicles scrapped are immediately replaced, the scheme leads to large and immediate (HC) emissions reductions (BenDor & Ford, 2006). Sandström (2003), on the other hand, points out that the effect of scrappage policies on emissions is ambiguous. Alberini et al. (1998) show that the reduction in emissions is small unless a substantial scrappage premium is offered, which reduces the cost effectiveness of the scheme. Van Wee, Moll, and Dirks (2000) go further and claim that scrapping programs may lead to an increase in CO2 emissions over the life-cycle of vehicles. This is due to the fact that an increase in

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39 Sometimes the rebate is conditional on the purchase of a new car (on which a purchase tax is levied).
40 Moral hazard is related to the problem of asymmetric information, a situation in which one party in a transaction (such as for example the car owner) has more information than the other (the regulator). Moral hazard occurs when the party with more information (e.g., about the vehicle being very old and not even in use or about to be scrapped anyway) has an incentive to hide the information from the other party (e.g., the regulator, who is about to pay the owner of the vehicle an amount for scrapping the vehicle).
41 A common assumption in policies is that scrappage vehicles are similar to the average vehicle in a given model year cohort in terms of remaining life, distance driven and emission levels. But the self-selection bias makes this assumption wrong (Alberini, Harrington, & McConnell, 1995).
car production leads to increased emissions in assembly and new cars are faster and bigger than the old cars they replace. Retrofitting high-pollutant vehicles may thus be more cost effective than scrapping.

Careful design of eligibility requirements for scrappage schemes is a fundamental issue. Some scrappage schemes require that the vehicle be driven to the scrappage site so as to prevent appropriation of bounties for vehicles that would not have produced emissions because they were not operational. When the scrappage system is local, and aimed at reducing local air pollution, another important requirement is that the vehicle be registered in the local area, so as to prevent an inflow of vehicles from other areas with the sole purpose of scrapping them (Hahn, 1995). The effectiveness of a scrappage program in reducing emissions is greater if low-quality cars are the largest emitters, which may not always be the case. If a large portion of high-quality used cars contributes to pollution, and their value is higher than the scrappage premium, the achieved emissions reduction from the scheme will be relatively low (Mazumder & Wu, 2008). This is due to the fact that the value of cars is not necessarily negatively correlated with the emissions produced, since for example a larger car (of higher quality), may have the same market value as a smaller newer car which produces less emissions. Hence the subsidy will not be enough to induce scrappage of these high-quality high-emitters.

Scrappage schemes can be either permanent measures, or temporary measures introduced, for example, to boost car demand at a specific moment. A permanent scrappage scheme will accelerate the vehicle fleet turnover as it is equivalent to a negative purchase tax. If a purchase tax is levied, and refunded with interests at the time of scrappage, the system is revenue neutral (i.e., it has no impact on the government's finances), and the life-length of vehicles is reduced, although the size of the vehicle fleet is not affected. Sandström (2003) argues that a scrappage scheme will lead to an increase in the size of the vehicle fleet, which increases emissions. As a consequence, emissions may not be reduced, and may even increase.

A temporary scrappage scheme, on the other hand, will not have a long-term impact on the life-length of vehicles (Sandström, 2003). Temporary scrappage schemes have the potential to reduce emissions and burst sales in the short-term. The temporary burst of sales will also depend on the fact that people will typically delay their scrapping decision between the time the policy is announced and the time the policy becomes effective. This medium-term effect will be a reduced level of activity, since the age distribution of the vehicle fleet will have changed (Adda & Cooper, 2000). The scrappage scheme might even lead to an increase in emissions relative to the no-scrappage incentive scenario, if some technological breakthrough appears when cars have just been replaced as a consequence of the scheme (Sandström, 2003). Finally, it should be noted that there is no market failure rationale for scrappage policies and they can even be efficiency-reducing if petrol is priced according to marginal social cost.43

The introduction of scrappage incentives has an impact on the second-hand car market. Used cars will be imported either to be scrapped so as to take advantage of the bonus, or to replace vehicles which have been scrapped (Hahn, 1995). This may partially offset the emissions reduction, depending on the age of migrating vehicles (Dixon & Garber, 2001 in Evans, 2008). Policy-makers can avoid this effect through the introduction of taxes on imports, or through restrictions on program eligibility (Hahn, 1995). Careful design of the scrappage program is thus essential, in particular, in order to avoid creating demand for old vehicles (Gorham, 2002, p. 116). Scappage subsidies could theoretically lead to an increase in the price of used vehicles due to an increase in their demand (Alberini et al., 1994). This price increase would weaken the scrappage incentive (Gorham, 2002) and have regressive distributional effects, since people on lower incomes are more likely to purchase used cars. This problem can be avoided by implementing strict eligibility requirements for scrappage, for instance, requiring that, prior to scrappage, a car must have been registered for a certain amount of time in the area with the scrappage policy in place. Regarding the distributional effects, it should also be noted that people on low incomes are generally also more likely to own old vehicles and hence benefit from the scrappage premium (Acutt & Dodgson, 1997).

The appeal of scrappage schemes weakens over time. As the vehicle fleet approaches a zero emissions level, the gains to be made by scrapping old vehicles decreases. Hence, a scrappage scheme is best suited to be a transitional strategy (Hahn, 1995). The effectiveness of a scrappage program is also a function of location. In fact, local schemes do not affect the supply of used cars and are therefore less costly, as they do not need to respond to increases in the value of used cars by offering higher premia (Alberini et al., 1994). Furthermore, the emissions-reducing potential of scrappage schemes is stronger in highly polluted metropolitan areas with a large share of old vehicles (Hahn, 1995).

Scappage schemes need not be stand-alone measures and indeed are unlikely to be effective instruments for emissions reduction, if adopted alone (Alberini et al., 1998, p. 20). It is possible to combine scrappage incentives with a system of fee-bates, or to graduate incentives on the basis of the fuel economy of the purchased car. This combination would have the additional benefit of creating incentives regarding both old and new vehicles, promoting adoption of the cleanest new vehicles (Evans, 2008). The system would lead to both a short-term and a long-term decrease in vehicle emissions, and fees on high-emitting new cars could finance subsidies on clean new vehicles and scrappage (BenDor & Ford, 2006).

The existence of inspection and maintenance programs can also affect the effectiveness of scrappage incentives. More stringency in inspection and maintenance leads to more scrappage for a given bonus, but could reduce cost effectiveness (Hahn, 1995, p. 222). Alberini et al. (1998) show that the introduction of a scrappage program under an existing inspection and maintenance program can be welfare-increasing. Gorham (2002) discusses the possibility of tying a scrappage program to the grant of a free public transport pass. Other alternative policies to scrappage incentives include a reduction of purchase taxes relative to usage taxes, or a differentiation of usage taxes on the basis of age (Sandström, 2003).

4.2. Taxes on usage of a vehicle

Usage taxes are imposed on the basis of a vehicle’s usage. If the vehicle is left idle, there are no usage taxes to be paid. Examples of these charges include fuel taxes, tolls, congestion charges and parking charges. Together with ownership taxes, usage charges constitute an important policy, which affects demand and supply

42 Sales tend to over-react, hence they are not a good benchmark when analysing car markets (Sandström, 2003).
43 If petrol is priced according to marginal social cost, all the externalities have been internalised already and there is no need for any further measures.
44 This statement depends on the type of emissions and fuel in question. While local air pollutants may be reduced and even approach zero, as long as vehicles run on oil-based fuels carbon dioxide emissions may go down in time but they will never approach the zero level. Carbon emissions are an inevitable product of oil-based fuel combustion.
and reduces road transport externalities, in particular environmental externalities, such as global warming resulting from CO₂ emissions.

4.2.1. Emission taxes

Emission taxes are in theory direct Pigouvian taxes, which should be set equal to the marginal cost of the actual level of emissions generated by each vehicle. Emission taxes are the first best instrument to correct emission externalities and induce the optimal driving behaviour and vehicle’s purchase and usage choice (Acutt & Dodgson, 1997, p. 23; Jonhstone & Karousakis, 1999, p. 100). However, under current technology, direct charging for emissions is not feasible due to the lack of cost effectiveness and impracticability of monitoring techniques (Fullerton & Gan, 2005, p. 300; Fullerton & West, 2000, p. 1; Plaut, 1998, p. 194).

The level of emissions generated by a vehicle does not only depend on fuel usage, but also on contingencies that are less easy to pin down. These include vehicle characteristics such as size, age, and maintenance, driving behaviour such as frequency of cold-start ups and speed, location and time-dependent characteristics such as air temperature (Fullerton & West, 2000, p. 3). Hence, implementation of an emission tax would require monitoring of actual pollution generated by each vehicle on circulation (Jonhstone & Karousakis, 1999). Remote sensing of emissions with infrared beams has been suggested as a possibility for effective monitoring. However, this system has many weaknesses. Complexity, lack of detection of non-tailpipe emissions and of some types of tailpipe pollutants reduces its appeal (Johnston and Karousakis, 1999, p. 100; Fullerton & Gan, 2005, p. 300).

Given the practical obstacles to the implementation of an emissions tax, Fullerton and West (2002) demonstrate that the same efficiency can be achieved with a petrol tax that depends on fuel type, engine size and pollution control equipment or with a tax that depends on distance. Having said that, given that emissions of air pollutants per km (which have mainly local and regional impacts rather than global impacts) have declined substantially with the introduction of tighter standards in different countries around the world, a tax to control these emissions may not be necessary after all.⁴⁵

One type of emission tax that focuses on a specific type of emission, namely carbon, is a carbon tax. This instrument would be almost ideal in the fight against global warming. Almost refers to the fact that other GHG such as methane are emitted from vehicles and contribute to global warming (Harrington & McConnell, 2003). A carbon tax would be implemented as a tax on the carbon content of fuel, and could be paid at the moment of fuel purchase, as it is currently the case with fuel duties. A carbon tax has an impact both on the choice of fuel type and on the aggregate fuel demand, as determined by vehicle fuel economy and vehicle usage (Gorham, 2002). It could lead to dynamically efficient choices of ‘ever cleaner technology and energy conservation’ (Pearce, 1991, p. 942). The tax could be accompanied by reductions in incentive distorting taxes such as income taxes (double dividend feature), so as to be revenue neutral on the government’s budget.

An “optimal” carbon tax would be one designed to meet a specific target of CO₂ emissions reductions. However, there are several drawbacks, which must be considered. First, in order to design such a tax, it would be necessary to know with precision the inter-fuel substitution elasticities (since the carbon tax on a fuel varies by its carbon content), as well as the income and price elasticities.⁴⁷ Second, a carbon tax is potentially regressive, since lower income households may spend more on carbon intensive activities. Third, the tax may impact competitiveness of the country where the tax is imposed relative to other countries (Pearce, 1991). Lastly, a tax based only on CO₂ may distort incentives in emission-reducing activities towards activities that increase local air pollutants, since CO₂ emissions and other emissions may be negatively correlated.

4.2.2. Fuel taxes

Fuel taxes are applied in virtually every developed and developing country in the world, as they constitute a low cost effective revenue generating instrument. However, they are typically defended by politicians on environmental grounds (Newbery, 2005a, p. 24) and indeed they have a potential role as emissions reducing instruments. The most striking advantage of fuel taxes over other policy instruments is their administrative simplicity, due to the fact that a tax collection system is already in place. Collection of fuel taxes can occur at the level of fuel wholesalers or refiners, of which there are relatively few, thus reducing collection costs and the likelihood of fraud or evasion (Wachs, 2003), or it can occur at the level of retailers, e.g. at the petrol station. The tax burden will ultimately fall on consumers, regardless of where they are collected, as the cost is likely to be passed on to them. Although fuel taxes generate revenues for the government which can be reinvested profitably, the burden on drivers, and spill-over effects on the economy contribute to making fuel taxes unpopular, and reduce the political viability of a tax increase.

Fuel taxes can serve as price signals (Wachs, 2003) to drivers on the social cost of the externalities they produce. In order for this to occur, it is crucial that the structure of the taxes be transparent and the composition of the tax well understood by motorists. If motorists are not aware of the changes in behaviour that will reduce their tax payments, the tax will not have the desired effects (Gorham, 2002). CO₂ emissions are closely correlated to fuel consumption, and thus they can be easily internalised through a fuel tax. Congestion, accidents, and local air pollution are externalities which are only indirectly targeted by fuel taxes. The total costs of these externalities, however, have been estimated as being 14 times the cost of GHG emissions (Evans, 2008). Hence a tax based on local emissions, distance driven, or peak-time congestion would be better suited than a fuel tax to internalise these other externalities (Newbery, 2005a; Parry & Small, 2005).

An optimal fuel tax requires a fair amount of information on private and social costs and benefits for its calculation. If ad valorem, the optimal tax would ideally need to be readjusted as oil prices change (Watters et al., 2006, p. 2). Fuel prices, however vary on a daily basis and there is typically going to be an implementation lag that prevents the adoption of the correct tax. Adjustment of taxes to match the oil price would also create uncertainties in government’s revenues (Leicester, 2005).

Nonetheless, fuel taxes have been cited as an extremely effective climate policy instrument. Sterner (2007) for example, stresses their importance as instruments to reduce environmental externalities and keep emissions under control. He argues that had Europe followed the US pattern of low fuel taxes, it would have had twice as much fuel consumption as it has now. Hence any attempt to abandon fuel taxation light-heartedly, e.g. as part of integration of transport into the EU ETS, could be dangerous.

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⁴⁵ This does not apply to CO₂ emissions, which have not been subject to the same type of standards.
⁴⁶ In its application a carbon tax would effectively be a specific type of fuel tax. Fuel taxes are discussed in more depth in Section 4.2.2.
⁴⁷ Note that this information can be learned and the tax adjusted iteratively.
The immediate effect of an increase in fuel taxes is on demand, in the form of a reduction in km driven, through the impact of taxes on the marginal cost of driving. More long-term effects include incentives to scrapping old vehicles, reducing car ownership, and favouring vehicles with higher fuel economy at the time of purchase. Cobb (1999) finds that the predominant effect of an increase in petrol tax in the US would be a shift towards more fuel efficient vehicles; this would be accompanied by a modest modal shift in the direction of public transport, and a minor shift to carpooling. A supply effect is also triggered by the fuel tax, by providing price-incentives to invest in cleaner technology and produce vehicles with higher fuel economy to gain a competitive advantage. Surveying the literature, Graham and Glaister (2004, pp. 270–271) and Goodwin, Dargay, and Hanly (2004, p. 282) report the short-run price elasticity of fuel demand to average around −0.2 to −0.3, and the long-run elasticity to average around −0.6 to −0.8. Small and Van Dender (2007), on the other hand, report a more moderate long run elasticity of −0.43 (p. 26). The immediate reduction in driving will be large; hence the environmental conditions will not improve in the short-term (Sipes & Mendelsohn, 2001). Fuel taxes thus have greater potential to reduce emissions in the long-term than in the short-term, although in the long-term a rebound effect may be observed. This occurs if a higher fuel economy of vehicles is accompanied by an increase in distance travelled, due to the reduced cost per km.

The distributional impacts of fuel taxes have been subject to considerable debate in the literature. In determining the distributional impact of fuel taxes it is important to keep in mind whether the measure of income considered is based on annual income, or the less variable life-time income. Poterba (1989) thus shows that the measure of income considered is based on annual income, or the less variable life-time income. There is general agreement that when all households are considered, fuel taxes are progressive, but if only car-owning households are considered fuel taxes are regressive (Acutt & Dodgson, 1997; Blow & Crawford, 1997; Leicester, 2005; Johnson, McKay, & Smith, 1990; Santos & Catchesides, 2005; West, 2004). Apart from car-ownership, other factors that play a role in the estimation of distributional impacts are the extent to and the way in which revenues are recycled. Bento, Goulder, Henry, Jacobsen, and von Haefen (2005) find that if revenues are recycled to households in proportion to their fuel-tax payments, the impact of the fuel tax is close to proportional. On the other hand, if revenues are recycled in proportion to households’ income, the impact of the fuel tax is highly regressive. Wachs (2003) argues that fuel taxes are less regressive than sales taxes as instruments to finance public transport expenditures, whose primary beneficiaries are lower income people. Graham and Glaister (2004, p. 270) find that the income elasticity of fuel demand is between 0.3 and 0.5 in the short-run and between 0.5 and 1.5 in the long-run. Goodwin et al. (2004, p. 284) report the mean income elasticity of fuel consumption to be 0.39 in the short-run and 1.08 in the long run. This is equivalent to saying that a household that doubles its income in the short-run will increase fuel demand by roughly 30–50 per cent. In other words, fuel taxes will hit harder on lower income groups, unless they are redistributed.

4.2.2.1. Fuel duty differentials. Different levels of taxes can be applied to fuels that differ in composition. Since different types of fuels are associated with different types and intensities of emissions, fuel duty differentials typically affect the mix of emissions (Acutt & Dodgson, 1997). Tax differentials can thus become important and cost effective instruments to discourage the use of the more environmental damaging fuels (Knight et al., 2000). Taxation of certain components of fuel, such as lead, sulphur, oxygen content or octane rating may also create strong incentives to change the fuel content (Gorham, 2002). It may not always be easy to find the efficient tax, i.e., the tax which covers the marginal external costs caused by a specific pollutant.

4.2.2.1.1. Diesel vs. petrol taxation. In the Pre-Euro I phase (pre-1993) when petrol cars were not equipped with a three-way catalyst, diesel was superior to petrol in terms of emissions (CO, HC, NOx, CO2) per vehicle kilometre in urban conditions, with the only exception of PM emissions, where petrol was far superior. However, under Euro IV (post-2006), diesel cars emit less CO50 and less CO2 than petrol cars with three-way catalysts, but their emissions of HC, NOx and PM particulates are greater (UK DfT, 2008, Table 3.6, p. 55). However, the difference between diesel- and petrol-fuelled cars in terms of their emissions of HC, NOx and PM gradually decreased over the period 1993–2006, as cleaner fuels and vehicles were introduced in Europe (UK DfT, 2008, Table 3.6, p. 55). Although these conclusions are drawn on the basis of European data, similar conclusions would be reached for countries which adopted similar standards. For countries that have not regulated emissions in this way, the case of Europe proves that it is (technologically) possible to achieve such emission differentials.

To compare the external costs of diesel and petrol, or in other words, to decide whether diesel or petrol should be favoured, either shadow prices need to be determined for each pollutant (a practical difficulty which is common to all Pigouvian taxes) or at least, some (subjective) weighting needs to be assigned. Crawford and Smith (1995, p. 46) suggest the use of relative weights but emissions from petrol and diesel have changed drastically since they wrote their paper. It is now clear from the data on emissions that, unless carbon monoxide and CO2 have a much higher social cost than all the other emissions, the use of diesel should now be discouraged and one way of doing this is through higher taxes.

Along these lines, Mayeres and Proost (2001a) argue that diesel has much higher environmental costs than petrol, since the social costs of PM emissions are high. Having said that, the difference in emissions of PM between petrol and diesel cars has declined between 2001 (when they wrote their paper) and 2009. If climate change and CO2 emissions were to be given the highest weight, there could be an argument for lower taxes on diesel than on petrol. Whatever the reason, EU countries have not increased diesel taxes relative to petrol taxes over time.

Mayeres and Proost (2001a) further argue that where there are constraints on the choice of tax instruments, a change in ownership taxes to discourage the use of diesel cars is an important instrument. Constraints could arise, for example, from a difficulty in differentiating fuel taxes between private and commercial use, and between cars and lorries. Giblin and McNabola (2009) find that ownership taxes have a higher impact on new vehicle purchases than any fuel duty differentials.50

48 On the other hand, de Palma and Kilani (2008) argue that the higher the price of petrol, the later a vehicle will be replaced, and hence the older the cars in circulation will be. This result differs from previous results (Fullerton & Gan, 2005; Fullerton & West, 2002) because it accounts for the fact that less usage leads to a lower replacement rate.

50 They model the potential impacts of the new carbon based tax system for new vehicles purchased from 1 July 2008 by the Irish government and find that CO2 emissions from private transport will be reduced by 3 per cent with respect to 2007 levels.
In practice, many countries tax diesel less than petrol, and produce an artificial price differential that is much higher than that warranted by international prices. The rationale behind these differentials is often political, to protect the competitiveness of sectors such as agriculture, that rely on diesel (Gorham, 2002). An economic rationale for the observed duty differential lies in the increased efficiency of taxation when applied to consumers, combined with the fact that diesel is used as an input in production by lorries and industries more than petrol. Diesel enjoys an intrinsic fuel economy advantage over petrol, since kilometres per litre of diesel are higher, all else being equal. The tax incentives add to that intrinsic advantage and constitute an added incentive for the use of diesel, which may not be warranted from a welfare perspective. The lower usage cost of diesel cars is also responsible for a rebound effect, i.e., an increase in distances travelled, triggered by a lower cost per km, which partially offsets the fuel savings (Gorham, 2002).

Finally, let us return to a point mentioned above, which could serve as an argument for having lower taxes on diesel than on petrol. Since CO₂ emissions per vehicle km from diesel are lower than those from petrol, increasing the share of diesel vehicles would reduce CO₂ emissions. If the key objective of the tax were to address the issue of climate change, putting a greater weight on CO₂ relative to other emissions could then be justified.

4.2.2.12. Cleaner fuels. In order to promote the adoption of cleaner fuels (unleaded instead of leaded petrol or low-sulphur instead of high-sulphur diesel, etc) governments can, and indeed have, set duty differentials or subsidies, so that cleaner fuels become more attractive. These subsidies or duty differentials can help curb emissions in transport, although their effect on congestion need not be beneficial (Timilsina & Dulal, 2008), as longer distances may be driven. Some specific successful examples are discussed in Section 5.

When alternative fuels can be utilised in existing vehicles without any adaptation, then theory prescribes that relative taxation should reflect the relative environmental damage produced by different fuels. When major adaptations on existing vehicles or their replacement is needed to adopt cleaner fuels, then both fuel and vehicle taxation matter. In particular, if cleaner fuels are subsidised, the risk is that subsidies will benefit users who drive long distances and would have found it profitable to buy the cleaner vehicles even without any fiscal incentives. Such a policy may thus imply a higher deadweight loss than a policy that subsidises cleaner vehicles rather than cleaner fuels (Crawford & Smith, 1995).

4.2.3. Vehicle km travelled taxation

A type of usage tax that can be used as an alternative or complement to fuel taxation in order to reduce emissions from road transport is a tax on distance driven. Such a tax could be levied once a year, as part of the annual ownership tax, on the basis of km travelled as registered on the odometer. Kilometres could be checked during regular mandatory safety inspections so as to reduce the administrative cost. Vehicle-km travelled (VKT) taxes are associated with a high incentive to cheat by rolling back old odometers. New electronic odometers will be more cheat-proof in this respect.

The advantage of a VKT tax is that it avoids the rebound effect previously discussed for fuel taxes, and acts directly to reduce distance travelled. Indirectly, a VKT tax will reduce congestion (Ubbels, Rietveld, & Peeters, 2002). However, such a tax is unlikely to trigger any technological emission abatement improvement, or any shift in demand for more fuel efficient vehicles. Hence its benefit is more limited. This drawback could be reduced by differentiating the VKT charge on the basis of the vehicle’s fuel economy (European Commission, 1995).

4.2.4. Congestion charges

The simple idea behind a congestion charge that reflects the additional social costs of congestion is to confront the trip maker with the true social cost of his journey. In theory, a congestion charge is a corrective, or Pigouvian, tax, set equal to the marginal cost of congestion. A congestion charge ensures that only cost-justified journeys are made and the scarce road space is allocated to those who value it most. The effect of charging is to reduce travel demand and congestion, and to increase speeds and the total net benefits of travel.

An efficient equilibrium is one in which the only journeys made are those whose value exceed their marginal social cost. Thus, journeys are made until the last valuable trip yields a benefit equal to the marginal social cost, which is equal to the private cost plus the congestion externality. The efficient congestion charge can be measured by the segment DE in Fig. 1 (in Section 1). The charge is equal to the MCC at the efficient level of traffic, qₚ, in Fig. 1. The government revenue from this charge is FDEG and the social gain is DMC. The disbenefit to those who are priced off is given by area DBC, which can be approximated by (qₚqₚ * FA)/2. The total loss of consumer surplus can be measured by FADC and the loss for existing users who continue to drive is FABD. The benefit to existing users (in the form of higher speeds) is given by area AGBE.

This description gives a very simplified representation of the problem. Nonetheless it is useful and conclusions do not change that much when the model is made more complex. This simple model of congestion charging has been extended in several directions.

Vickrey (1969) proposed a bottleneck model, where congestion is assumed to arise when vehicles queue behind a bottleneck. If commuters need to arrive at work at a certain time and there is a bottleneck on the way, they will bear costs associated with early and late arrival, which together with the congestion charge, are added to the cost of the trip. This bottleneck model was further extended, mainly by Arnott, De Palma, and Lindsay (1988, 1990a, 1990b, 1992, 1993, 1994, 1998) to include heterogeneous commuters, route choice, stochasticity in capacity, elastic demand, and time-varying congestion charges.

Congestion charges in the static model and in the bottleneck model are first best solutions, as the underlying assumptions are that there are no constraints of any sort (i.e., the regulator can price any link at any time) and no imperfections in the rest of the economy. However, in real world situations, due to constraints, congestion charges are likely to be second best solutions. Constraints could include a situation where only part of the road network or only some road users are priced (Liu & McDonald, 1999; Small & Yan, 2001; Verhoef, 2000, 2002a, 2002b; Verhoef, Nijkamp, & Rietveld, 1996). Ignoring these constraints and naively implementing first best congestion charges (equal to marginal congestion costs) can lead to lower welfare gains from pricing than the gains from second best pricing, which by definition achieves the highest gains given the constraints that apply (Small & Verhoef, 2007). Second best scenarios may also include complements to,

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51 A different tax on diesel could in theory be applied to cars and lorries (or private and commercial vehicles) but this could in practice prove administratively expensive.
and substitutes for, a priced link or area. Parking charges, for example, have frequently been suggested as an alternative to congestion charges (Verhoef et al., 1995).

Another problem that would interfere with first best pricing is that of distortions in other sectors of the economy. One classic example is that of pre-existing distortionary taxation on labour. The interaction between congestion charges and distortionary labour/income taxes has been analysed by Mayeres and Proost (2001b) and Parry and Bento (2001). An important result in this respect is that the overall welfare gains from congestion charging not only depend on the initial impacts of the charge, but also on the use of the revenues, which could be allocated, as already discussed for the case of fuel taxes, to reducing distortionary taxes in the economy, such as for example, labour/income taxes.

Parry and Bento (2001) thus find that congestion taxes tend to increase the total cost of commuting (although this is partly offset by lower travel time costs), discouraging labour force participation, leading to a welfare loss that can be greater than the Pigouvian welfare gain resulting from the internalisation of the congestion externality. A positive net impact on labour supply, on the other hand, can be achieved by using the congestion charge revenues to reduce income taxes, raising 'the overall welfare gain from the congestion tax by around 100 percent' (Parry & Bento, 2001, p. 645).

Furthermore, the congestion charge could be set so as to also internalise the environmental externalities from driving, thus becoming a combined congestion and environmental charge. As already explained in Section 1, the environmental externality includes a variety of elements, ranging from local, regional and global pollutants, to noise, water contamination, and visual intrusion. A corrective charge could then be designed to include both the congestion and the environmental externality (Button, 2004, p. 14).

As discussed above, however, fuel taxes can approximate optimal environmental charges because most environmental externalities are closely dependent on fuel consumption.

Finally, an important link to congestion charges is that of capacity choice. Keeler and Small (1977) build on previous work by Mohring and Harwitz (1962), Vickrey (1963), Strotz (1964, in Keeler & Small, 1977), and Mohring (1970, in Keeler & Small, 1977) and show that revenues from an efficient congestion charge (equal to the marginal congestion cost) are equal to the rental cost of the highway under the assumptions of constant returns to scale, indivisibility of road construction, and independence of demand in each period from prices in other periods. They define rental cost of the highway as the sum of construction costs (on which interest and amortisation are computed), maintenance costs, and land acquisition costs. Although at first sight there may seem to be no link between efficient congestion charges and the rental cost of the road there is one, and this is given by a function of net benefits that includes both benefits and costs to the users and the costs of the highway. By maximising the net benefits of all trips on that road over the life of the road, computed as the difference between the net benefits to road users and the rental cost of the road they arrive at fairly standard results: (a) the price paid by the road user should equal the average rental cost plus the congestion externality, and (b) lane capacity (or width) should be expanded to the point where the marginal cost of an extra unit of capacity is equal to the marginal value of user cost savings (typically vehicle operating and time cost savings) brought about by that expansion. Combining these results, simple algebraic manipulation shows that the revenue from the efficient congestion charge is exactly equal to the rental cost of the road, under the assumptions already listed above.

The main problem with the model above is that it assumes the rental cost of the highway to be exclusively dependent on its width, and not on traffic. Newbery (1989) improves the model by assuming that maintenance to repair road damage does depend on traffic. Rather than having just a congestion charge, he proposes that vehicles should also be charged for the damage they cause on the road. A road user charge would then include a congestion charge element and a road damage charge element.

4.2.5. Parking charges

Much of the literature on pricing relates to parking as a common property resource, which is over-exploited if under-priced (Anderson & de Palma, 2007). The provision and use of the parking slot entails a marginal cost (Verhoef et al., 1995). The search for parking space in congested urban areas is also one of the factors contributing to travel congestion itself (Arnott & Rowe, 1999, Arnott & Icli, 2006). Nonetheless, it is not uncommon in urban areas for drivers to park for free, at least at the workplace. The problem of parking being under-priced when considering its social costs has a simple efficient solution: appropriate pricing of parking slots. The administrative complexity of the system is likely to be low, requiring little investment costs and being easy to introduce. Political acceptability for parking fees is likely to be higher than for congestion charging (Zatti, 2004). Revenues raised through parking fees can be reinvested. However, there is the need for monitoring and for setting penalties to mitigate the risk of avoidance of parking charges (Accot & Dodgson, 1997).

Parking charges have also been suggested in the literature as a second best instrument to deal with road transport externalities, such as for example, congestion (Verhoef et al., 1995) and emissions. Parking charges increase the total cost of a trip (Timilsina & Dulal, 2008). This could reduce the attractiveness of a given urban area altogether, or discourage vehicle usage, resulting in a decrease in the inflow of vehicles (Eisenkopf, 2008). In addition, the introduction of parking fees could help matching demand with supply, thus reducing parking search time, and improving mobility (Zatti, 2004). However, parking policies are not optimal instruments (in a first best sense) to regulate costs that are largely dependent on length and route of the trips. They may even be counterproductive, as they charge the average, rather than the marginal, external cost, thus cross-subsidising longer trips through shorter trips (Verhoef et al., 1995).

The possible effects of a parking charge on drivers include, according to Feeney (1989, in Timilsina & Dulal, 2008, p. 27) a change in parking location and/or trip destination, a change in journey-time, a modal change, or the cancellation of the trip altogether. The long-term effect could also be a reduction in car-ownership level, although the impact is likely to be marginal (Accot & Dodgson, 1997, p. 27).

Efficient or inefficient parking fees, however, both allocate scarce parking spaces without necessarily reducing peak traffic congestion and emissions. These charges affect drivers whose destinations are in the area where the parking charges apply, and fail to differentiate according to trip lengths or routes followed (Verhoef et al., 1995) and emission levels. Thus, parking fees are not effective in reducing through traffic. In fact, the number of drivers passing through the area and the length of their trips may increase in response to the parking charges, as they extend their trips to take advantage of free parking slots outside the area where the charges apply. One way to counteract this would be to avoid discontinuities in the charges applied, and to have uniform charges in wide areas to increase effectiveness (Zatti, 2004).

Although information and enforcement intensive, parking charges could be designed to vary according to time of day, location

53 This comes at a cost in terms of competitiveness of the city, if they are applied on a large scale (Eisenkopf, 2008). On the other hand, effective, fast and reliable public transport provision may counteract this effect.
of the parking place, and according to the vehicle’s fuel economy. The first option is an important element both as a first best instrument to regulate time-dependent parking externalities, and as a second best instrument to regulate time-dependent driving externalities. The second option is important because external costs vary with the congestion of the area. For example, external costs tend to be higher in the city centre, and generally lower for off-street parking places (e.g. underground) than for on-street ones (Zatti, 2004).

Glazer and Niskanen (1992) show that when road usage charges are sub-optimal, parking charges can increase welfare if they are lump-sum fees, but not if they are increasing in parking time. In fact, if charges vary with length of stay, people park for a shorter time, and more people use the parking slots, thus increasing traffic. Verhoef et al. (1995) show that efficient parking charges can only reduce traffic congestion (by mimicking congestion charges) under very restrictive assumptions. These assumptions were already summarised in Section 2, and include: (a) each individual drives the same distance, (b) congestion is equally spread over the road network, (c) the government (regulator or local authority) has full control over all parking spaces, (d) every car is parked in a public parking space, and (e) all cars are parked for the same length of time. Arnott, De Palma, and Lindsay (1991) and Calthrop, Proost, and van Dender (2000) show that congestion and parking charges need to be set simultaneously. Using the bottleneck model, originally developed by Vickrey (1969), Arnott et al. (1991) demonstrate that the joint implementation of an optimal congestion charge and a location-dependent parking fee yield an efficient equilibrium. Using a numerical simulation model and assuming a cordon charging system, Calthrop et al. (2000) find that as parking charges approach their efficient level, the level of optimal cordon charges falls. Similarly, with a cordon charge, the level of efficient parking charges falls. In this sense, parking charges can be seen as a substitute for congestion charges. However, Button (2006) argues that the effectiveness of parking fees as a road pricing surrogate are highly context specific, which brings us back to the restrictive assumptions made by Verhoef et al. (1995).

Arnott and Inci (2006) develop a model which integrates traffic congestion and saturated on-street parking, with cars causing congestion when looking for a parking space. They find that the efficient on-street parking charge is a charge that eliminates the activity of driving to find a parking space, regardless of whether the quantity of on-street parking spaces is optimal or not. Arnott and Rowe (2009) extend that model to include on-street parking, off-street privately provided parking, and traffic congestion, and conclude that increasing on-street parking charges generates efficiency gains that may be several times as large as the increased revenue raised.

Arnott (2006) and Calthrop and Proost (2006) arrive to similar conclusions using different models which integrate both on-street and off-street privately provided parking. Arnott (2006) finds that the number of cars looking for a parking space adjusts to equalise the full prices of on-street and off-street privately provided parking spaces. Calthrop and Proost (2006) show that the optimal on-street parking charge equals the marginal cost of providing off-street parking spaces at the optimal quantity. If off-street parking is supplied under constant returns to scale, they prove that the efficient on-street parking charge should be equal to the off-street one.

An alternative to parking charges would be a mandated restriction on parking supply. Verhoef et al. (1995) show that mandated restrictions are inferior on the basis of information and efficiency arguments, as discussed in Section 2. In fact, individuals may not know that there are parking restrictions, hence undertake the trips anyhow, and then drive around to find a parking place. Parking charges discriminate on the basis of willingness to pay, while parking restrictions do not.

### 4.2.6. Pay-as-you-drive insurance

Pay-as-you-drive (PAYD) insurance differs from standard insurance in that the premium is dependent on the annual distance travelled. Thus, insurance becomes a variable cost (Litman, 2008). As in standard insurance, the premium can be conditioned on the driver’s rating factor (Parry et al., 2007, p. 394), which is a function of age, crash record, and region (Parry, 2005, p. 288). The advantage of PAYD insurance over conventional insurance is that it price-discriminates more successfully between travellers on the basis of the distance they drive, which is correlated to their willingness to pay for insurance. As PAYD insurance reduces premia for short distances driven, implementing such insurance schemes can be expected to reduce the number of uninsured drivers (Parry, 2005, p. 288). Insurance schemes such as PAYD are also more accurate, as they increase the correlation between premia and expected claims to the insurance company (Litman, 2008). Under PAYD, travellers do not “over-consume”, or “over-drive”, as the amount they pay increases with distance, very different from a scenario where they pay a fixed sunk-cost at the beginning (Evans, 2008).

Parry (2005) argues that although fuel taxes are more cost effective in reducing emissions and improving fuel economy, PAYD insurance, which is politically more acceptable, reduces distance-related externalities, such as congestion, accidents and local emissions, ‘far more than fuel taxes’ (p. 288). He also shows that PAYD is slightly more efficient than a VKT tax for a given reduction in demand for petrol. Evans (2008) also argues that the extra-premium paid by drivers who drive longer distances will net-out with the lower premium paid by drivers who drive shorter distances.

The distributional impacts of PAYD insurance have been suggested to be progressive, since there is a positive correlation between income and km travelled. Bordoff and Noel (2008) thus note that on average families on low incomes will benefit from PAYD insurance schemes, as they ‘make up a disproportionately large fraction of the low-mileage drivers within any risk class’ (p. 39). However, Wenzel (1995) argues that PAYD insurance could be regressive, as the burden of the scheme relative to a standard insurance scheme could be higher for lower and middle-income households. It should also be noted that if the elasticity of distance driven with respect to income is lower than 1, PAYD insurance is likely to be regressive.

Considering the issue of political feasibility, PAYD insurance schemes may compare favourably to other policy measures such as fuel taxes. Unlike fuel taxes, implementing PAYD insurance schemes does not transfer large tax payments from motorists to the government. For the average driver, PAYD schemes do not increase driving costs, leading Parry (2005) to conclude that ‘hence political opposition to this policy should be more muted’ (p. 288).

Implementation of PAYD at a large scale may be difficult because insurance companies would have to incur costs of monitoring that would outweigh their direct benefits. A socially inefficient choice by insurers (i.e. failure to implement PAYD insurance schemes) is likely, as the benefits to society from such a scheme would be large, but these benefits would not be internalised by insurance companies. Edlin (2003, pp. 56–57) lists the positive externalities or social benefits from PAYD. These include: reduction in accident costs (which benefits insured vehicles, with or without PAYD, uninsured and underinsured ones), reduction in congestion, reduction in pollution and reduction in road maintenance costs.

The main concern with PAYD insurance implemented at a large scale is related to monitoring costs. Odometer reading would be necessary to determine the appropriate charges, and fraud would be an obstacle as it is quite possible to roll odometers back (Bordoff &
Electronic odometres, more difficult to tamper with, may be the solution. Other implementation costs include the calculation of new premia and development of new procedures for the insurance companies (Litman, 2008). There exist patents for PAYD schemes that may discourage adoption of the system by other insurers (Bordoff & Noel, 2008). In addition, first-movers will face adverse-selection in consumers’ choice, reducing the benefits of PAYD. In order to counteract this effect, an increase in the average premium of the fixed insurance schemes could be introduced at the same time in order for the insurance companies not to make losses. Insurers also have concerns that the distance foregone (not driven) after the introduction of a PAYD scheme will be less risky than the average driven distance or average number of km. Hence, the reduction in the premium will be more than the reduction in claim costs, leading to losses for the insurers. Litman (2008) argues that empirical evidence suggests that the opposite holds: the reduction in cost claims is more than one to one with the reduction in km travelled. Along the same lines, Bordoff and Noel (2008) argue that the number of accidents increases less than one to one with vehicle-km travelled. In any case, if the adoption of a PAYD scheme is successful and leads to a reduction in km travelled, the total premium will be reduced. This, even if matched by a corresponding reduction in claim costs, will lead to a lower investment income for insurance companies, and hence to lower profits (Litman, 2008).

4.3. Ownership vs. usage taxes

The transport economics literature is very critical of ownership taxes, because they are blunt and indirect tools to address transport externalities. However, ownership taxes are easier to implement and may face less political opposition (Barter, 2005). The choice of tax weights applied to the different stages of ownership, purchase, usage and scrappage of a vehicle is a key determinant of purchasing and travel behaviour, and consequently of the life-time CO₂ emissions of a vehicle from production to usage, maintenance and disposal (Hayashi et al., 2001). In principle, variable taxes such as fuel duties, distance based taxes and congestion charges are superior to fixed taxes (on purchase and/or ownership) at tackling externalities.

However, when the government is constrained in its capability to impose the efficient variable taxes, fixed taxes can be important second best instruments (De Borger & Mayeres, 2007, pp. 1178–1179). Ownership and purchase taxes may be better tools for directing purchase behaviour towards vehicles associated with less negative externalities (Barter, 2005, p. 526), since consumers’ choice of vehicle class and their decisions regarding vehicle disposal and repurchase are less directly targeted by usage taxes (Hayashi et al., 2001).

Koopman (1995) compares the CO₂ emissions reducing potential of a range of fixed and variable taxes through simulations with a partial equilibrium model of European passenger transport. He finds that carbon taxes are superior to feebate schemes; these in turn perform better than annual ownership taxes, while purchase taxes are the least cost effective instrument. Different policies induce different behavioural responses. Feebates induce vehicle fleet compositional change, towards more fuel efficient vehicles. The purchase of more fuel efficient vehicles leads to a moderate rebound effect, since the lower marginal cost of driving leads to higher usage. Feebate schemes are increasingly more costly relative to CO₂ taxes as CO₂ reduction targets tighten, as the costs of achieving a given fuel efficiency ‘increase more than proportionally’ in the fuel efficiency target. Finally, a feebate scheme may be needed in combination with a CO₂ tax if consumers underestimate the total vehicle life-time fuel consumption at the time of purchase (Koopman, 1995, p. 67).

De Borger and Mayeres (2007) construct a numerical optimisation model to study optimal fixed and variable taxes on petrol and diesel cars in Belgium. They find that shifting the balance of taxes towards variable taxes does not necessarily improve welfare. They also show that the combination of peak road pricing with revenue recycling in the form of higher transport subsidies or lower petrol taxes is the policy with the highest marginal effect.

Fullerton and West (2003) show on the basis of Californian data that a fuel tax alone can achieve 62 per cent of the welfare gain obtained in a first best scenario with a tax on emissions. The introduction of a two-part schedule consisting of a fuel tax and a subsidy to new cars is an improvement upon this, achieving a welfare gain that is 71 per cent of that obtainable in the first best scenario. However, distributional effects and administrative costs are not considered in the analysis, and may lead to different conclusions.

Chia et al. (2001) compare ownership and usage taxes on the basis of Singapore’s data in the mid 1990s. They assume that travellers can choose whether to travel by car or by bus. Trips by car entail fixed costs, but lower variable costs. Usage taxes are found to be superior to ownership taxes for internalising congestion externalities, but are less revenue-efficient. They argue that when combining ownership and usage taxes, one should first optimise over ownership taxes, and then introduce usage taxes.

Newbery (2005a, p. 29) argues that petrol and diesel taxes can be set at equal rates, as long as the difference in the damage caused by emissions from diesel and petrol vehicles is collected through the annual vehicle excise duty.

4.4. Company cars and other incentives

The tax treatment of commuting expenses and employer provided transport-related fringe benefits plays an important role not only in transport policy but also in environmental policy (Potter, Enoch, Rye, & Black, 2002). Company cars are cars provided by employers to employees and can typically be used for both business and private trips (Cohen-Blankshain, 2008, pp. 66–67). The provision of company cars acts as a disincentive to the use of public transport or other modes such as car-pools. Hence, from an environmental point of view, company cars should be limited (Cohen-Blankshain, 2008). Taxation can help integrate employers’ behaviour with employees’ behaviour towards a low carbon transport economy. Commuting expenses can be seen as personal expenses, subject to income taxation, or as tax deductible expenses (Potter et al., 2002). Whether commuting expenses are taxed or not can make a big difference to commuters’ incentives regarding their travel behaviour.

If travel is classified as an intermediate good, the Diamond–Mirrlees productive efficiency result applies – intermediate goods in production should not be subject to distortional taxes (Diamond & Mirrlees, 1971). Related to that is the idea that expenses incurred in earning income should not be part of taxable

55 Adverse selection refers to a situation where an individual’s demand for insurance is positively correlated with the individual’s risk and the insurance company cannot introduce that correlation in the premium. PAYD would probably be chosen by those who already drive less.

56 Their analysis is based on a logit model and takes a partial equilibrium approach which ignores international competition issues.

57 It should be noted, however, that this sub-section, dealing only with commuting and work-related trips, ignores a large proportion of trips that people make. According to the 2001 US National Household Travel Survey, for example, only 14.8 per cent of trips made were commuting trips, with an additional 2.9 per cent of trips for other work-related purposes (US DoT, 2003, p. 10).
income (Richter, 2006). According to this principle, commercial vehicles (and the fuel they consume) that are used as inputs in production should only be taxed to correct for externalities, but not for the sole purpose of raising revenue for the government. Whether a commuting trip is considered an intermediate good or not can be subject to debate. In the UK commuting trips are not considered working trips and therefore taxes on commuting trips are not refundable. Furthermore, the UK DfT (2009b) places very different values on working and commuting time.

One approach towards company cars is to differentiate taxation at the purchase and ownership level, by making excise duties refundable for commercial vehicles, even if they are going to be used for private trips as well. Fuel taxes, on the other hand, can be made refundable for working trips only (and commuting trips, depending on the individual government’s view).

This lower tax while in the course of production is also one of the rationales behind the lower taxation of diesel versus petrol. However, this may distort incentives of private individuals towards purchase and usage patterns that create more damage to the environment. Economic theory suggests that the optimal tax on a company car should be based on the total net cost for the company of providing the car to the employee. This is the case in the US but not in many other countries (Gutiérrez-i-Puigarnau & van Ommeren, 2007). The tax treatment of employer provided transport benefits in kind (such as company cars, reimbursement of commuting business related travel expenses) is an important factor directly affecting modal choice and travel behaviour at large (Knight et al., 2000). This has significant impacts on road transport emissions control, especially in countries where company car use is high (e.g. the UK) and in urban centres where travellers commute long-distances. In some countries, commuting expenses can be deducted from taxable income.

The tax on a company car can be based on the fuel economy, engine size or on the vehicle’s CO$_2$ emissions, the latter being currently the case in the UK (UK Inland Revenue website, Gross Hoptonstall, Anable, Greenacre, & E4tech, 2009, Evidence Table: Company Car Tax). Another key issue is the tax treatment of car purchase by companies, because it directly affects the incentives of the company to provide the car in the first place. The extent to which the purchase price can be treated as a capital allowance and the amount of yearly allowable writing-down against corporation tax is also important, as is the treatment of value added tax (VAT) on purchase and VAT on fuel as a recoverable or not-recoverable input tax.

Some employers provide company cars, free parking spaces, or public transport passes to the workplace (Knight et al., 2000). The tax treatment of these benefits in kind is important in strengthening or weakening these incentives on the basis of environmental considerations. Taxation of employer provided parking as a benefit would be an important tool to reduce employers’ subsidies to automobiles. However, there could be valuation issues, since some employers may provide parking on private premises that could not be profitably used for other purposes. An alternative to taxation would be the requirement of employers to provide monetary benefits (parking “cash-outs”) for employees who decide not to take advantage of the free parking places (DeCorla-Souza, 2004; Toder, 2007). Making employer provided public transport passes a non-taxable benefit could be an option to encourage modal shift towards public transport. However, this policy need not be the most cost efficient, and may raise equity issues (Di Domenico, 2006), as it targets only a (relatively wealthy) share of the population. Furthermore, under-taxation of one benefit to compensate for under-taxation of a substitute benefit need not be the right approach. Di Domenico (2006), for example, argues that correcting the under-taxation of employers’ provided parking spaces in Canada through another distortion, namely non-taxed transport passes is not the direction to take. The best approach is to appropriately tax the benefit derived from free parking provision.

Tax concessions for commuting expenses$^{58}$ are found to induce distortions in residential land use (Wrede, 2003), stimulate commuting by car and longer trips (due to distorted location choice of residence and or/working place), with negative effects on the environment (Potter et al., 2002). Non-deductibility could be justified by a lack of other instruments to internalise congestion and environmental externalities (Wrede, 2003). The literature is not unanimous on the optimal treatment of commuting expenses. Even if concessions are only targeted to public transport commuting expenses, they need not be advisable. In fact, they contribute to trip lengthening, and they impose a heavy burden on finances (Potter et al., 2002).

### 4.5. Revenue allocation

The use of revenues generated through road transport policies that correct externalities has important welfare and distributional consequences, and is ultimately one of the determinants of the political acceptability of these policies. Revenues can be redistributed as lump-sum subsidies, to counteract the regressive impacts of the policy itself. They can thus be used to cut labour/ income taxes or other distortionary taxes, or be earmarked for investment in specific sectors, in particular, the transport sector.

Earmarking revenues means that the revenues generated through a tax program will be used towards specific pre-defined projects. Earmarking can increase the political acceptability of certain policies; it can establish a link between tax and use of resources if, for example, road taxes are spent on roads projects; it is also a way to ensure a stable revenue flow to government needed for undertaking desirable governmental functions. However, earmarking revenues hampers flexibility for the government, since it may prevent adaptation to changing conditions or to a greater amount of available information. Earmarking may also allocate revenues in an inefficient way, or to projects that have lower priority (Santos, 2005). Funds may be misallocated among functions, and a lack of periodic review may leave systems in place beyond the necessary time (Knight et al., 2000). However, Bracewell-Milnes (1991) argues that in an imperfect world, earmarking can serve as an instrument for better decision-making, for instance, earmarking can solve policymakers’ time inconsistency problems to some extent (Marsiliani & Renstrom 2000).

When a government does not have any need for raising additional revenue or when it can use lump-sum taxes, economic theory suggests that taxation for environmental damage should equal marginal external costs. When the government needs additional revenues, environmental taxes may be raised above the marginal damage (Fullerton, 1997). Along these lines, West and Williams (2007) show that since petrol is a complement to leisure, the optimal petrol tax is much higher than the marginal external cost.

Early research suggested that environmental taxes can generate a “double dividend”. This refers to the idea that not only do environmental taxes correct a market failure, but they also generate revenues which can be used to reduce distortionary taxes. Pearce (1991) emphasises the role of the double dividend argument in carbon taxation policy, as ‘the corporate and public acceptability of such a tax is greatly enhanced if the tax is introduced as part of a “package” of fiscally neutral measures’ (Pearce, 1991, p. 940).

However, later analyses have refined the notion of the “double dividend” (Goulder, 1995). In particular, we can differentiate between “weak” and “strong” versions of the concept. The “weak” version, broadly supported by empirical literature, refers to the idea that shifting tax distortions toward distorting the production mix (Bovenberg & Goulder, 1997). An alternative taxation schedule consists of a combination of higher taxes on labour with spending the larger the efficiency cost on the economy of existing capacity. Pigouvian taxes close the difference between marginal private and marginal social cost. Their effectiveness depends on the strength of the link between the externality they are aimed at correcting and the tax itself.

There are many types of taxes and charges in the road transport sector. These include taxes on purchase and ownership of a vehicle and taxes on usage of a vehicle, such as emission taxes, fuel duties, distance travelled taxes, congestion and parking charges, and pay-as-you-drive insurance. Also, subsidies to efficient vehicles can be used in combination with feebottoms and/or scrappage incentives, to help consumers decide towards cleaner vehicles, even when these are more expensive than more polluting ones.

The use of revenues generated through these taxes has important welfare and distributional consequences, and may help increase public and political acceptability.

5. Taxes and charges in the road transport sector

Except for the very early attempts of credit and permit systems in the US, including the inter-refinery averaging and banking of credits instituted by the US EPA in the 1980s to facilitate the phase-down of lead in petrol, cap-and-trade has not been implemented in the road transport sector in any country or region to date. Its application has been and still is contemplated and debated, to deal mainly with congestion and emissions, and in particular with CO₂ emissions. In this section we concentrate on IE policies that have been implemented, and thus, we focus on fiscal measures and omit permits.

Taxes and charges on road transport are used extensively throughout the developing and developed world. From an economic point of view, Pigouvian fiscal measures can correct externalities and achieve efficient levels of traffic (and therefore congestion and emissions). This, however, has not historically been the reason for the introduction of taxes and charges, which existed even before any concerns regarding externalities in general, and climate change in particular, were raised. Governments need funds, not just to build and maintain roads, but also to provide other public goods and services, such as defense, education and health care. Road transport increasingly became an excellent source of revenues, regardless of any consideration of negative externalities.

Nevertheless, road taxes and charges have helped (perhaps unintentionally) to reduce excessive levels of negative externalities. This is not to say that they have been designed to reduce them or that they have reduced them to their efficient levels.

In what follows, we collate evidence of different types of road transport taxes and charges applied in different countries, their relative differences and, to the extent which is possible within a review, their impact.

5.1. Taxes on purchase and ownership of a vehicle

5.1.1. Purchase and registration taxes

All European countries levy a value added tax (VAT) on the purchase of new motor vehicles, just as they do on purchases of most goods and services. VAT rates vary between 15 per cent and 25 per cent, with Denmark, Hungary and Sweden at the higher end. Many countries also levy a registration tax, based on the pre-tax price, fuel consumption, cylinder capacity, CO₂ emissions and/or other emissions, and vehicle length (European Automobile Manufacturers’ Association, 2009a, p. 1).

Registration taxes are typically charged when the vehicle is registered for the first time. The exceptions are Belgium and Italy, where the registration tax is levied every time the vehicle changes ownership (Knight et al., 2000, p. 37). Kunert and Kuhfeld (2007, p. 309) show that registration taxes exhibit great variations across countries, with Denmark, Ireland, Malta and Norway imposing the highest registration taxes in Europe.

Ryan, Ferreira, and Convery (2009) conduct a study for the EU-15 and find that registration taxes are significant in purchasing decisions (especially regarding the decision of whether to buy a petrol or diesel car) when the specificities of each country (country fixed effects) are ignored, but they stop being significant when country fixed effects are included in the model (pp. 372–373). They also find that registration taxes are not significant in changing average vehicle CO₂ emissions (p. 372) and highlight the fact that registration taxes in the EU are levied mainly on cars and that such taxes apply at reduced rates, if at all, to commercial vehicles like buses, vans or lorries (p. 36).

59 It must be noted, however, that this result is particular to the US, and need not apply to other countries, such as the UK.

60 Fuel is also an input to production to the extent that it is used by commercial vehicles.

61 Note that, as explained in Section 1, the efficient level is not zero, but that at which marginal social costs and marginal social benefits are equal.

62 Bulgaria, the Czech Republic, Estonia, Germany, Lithuania, Luxemburg, Slovakia, Sweden and the UK do not have any registration tax (European Automobile Manufacturers’ Association, 2009a, p. 1).
The case of Singapore is worth highlighting, as it has very high ownership taxes. On top of the COE described in Section 2, motorists need to pay the Additional Registration Fee (ARF). The ARF is an ad valorem duty on a vehicle’s open market value payable by buyers of new motor vehicles, in addition to an administrative fee, referred to as the Basic Registration Fee. The ARF rate was raised through the 1970s, reaching 125 per cent in 1978 and 150 per cent in 1980 (Santos, Li, & Koh, 2004). However, as the ARF rate rose, it also discouraged existing vehicle owners from replacing their cars and encouraged new car buyers to buy used cars. Concerned with a stock of aging vehicles, when the applicable ARF rate was raised to 100 per cent in 1975, the government introduced a Preferential Additional Registration Fee (PARF) to counterbalance the disincentives on vehicle renewal. The purchaser of a new vehicle paid a substantially lower PARF rate if he de-registered an old vehicle (i.e. by exporting or scrapping it) of the same engine category at the time of his new purchase. Since 1997, the PARF has been amended to a system where the applicable discount is a function of the age of the vehicle to be de-registered.64 Vehicles older than ten years no longer qualify for PARF treatment.

5.1.2. Ownership taxes

Annual ownership taxes, sometimes also called annual circulation taxes or vehicle excise duties, are levied in most countries, usually on private and commercial vehicles. In Europe, the criteria to set these taxes vary across countries. For cars these include cylinder capacity, fuel efficiency, vehicle age, gross weight, CO2 and/or other emissions; and for commercial vehicles, which tend to be heavier and damage roads more, these depend mainly on weight and axles, although also on noise and CO2 and/or other emissions (European Automobile Manufacturers’ Association, 2009a, p. 2).

In the UK, for example, vehicle excise duties levied on cars registered on or after 1 March 2001 depend on the amount of CO2 emitted by the automobile (Driver and Vehicle Licensing Agency, 2009). Excise duties have varied over the years. The levels applying from 1 May 2009 are as follows. If the car emits less than 100 grams per km the vehicle is exempt from vehicle excise duty. Cars emitting between 101 and 120 grams per km are charged £35 (€40, $56)65 per year, cars emitting between 121 and 140 grams per km are charged £120 (€138, $193) per year, cars emitting 141–150 grams per km are charged £125 (£144, $200) per year, and cars emitting 151–165 grams per km are charged £150 (£173, $241) per year. The charge continues to increase with the CO2 emitted per km and reaches a maximum of £400 (£461, $642) per year when CO2 emissions exceed 226 grams per km (Driver and Vehicle Licensing Agency, 2009).

Some European countries, however, do not levy any annual tax. These include the Czech Republic, Estonia, France, Lithuania, Poland, Slovenia and Slovakia (European Automobile Manufacturers’ Association, 2009a, p. 2).

Ryan et al. (2009, p. 373) find that annual ownership taxes, in contrast with registration taxes, show a strong impact on total new car sales in Europe. Consumers seem to be more sensitive to taxes they will pay every year for as long as they own the vehicle than to one-off registration taxes.

5.1.3. Subsidies to efficient vehicles and feebates

A showcase example of this type of subsidies is the ‘ecoauto Rebate Program’, which was run in Canada between March 2007 and March 2009. The idea behind it was to encourage Canadians to buy new fuel efficient vehicles.

Eligible vehicles included cars with fuel efficiency of 6.5 litres per 100 km or better, vans with fuel efficiency of 8.3 litres per 100 km or better, and flex-fuel vehicles, running on a combination of petrol and ethanol, with a combined fuel consumption rating of 13 litres per km or better (Banerjee, 2007, p. 2).66

During the two years that the program operated, applicants who purchased or leased (12 months or more) eligible 2006, 2007 and 2008 model-year, fuel efficient vehicles, could apply for and receive rebates ranging from CA$1000 to CA$2000 (£502–£1004 or €655–€1310 or $925–$1850).67 By the time the program finished it had issued over 167,000 rebates, amounting to CA$187.7 million (Transport Canada website).

The rebate was combined with a tax on inefficient vehicles, the Green Levy, which started at CA$1000 for vehicles with fuel (in)efficiency of between 13 and 14 litres per 100 km and increased in CA$1000 steps for every litre in consumption up to 16 litres per 100 km, at which point the tax was capped and all vehicles using 16 litres per 100 km or more paid a maximum tax of CA$4000 (Banerjee, 2007, p. 2).

The impact of the program on new vehicle purchases and emissions has not been published by the Canadian Government yet, and therefore it is difficult to assess its success.

5.1.4. Scrappage incentives

A number of European countries have scrappage schemes in place. In December 2007 France introduced a system of bonuses and penalties on the purchase of new vehicles with a bonus between €200 and €5000 on vehicles emitting less than 130 grams of CO2 per km and penalties between €200 and €2600 for vehicles emitting more than 160 grams of CO2 per km. The system also included a “super-bonus” - a scrappage scheme offering an additional incentive of €300 for scrapping a car more than fifteen years old. In December 2008, as part of an economic stimulus program, the policy was reviewed, increasing the “super-bonus” to €1000, and extending it to include the scrapping of cars more than ten years old and the purchase of new vehicles emitting less than 160 grams of CO2 per km. The maximum total bonus under the revised scheme is €2000, which would be attained by replacing an old vehicle by one emitting less than 100 grams of CO2 per km (French Ministry for the Environment, Energy, Sustainable Development and the Sea website). In Germany the incentive is €2500 for scrapping cars older than nine years and buying new fuel efficient cars, which are required to satisfy the Euro 4 criteria on emissions (German Federal Ministry of Economics and Technology, 2009).

The scrappage scheme in the UK was launched on 18 May 2009 and will last until the end of February 2010, or until the £300 million (€346 m; $482 m) funding is exhausted (if this occurs first), offers an incentive of £2000 (£2307; $3212) to scrap a car over ten years old and buy a new one, where £1000 of this incentive is provided by the government and £1000 from the vehicle manufacturer. The scheme, not specifying any restrictions on the new

63 The open market value is essentially similar to Cost plus Insurance and Freight (CIF). It includes purchase price, freight, insurance and all other charges incidental to the sale and delivery of the car to Singapore.

64 This is a common feature with scrappage incentives, discussed in Section 5.1.4.

65 The average exchange rate in the period May-July 2009 was £1 = €1.15 = $1.6 (IMF Exchange Rate Query Tool).

66 These numbers are equivalent to 15.4 km per litre for cars, 12 km per litre for vans, and 7.7 km per litre for flex-fuel vehicles. The only eligible vehicles for the rebate in Canada, however, were essentially either hybrid or compact, or running on a combination of petrol and ethanol.

67 The average exchange rate in the period Mar 2007 - Mar 2009 was CA$1 = £0.5 – €0.65 – $0.93 (IMF Exchange Rate Query Tool).
car’s CO₂ emissions, is mainly intended to provide a short-term boost to the demand for cars (UK Department for Business, Innovation and Skills website).

Other European countries, including Austria, Cyprus, Italy, Luxembourg, Portugal, Romania, Slovakia, Spain and The Netherlands have also introduced scrappage schemes (European Automobile Manufacturers’ Association, 2009b, website). Although many of the scrappage schemes in Europe specify conditions regarding CO₂ emissions, the main reason for the proliferation of these incentives is the economic recession and its impact on the car industry.

While many countries in Europe have been keen to adopt scrappage schemes, these schemes have not been limited to Europe. In the US a scrappage scheme was introduced with the signing into law of the Consumer Assistance to Recycle and Save Act of 2009, by President Barack Obama in June 2009 (US NHTSA website c). The Car Allowance Rebate System (CARS),68 is an incentive scheme which aims to both stimulate car and lorry sales while also removing older and less fuel efficient vehicles from the roads. The scheme offers an incentive of either $3500 (£2141; $2588)71 per vehicle (Japan Automobile Manufacturers’ Association website c).

The Japanese government’s stimulus package approved by the Japanese Cabinet in April 2009 includes a scrappage scheme as well as incentives for the purchase of environmentally friendly vehicles without scrapping requirements. The vehicle replacement incentive scheme applies to cars and mini-vehicles as well as lorries, requiring the scrapped vehicles to be at least 13 years old. For cars in the standard and small categories the subsidy upon replacement of an old car by one meeting the 2010 fuel efficiency standards is ¥250,000 (£1647; €1884; $2588)71 per vehicle (Japan Automobile Manufacturers’ Association website).

5.2. Taxes on usage of a vehicle

5.2.1. Emission taxes

As discussed in Section 4.2.1, although these would be the first best instrument to internalise CO₂ emissions from road transport, direct charging for emissions is not feasible under current technology due to a lack of cost effectiveness and impracticability of monitoring techniques.

There is a very good proxy for emission taxes, however, that would be virtually as effective: a carbon tax. This would be a tax on the carbon content of the fuel in question or on the estimated CO₂ emitted in the fuel combustion process. The idea of a carbon tax, however, has frequently encountered public and political opposition, even more so than the idea of a cap-and-trade system. In 1992, 1995 and 1997 the European Commission advanced proposals for a European carbon tax (Vehmas, Kaivo-Oja, Luukkanen, & Malaska, 1999, p. 344). These, however, failed due to strong opposition from industry and several member states (Richardson, 2002, p. 194).

There are some countries in Europe that have implemented a carbon tax, but the efforts have never been coordinated or agreed at EU level. These countries are Finland, which introduced a carbon tax in 1990, Sweden and Norway, in 1991, the Netherlands in 1992, Denmark in 1993 (Richardson, 2002, p. 194), and Italy, in 1999.

Although most countries argue that these carbon taxes have decreased CO₂ emissions to some extent (Vehmas, 2005, p. 2180), looking at Fig. 3 it is evident that the carbon tax component of the petrol and diesel duties have not made them significantly different from those applied in other countries.

In July 2008 British Columbia in Canada implemented a carbon tax (British Columbia Ministry of Small Business and Revenue, 2008). It started at CA$10 (£5; €6.4; $9.4)72 per tonne of carbon dioxide equivalent (CO₂e)73 and will rise by CA$5 per year reaching CA$30 in 2012. The petrol tax will then reach 7.24 Canadian cents per litre (British Columbia Government website), equivalent to 4.6 Euro-cents, which represent a very small increase relative to the current tax component of 22 Euro-cents shown on Fig. 2.

5.2.2. Fuel taxes

Although fuel taxes are in place in most countries, they vary widely from one to another. In fact, no other product seems to be subject to such divergent treatment (Gupta & Mahler, 1995, p. 101).

Fig. 2 shows unleaded petrol taxes and prices for OECD countries during the third quarter of 2008 in € per litre.

As it can be easily seen from Fig. 2, Mexico and the US have remarkably low petrol taxes, when compared to the rest of the OECD countries. Unsurprisingly there is not that much difference in the pre-tax price across countries, except, as expected, for Mexico. There is, however, considerable variation regarding taxes.

Fig. 3 shows fuel duties for the same countries, expressed as percentage of pre-tax price. It should be noted that these shares change with the pre-tax price, increasing when oil prices decrease and decreasing when oil prices increase (Newbery, 2005a, p. 6).

Most countries have fixed petrol and diesel taxes, rather than ad valorem. The advantage of this practice is that stable revenues are guaranteed regardless of world oil prices. In order to keep a constant revenue flow, governments would need to adjust ad valorem taxes every time there was an important change in oil prices. Gupta and Mahler (1995, p. 103) point out that ‘if the international price of the product is subject to wide variations, the quantities of petroleum products consumed may be more stable than the value of the petroleum product consumed’ and in that case revenues from a specific tax would be more predictable and stable, while revenues from ad valorem taxes would be more elastic. On the other hand, ad valorem taxes are automatically adjusted with inflation whereas fixed rates need to be adjusted to keep up with changes in domestic prices and exchange rates, if the real value of taxes is not to be eroded through inflation (Smith, 2006). In the EU inflation and exchange rates are not volatile, and hence the preferred practice of fixed fuel duty rates does not pose the inconveniences it may pose in developing countries.

Issues related to inflation and exchange rate volatility, however, can be a problem in African countries, for instance. Metchies (2001, in Smith, 2006) argues that in countries where the domestic currency is substantially devalued, regulated fuel prices

68 In the US this type of scheme is usually referred to as Cash for Clunkers.
69 The average exchange rate in the period June-July 2009 was $1 = €0.65 = £0.7 (IMF Exchange Rate Query Tool).
70 The requirements for work lorries are different and the CARS Act limits the amount of funds that can be used to provide credits for purchases or leases of work lorries to 7.5 per cent of the funds available for the program.
71 The average exchange rate in the period April-July 2009 was £1 = ¥152 = €1.14 = $1.57 (IMF Exchange Rate Query Tool).
72 The average exchange rate in the period Jan-Dec 2008 was CA$1 = £0.51 = €0.64 = $0.94 (IMF Exchange Rate Query Tool).
73 Using a stoichiometric conversion factor of 44/12 = 3.6667, this is equivalent to CA$2.23 per tonne of carbon.
and taxes) fall relative to world prices, and the government loses real revenues, effectively providing a subsidy to drivers.

Fig. 4 shows fuel duty as a percentage of pre-tax fuel prices for some selected Sub-Saharan countries. 1991 is the last year for which data was available. For poor and/or developing countries, it is not unusual to find contradictory information in different sources regarding fuel tax rates.

From Figs. 3 and 4, it is puzzling that countries so close to each other geographically and sometimes members of the same economic community, have such variations in fuel duty rates across them.

Newbery (2005a, p. 6) points out that the European Commission aims at harmonising energy taxes within the EU and that a European approach to reducing emissions requires that each country and GHG source face the same charge per tonne of carbon. The argument is perfectly reasonable from an economic point of view, since, as explained in Section 1, marginal abatement costs need to be equalised across polluters to achieve efficiency. The EU therefore faces the challenge of getting all its member states to agree on a uniform rate of petrol and diesel taxes.

The next question is what this uniform rate ought to be. Should it be at the lower end, like the one applied in Greece, or at the higher end, as in the UK? If the revenue raising component of fuel taxes is the VAT, the fuel duty is left to cover the external costs from road transport that may be deemed worth internalising through fuel duties.

Parry and Small (2005) develop a model that estimates the optimal fuel tax and they calibrate such a model for the US and the UK. The externalities they include are damage from CO2 emissions and air pollution, congestion and accidents. They conclude that for the year 2000 the optimal petrol tax in the US would have been $1.01 per US gallon (26 cents, 28 Euro-cents, 17 pence per litre),74 more than twice the actual rate for that year, and the optimal petrol tax in the UK would have been $1.34 per US gallon (35 cents, 38 Euro-cents, 23 pence per litre), slightly less than half the rate for that year (p. 1283). Newbery (2005b, Table 7.3, p. 217) finds that the external costs of road damage,75 air pollution, global warming, water pollution and noise in the UK in 2000 amounted to 36 pence per litre of unleaded petrol for the third quarter of 2008 is equivalent to 50 cents per gallon. Source: IEA (2008), Fig. 8, p.xxxiv.

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74 The average exchange rate in the period Jan-Dec 2000 was $1 = €1.08 = £0.66 (IMF Exchange Rate Query Tool).
75 Newbery (2005b) points out that the damage per litre of fuel varies across vehicle types and ages, and although the vehicle excise duty can be fine-tuned to allow for those differences, fuel duties cannot.
petrol. Since the fuel duty in the UK for the year 2000 was 48.8 pence per litre (UK DfT, 2008, Table 3.3, p. 53) it can be concluded that the fuel duty more than covered those externalities.

Both Newbery (2005b) and Parry and Small (2005) conclude that the UK had a fuel duty that more than covered the environmental costs of petrol. The numbers could be easily updated to 2009 and the same conclusions would be reached.

This is the kind of analysis that would need to be done when trying to agree on a harmonised European rate of fuel duties.

Smith (2006) highlights an additional problem resulting from such large differences in fuel duties across African countries. He explains that if neighbouring countries in Africa have very different fuel taxes, this can lead to problems of smuggling and some diversion of activity. Harmonisation even in poor countries makes sense, if only to address the problems of smuggling, reduced fuel tax revenues, and diversion of transport routes and economic activity to regions where taxes are lower.

It is worth keeping in mind that in the UK, and probably in other countries with comparable fuel tax rates, the global warming externality seems to have already been internalised, which would make substantial increases in tax rates inefficient from an economic point of view. CO₂ emissions from road transport in these countries are probably lower than they would have been had no such high taxes been in place (Sterner, 2007), but they still seem to be too high to meet the various commitments that different governments have adhered to. The only way to defend higher fuel tax rates would be to use a much higher shadow price of carbon, and this in turn would need to be justified by the type of models mentioned in Section 1. The effects from climate change can only be modelled (not predicted with certainty) and this modelling process involves making a large number of assumptions regarding possible impacts and adaptation, discount rates and other parameters.

It is also worth bearing in mind that there are countries with significant scope to increase fuel taxes, especially when the idea of harmonisation is brought into the discussion.

5.2.2.1 Fuel duty differentials

5.2.2.1.1 Diesel vs. petrol taxation. It is evident from Figs. 3 and 4 that diesel is often favoured by governments relative to petrol. The reasons for and against this were discussed in Section 4.2.2.1.

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76 Note that Newbery (2005b) excludes accidents and congestion from these calculations.
Newbery (2005a, p. 29) points out that similar tax rates on petrol and diesel can be compensated with differentiated vehicle excise duties, by charging higher duties to heavy goods vehicles, which cause more road damage and run on diesel.

Ryan et al. (2009) find that when diesel fuel prices and diesel vehicle taxes are high, the share of petrol vehicles increases, although these impacts disappear when the specificities of each country are controlled for. The reason for this, they argue, might be that fuel and vehicle taxation are political decisions and, as a result, the favouring or not of diesel cars and fuels is determined by country and is a fixed country effect (p. 371, p. 373).

Whatever the reason for governments to favour diesel in their tax policy, diesel passenger car market penetration in the EU was 40 per cent in 2003, with Austria and France reaching 60 per cent (Zervas, Poulopoulos, & Philippopoulos, 2006, p. 2848). The percentage penetration for that year was at least 30 per cent in every EU-15 country, with the exception of Ireland, Finland, Sweden and Greece where it was 17 per cent, 13.6 per cent, 7 per cent and 1 per cent, respectively (Zervas et al., 2006, p. 2850).

5.2.2.1.2. Cleaner fuels. Governments can use taxes to incentivise motorists to switch to cleaner petrol, when this becomes available in the market. The main economic advantage of this type of taxes is that they leave the user to decide how best to respond, rather than forcing him to choose one fuel. In the UK for example, the higher tax on leaded than on unleaded petrol, raised the proportion of motorists buying unleaded petrol from five per cent in 1988 to 63 per cent by 1993 (Newbery & Santos, 1999, p. 117). Similarly, in Sweden the tax differential between leaded and unleaded petrol decreased the share of leaded petrol from 70 per cent in 1986 to practically zero in 1994 (Ekins & Speck, 1999, p. 384).

A few years later, the differential duty on unleaded petrol and ultra-low sulphur petrol followed in the UK. In October 2000 the fuel duty on ultra-low sulphur petrol, which at the time was difficult to find at petrol stations, was cut down by 1 penny per litre. In March 2001 it was further cut down by two pence. The pump prices were 78.2 and 75.9 pence per litre (UK DfT, 2008, Table 3.3, p. 53). At that point most drivers switched from unleaded petrol to unleaded ultra-low sulphur petrol, and by 2006 unleaded petrol was eventually phased out.

5.2.3. Vehicle km travelled taxation

Switzerland introduced a Heavy Vehicle Fee (HVF) in January 2001, which applies to lorries over 3.5 tonnes. The HVF varies with their gross weight, kilometres driven within Switzerland, and emission category of the model (Suter & Walter, 2001, p. 383).

The fee is collected with the help of on-board units, which record distances driven and routes taken. At the end of each month the data are transmitted to the Swiss Customs Agency either by mail or over the Internet. This information is used to generate a bill that is sent to the vehicle owner.

As a result, the total volume of goods transported by road through the Alps increased by 3 per cent but lorry trips declined 14 per cent between 2000 and 2005, indicating that pricing encourages more efficient use of lorry capacity.

5.2.4. Congestion charges

Although congestion charging has been advocated by transport economists for many decades its implementation has been limited. The main barriers are typically public and political opposition, linked to equity concerns. As a result, there are only a few examples
of congestion charging as of 2009. These are briefly described below.

5.2.4.1. Norwegian toll rings. The Norwegian toll rings, very often cited in the road pricing literature as examples of congestion charging, were designed to generate revenues to finance infrastructure. The aim was not to manage traffic demand. Nonetheless, since it has become common practice to bring them into any discussion on congestion charging, they deserve a mention. Since the late 1980s - early 1990s a number of towns in Norway, including Oslo and Bergen amongst others, have tolls, usually surrounding the whole town rather than the city centre, with daily charges which never exceed NOK 20 (roughly £2 or €2.2)\(^\text{77}\) for cars and light vehicles. All schemes in Norway have flat rates 24 h a day every day, except for Namsos, where the scheme only operates Monday to Friday, from 6:00 a.m. to 6:00 p.m. Since the original aim of these toll rings was not to reduce traffic levels and congestion, the decrease in demand for car travel has been low, with estimates varying from zero to 10 per cent reduction at most. Similarly there have been no significant changes in private car occupancy rates or demand for public transport (Ramjerdi, Minken, & Østmoe, 2004).

5.2.4.2. Toll highways. There are a number of toll highways around the world. Although the only objective of many of these schemes is to generate revenue, some also aim at relieving congestion. Examples include the M6 Toll in England, the 407 Express Toll Route (ETR) in Toronto and a number of roads in major Australian cities, such as for example, City Link in Melbourne and the Westlink M7 Toll Road in Sydney. In all these cases the toll highways are in privately owned or managed. Drivers have the option of choosing between the toll road with lower journey times and the publicly provided alternative with higher journey times.

The M6 Toll in England is a parallel\(^\text{78}\) segment to the M6 motorway, which extends 27 miles (43 km). Drivers have the option of using the publicly provided alternative for free or using the toll road. The M6 toll runs from junction 3a on the M6 and rejoins it at junction 11a. Charging is done at toll plazas along the road or at the exit and is not electronic. The Highway 407 ETR in Toronto extends 67 miles (108 km) from Brock Road in Pickering in the east to the QEW/403 interchange near Hamilton in the west. The 407 ETR charges tolls electronically, based on distance driven. The City Link in Melbourne is a toll road in the centre of Melbourne in Australia, which extends 14 miles (22 km), from Tullamarine Freeway to the West gate Freeway and the West Gate Freeway to the Monash Freeway. The system operates electronically and charges per trip made along the toll segment. The Westlink M7 Toll Road in Sydney extends 35 miles (40 km), connecting the M2, M4 and M5 motorways. It operates electronically and charges per distance driven.

5.2.4.3. High occupancy toll lanes in the US. High Occupancy Toll (HOT) lanes in the US are lanes where tolls are applied on low occupancy vehicles wanting to use lanes which are free to use for high occupancy vehicles (HOV). High occupancy is usually defined as vehicles with two or more occupants.

\(^\text{77}\) The average exchange rate in the period Jan-May 2009 was NOK 1 = £0.10 — €0.11 (IMF Exchange Rate Query Tool).

\(^\text{78}\) It is not parallel in the strict sense, as it arcs around the north-east of the West Midlands conurbation before re-joining the M6.

The State Route 91 (SR-91) Express Lanes, which opened in December 1995, were the first practical example of congestion pricing in the US (Sullivan & El Harake, 1998). Although they were originally privately operated, in January 2003 their operation was taken over by the Orange County Transportation Authority. The SR-91 Express Lanes, which extend 10 miles (16 km) operate between the Orange/Riverside county line and the Costa Mesa Freeway (SR-55) interchange in eastern Anaheim. Tolls apply 24 h a day, 7 days a week, with the lowest one being $1.25 and the highest one, $9.50 (Orange County Transportation Authority, 2008).

As of 2009 there are an additional seven HOT lane projects in operation in the US, which have been partly funded by the Value Pricing Pilot program or by its predecessor, the Congestion Pricing Pilot Program. Projects include segments of the I-15 in San Diego, California (implemented in 1996), the I-25 in Denver, Colorado (implemented in 2006), the I-394 in Minneapolis, Minnesota (implemented in 2005), the Katy Freeway (I-10) and the US 290 in Houston, Texas (implemented in 1998 and 2000 respectively), the I-15 in Salt Lake City, in Utah (implemented in 2006), and the SR 167 in King County, Seattle, Washington (implemented in 2008). The individual designs vary, and tolls range from 50 cents to $9. In some cases tolls apply in the morning peak, in others in the afternoon peak, and in others they change in real time with traffic demand. In this case, drivers are informed of the toll rate changes through variable message signs located in advance of the entry points. An advantage of HOT lanes over other congestion charging systems is that with HOT lanes drivers ‘can choose between meeting the vehicle occupancy requirement or paying the toll in order to use the HOV lane’ (DeCorla-Souza, 2004, p. 288).

5.2.4.4. Singapore. In 1975, congestion pricing was implemented in Singapore. The system was a paper-based area licensing scheme (ALS). Vehicles had to purchase a licence and display it on their windscreen before entering the restricted zone (RZ). The charge was per day, not per entry, meaning that they could enter and leave the RZ an unlimited number of times during the day. Exemptions included police cars, ambulances, fire engines and public transport buses. Motorcycles, goods vehicles and car parks were initially exempted, but from 1989 onwards they were also required to buy a licence (Chin, 2002). No discounts or exemptions were given for residents living inside the RZ, although driving inside the zone without crossing the boundary could be done for free. The hours of charging and the levels of the charges varied throughout the years. They started with the morning peak Monday to Saturday, but later included the inter-peak period, from mid-morning till middle afternoon, and the evening peak. Drivers could purchase a weekly licence, which allowed them to drive inside the RZ at any time during the hours of operation of ALS, or a part-day licence, which only allowed them to drive during the inter-peak hours on weekdays and the post-peak period on Saturdays.

The system was manually enforced by enforcement officers standing at the boundaries of the RZ, and was thus prone to error. The impacts on traffic were drastic. Phang and Toh (1997, p. 99) report that the introduction of ALS increased average speeds from 11.8 to 22.4 miles (19–36 km) per hour, exceeding the government target of 12.4–18.6 miles (19–30 km) per hour. According to Willoughby (2000, p. 10), traffic volumes during the morning peak hours fell by 45 per cent, well above the expected reduction of 25–30 per cent. Car entries were reduced by 70 per cent. This project was for many years the only congestion charging project in the world. It has to be borne in mind that Singapore is a very special case, and as such, replication of the ALS elsewhere would not have been straightforward. Geographically, it is an island...
city-state, which measures 26.1 miles (42 km) East to West and 14.3 miles (23 km) North to South. It has 1955 miles (3149 km) of roads for a population of about 4.2 million people and 707,000 registered motor vehicles in the year 2002 (Santos et al., 2004). Politically, there is a dominating political party, the People’s Action Party, which has won all the elections since 1959.

In June 1995, a paper-based Road Pricing Scheme (RPS), operating in the same way as the ALS, was implemented on an expressway (East Coast Parkway). This was later extended to other expressways. The aim of the RPS was to reduce congestion on the expressways during morning peak times, and to familiarise Singaporeans with both linear passage tolls and road charging outside the CBD (Goh, 2002).

In September 1998 Electronic Road Pricing (ERP) replaced the ALS. Rather than a licence to use the RZ, charges apply per-passage. The charging area is divided into central business districts (including the areas previously covered by ALS), where charging applies from 7:30 a.m. to 8:00 p.m., and expressways/outdoor ring roads, where charging applies from 7:30 to 9:30 a.m., Mondays to Saturdays, except public holidays. Vehicles are charged automatically on an electronic card, which is inserted in an In-vehicle Unit, each time the vehicle crosses a gantry. If the charge cannot be deducted from the card, either because it is not properly inserted or because it does not have sufficient credit, a fine is issued to the vehicle owner.

The only exemptions include emergency vehicles, such as ambulances, fire engines and police cars. ERP rates vary with vehicle type, time of day and location of the gantry. Charges for passenger cars, taxis and light goods vehicles for example vary between $0.50 and $3.50, charges for motorcycles vary between $0.25 and $2, charges for heavy goods vehicles and light buses vary between $0.75 and $6, and charges for very heavy goods vehicles and big buses vary between $1 and $8.

In February 2003 a graduated ERP rate was introduced in the first five minutes of the time slot with a higher charge in order to discourage motorists from speeding up or slowing down to avoid higher charges (Land Transport Authority website). Since the graduated rate is introduced in the more expensive band, drivers save some money. For example, where the charge for passenger cars would be $2 between 8:00 and 8:30 a.m. and $3 between 8:30 and 9:00 a.m., it is now $2 between 8:05 and 8:30 a.m., $2.50 between 8:30 and 8:35 a.m., and $3 between 8:35 and 8:55 a.m., when it changes to $2.

The Singaporean ERP is the most fine-tuned road pricing system in the world to date. Since charges vary with vehicle type, time of day and location of the gantry and are only debited per passage, they incorporate a fair degree of differentiation.

5.2.4.5. London. The introduction of congestion charging was a central part of Ken Livingstone’s manifesto for the mayoral election in May 2000. After a number of public consultations, the London Congestion Charging Scheme (LCCS) was implemented in February 2003. All vehicles entering, leaving, driving or parking on a public road inside the Charging Zone (CZ) between 7:00 a.m. and 6:00 p.m.80 Monday to Friday, excluding public holidays, must pay the congestion charge. This was initially £5, but in July 2005 it was increased to £8 per day. Payment of this charge allows road users to enter and exit the CZ as many times as they wish to and drive inside the CZ as much as they want on that day.

The CZ is relatively small. It covers roughly 15 mi² (39 km²), representing 2.4 per cent of the total 617 mi² (1579 km²) of Greater London. No charge is made for driving on the roads that limit the CZ and there are two free corridors: one north to south, crossing roughly in the middle of the CZ, and another one north-west of the zone, east to west, as the diversion route would have been too long for drivers just wanting to cross a short segment of an A-road81 that falls inside the CZ.

The charging zone is set to shrink. The new Mayor of London, Boris Johnson, who took post in May 2008, conducted a public consultation on the Western Extension between September and October 2008, giving the public and stakeholders the options of keeping it, removing it or altering it. Following this consultation, in November that same year, he announced that the Western Extension would be removed, although not earlier than 2010 (Transport for London, TFL, 2008b). When this eventually happens, the new CZ will be the original one, only 8 mi² (21 km²), or 1.3 per cent of Greater London.

There are a number of exemptions and discounts, which apply to two-wheelers, emergency vehicles, vehicles used by or for disabled people, buses, taxis and mini-cabs, some military vehicles, alternative fuel vehicles (with stringent emission savings), roadside assistance and recovery vehicles. Finally, vehicles registered to residents of the CZ are entitled to a 90 per cent discount when buying at least a week worth of congestion charge. Enforcement is undertaken with Automatic Number Plate Recognition (ANPR), and violators (vehicles with number plates using the CZ without having paid the charge) are fined.

Both the number of vehicles with four or more wheels entering the original CZ as well as the number of vehicle-kilometres driven by vehicles with four or more wheels inside the original CZ during charging hours decreased by around 20 per cent, with most of this reduction having taken place in the first year of the LCCS and maintained throughout (TFL, 2007, p. 17 and Table 2.4, p. 26, TFL, 2008a, p. 41).

Congestion, defined as ‘the difference between the average network travel rate and the uncongested (free-flow) network travel rate in minutes per vehicle-kilometre’ (TFL, 2003, Table 3.1, p. 46), decreased by 30 per cent in the first year of the LCCS. This reduction in congestion deteriorated over the period 2005–2007, when average delays eventually went back to pre-charging levels, despite the decrease both in the number of vehicles entering the original CZ and in the number of vehicle-kilometres driven by vehicles with four or more wheels having stayed constant between 2003 and 2007. TFL gives a number of reasons explaining the increase in congestion, including a high number of road works, particularly in the second half of 2006, as well as during 2007 and into 2008 (TFL, 2007, point 3.2, p. 35 and point 3.10, p. 45), traffic management programs to reduce the number of road traffic accidents, improved bus services, and a better environment for pedestrians and cyclist (TFL, 2007, p. 2). In other words, part of the newly recovered network space was allocated to users other than private car drivers.

The two main modes of public transport in London are buses and the underground. The LCCS had impacts on bus use but not on underground use. The number of bus passengers entering central London increased by 18 per cent and 12 per cent respectively during the first and second years of the LCCS, and have since settled (TFL, 2007, p. 58, TFL, 2008a, p. 86). By the time congestion charging started, bus services had been improved by a combination of more frequent services, new and altered routes, and bigger buses.

The LCCS is an unsophisticated flat charge, which does not differentiate by vehicle type or time of day. However, it achieved its

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79 The average exchange rate in the period Jan-May 2009 was S$1 = £0.46 = €0.51 (IMF Exchange Rate Query Tool).

80 The end time was originally 6:30 p.m. but it was brought forward to 6:00 p.m.

81 An A-road in the UK is a main road.
objective of reducing car use. Although speeds increased in the first two years, they then started to deteriorate and by 2007 delays were back at pre-charging levels. The decrease in average speeds however, is not linked in any way to an increase in traffic but rather, to a reallocation of network space to buses, cyclists and pedestrians, plus the unfortunate timing of road works in central London.

5.2.4.6. Stockholm. The congestion tax in Stockholm was implemented in August 2007 with the objectives of reducing traffic congestion and emissions. It is a cordon toll system, with a cordon that surrounds the entire Stockholm City, which has a total area of roughly 13.7 mi² (35.5 km²). Like in London, enforcement is undertaken with ANPR, with cameras located at each of the 18 entry and exit points.

Each passage into or out of the area surrounded by the cordon costs SEK 10, 15 or 20 (roughly between £0.8 and £1.7 or €0.9 and €1.8) depending on the time of day. The accumulated passages made by any vehicle during a particular day are aggregated and the vehicle owner is liable for either the sum of the charges or SEK 60, whichever is lower.

Exemptions include emergency vehicles, buses, diplomatic cars, motorcycles, foreign registered vehicles, military vehicles, disabled parking permit holders, and vehicles that according to the Swedish Road Administration’s vehicle registry, are equipped with technology for running (a) completely or partially on electricity or a gas other than LPG or (b) on a fuel blend that predominantly comprises alcohol (Swedish Road Administration, 2007). In addition, vehicles driving on the motorway which goes through the charging zone are also exempt, as there are no alternative routes. Finally, traffic to and from Lidingö island, with its only access to the mainland through the charging zone, is also exempt.

The congestion tax is paid retroactively. The payment must reach the Swedish Road Administration within 14 days of crossing the cordon. Regular users can set up a direct debit. Other ways of paying include cash or credit card at convenience stores, credit or debit cards on the congestion tax website, or directly to the Road Administration’s account (Swedish Road Administration, 2007). Drivers not paying the tax within 14 days are issued a fine. Eventually, if the vehicle owner continues to refuse to pay, his name is entered in the Enforcement Register (Swedish Road Administration, 2007).

5.2.5. Parking charges

Parking charges can be divided into three groups: parking charges for using a space on a public road, parking charges for using a space in a privately provided parking lot, and parking charges for parking at the workplace.

Except for the charges paid in privately provided spaces, which typically cover all costs of parking and even yield some profit to the owner, the other charges tend to be low or non-existent. Zatti (2004) discusses the effectiveness of the parking charges in the city of Pavia (Italy). In Pavia, the parking charges are limited to a small area, the charges are modest, and many categories of drivers are exempt from payment. He argues that these factors, together with the vast use of illegal parking contribute to making parking fees in the city of Pavia a revenue instrument rather than an instrument to internalise externalities. In fact, the majority of the burden falls disproportionately on occasional visitors to the city.

The Transport Act 2000 (Acts of Parliament, 2000) gave local authorities in England and Wales powers to introduce workplace parking levies. However, as of August 2009, Nottingham City Council is the only local authority to have concrete proposals for a workplace parking levy, and the scheme will not start until specific regulations regarding workplace parking charges have been detailed (Nottingham City Council website). All employers with more than ten parking spaces would be liable and the Council hopes that the policy will reduce peak-time congestion and encourage the use of an improved public transport system.

5.2.6. Pay-as-you-drive insurance

Parry et al. (2007) report that PAYD schemes are slowly emerging at state level in the US. Oregon has offered insurance companies a state tax credit of $100 per motorist for the first 10,000 motorists who take up a PAYD policy and Texas has passed legislation which allows insurance companies to offer PAYD (p. 394).

In May 2009 the New America Foundation Climate Policy Program published a survey of 33 US states climate change emission reduction plans (New America Foundation website). Twelve include PAYD as a transport emission reduction strategy. These states include Arizona, California, Colorado, Maryland, Maine, Minnesota, New Hampshire, New Mexico, North Carolina, Rhode Island, Vermont and Virginia. The degree of emphasis and support for PAYD varies. For example, the Rhode Island plan, while endorsing the importance of the strategy, explicitly says that the state will likely wait for other states to figure out how to promote PAYD insurance before it does so. The California Department of Insurance is now in the process of rule making to support PAYD (California Department of Insurance website).

There are some limited examples in other countries, with some companies offering PAYD in Australia, Israel, the Netherlands and South Africa. It is not a widely spread practice yet, and so there are not many implementation examples to report on or assess.

5.3. Revenue allocation

As already explained in Section 4.5, revenues from taxes can be earmarked for specific uses, such as for example, the road transport sector. Thus, road infrastructure and public transport and/or other environmentally friendly transport projects can be financed with a stable flow of revenues from ring-fenced taxes or from the general government funds. Many governments depend on fuel and other road taxes for general revenues. Fuel taxes are relatively easy to administer and collect and so it is difficult for governments to let go.

The US, however, allocates most of the fuel tax revenues to expenditure on building and maintaining the national highway system. Although this used to be enough in the past, it only covers about one third of total expenses now, and the gap is financed with real estate and other taxes (Schäfer et al., Chapter 7, p. 224). Fuel taxes in the US are lower than in other developed countries, as was shown in Section 5.2.2. About three-quarters of the revenue goes to the highway account, a small portion is allocated to public transport and only the residual goes to the general budget.

In the UK, fuel duties and vehicle excise duties, exclusive of VAT receipts, amounted to almost £26 (€28.6) billion in 2002–2003, or 6.7 per cent of total tax revenue, around 2 per cent of GDP. Only £5.5

84 The excel sheet summarising the plans is available on www.newamerica.net/files/State%20Climate%20Policy%20Tracker%205-4-09.xls

82 The average exchange rate in the period Jan–May 2009 was SEK 1 = 8 pence = 9 Euro-cents (IMF Exchange Rate Query Tool).

83 Under the 1961 Vienna Convention, diplomats are exempt from paying taxes. Staff at the US and German embassies refuse to pay the congestion charge in London, arguing that under the 1961 Vienna Convention, they are exempt from paying taxes. However, the London congestion charge is not a tax but a charge. The Stockholm congestion tax on the other hand is a tax.
On the other hand, do not need to be specifically earmarked, as they specifically earmarked for some use. The revenues from a charge, return to the individual unit making the payment, or they can be fund, as the government does not need to provide anything ‘in return’ to the individual unit making the payment. (OECD, 1993: System of National Accounts 1993).

Different challenges and opportunities. It is now also worth highlighting the difference between a charge and a tax. According to the System of National Accounts 1993 (OECD, 1993):

‘Taxes are compulsory, unrequited payments . . . to government units. They are described as unrequited because the government provides nothing in return to the individual unit making the payment, although governments may use the funds raised in taxes to provide goods or services to other units, either individually or collectively, or to the community as a whole’ (OECD, 1993, p. 46).

In other words, the revenues from a tax can go to the general fund, as the government does not need to provide anything ‘in return to the individual unit making the payment’, or they can be specifically earmarked for some use. The revenues from a charge, on the other hand, do not need to be specifically earmarked, as they already are, ipso facto.

While the Stockholm congestion tax has been classified as a tax, the London and Singapore congestion charges have been classified as charges, not taxes. All three schemes, however, generate revenues which are allocated to the road transport sector. In the case of Stockholm, these have been earmarked for new road construction in and around Stockholm and for improvement of public transport. In the case of London, the London Greater Authority Act (Acts of Parliament, 1999), which gives the Mayor power to introduce congestion charges, requires that net revenues are used for the local transport plan. Most have and continue to be allocated to the bus network. A smaller percentage is allocated to improvement of roads and bridges, road safety, walking and cycling facilities and facilitation of distribution of freight (TfL, 2007, Table 6.3, p. 114). In the case of Singapore, which has the longest history of congestion charging, revenues are also used to finance projects in road transport.

The fact that net revenues from the urban congestion charging schemes have been earmarked to road transport has helped political acceptability. Surveys carried out in London between March and August 1999 found that people changed their attitude towards the idea of congestion charging when they were told that revenues would be earmarked to transport. 67 per cent of the general public thought that road user charges in central London would be a good idea if net revenues were spent on transport improvements, and the proportion increased to 73 per cent when the respondents’ spending preferences were introduced (ROCOL Working Group, 2000, p. 57).

5.4. Concluding remarks

Taxes and charges in the road transport sector are used extensively throughout the world. Most countries levy a registration tax on new vehicles and annual vehicle excise duties. More recently many have also introduced subsidies to efficient vehicles, feebates and/or scrappage incentives.

Most countries also have fuel duties, some taking the form of carbon taxes. Fuel taxes vary substantially from one country to another. Mexico and the US, for example, have very low petrol taxes, when compared to other OECD countries. Great differences can also be observed across African countries.

Many European countries depend on motor fuel excises, which typically represent 4–6 per cent of total tax revenues. If these taxes were earmarked for road transport only, other sectors in need of funding would suffer. If the road transport sector were decarbonised and alternative forms of energy were exempt from taxes, governments would need to introduce taxes in other sectors of the economy to make up for the loss of revenues they would suffer.

6. Conclusions and policy recommendations

6.1. Conclusions

In this paper we have summarised the main road transport externalities and economic policies proposed and implemented to deal with them.

The most important negative externalities from road transport include accidents, road damage, environmental damage, congestion and oil dependence.

Since the market is incapable of reaching an efficient equilibrium in the presence of externalities, a number of government interventions have been suggested in the literature. From an economic point of view, these interventions fall under two headings: command-and-control (CAC) and incentive-based (IB) policies. CAC policies are essentially regulations, which force consumers and producers to change their behaviour. IB policies are
The main advantage of IB policies (both price and quantity control) over CAC ones is their cost effectiveness. Producers or consumers with low costs of abatement tend to reduce the amount of externalities generated by themselves, rather than buying permits or paying taxes, while those with high costs of abatement choose to buy permits or pay taxes. The cost of reducing the externality is thus minimised compared to the more direct regulatory approach of setting standards, which obliges everyone to reduce their generation of externalities to the level fixed by the regulator, regardless of how costly abatement for each firm or individual is.

A number of CAC and IB policies have been reviewed. CAC policies are not efficient from an economic point of view. Even in a context of perfect information when the regulation or standard is set at the optimal level, the target is not achieved at the minimum cost, and worse yet, the social costs could exceed the potential benefits (thus replacing market failure with government failure). Despite all this, they are the type of instrument most favoured by policy makers.

Regulations, however, seem to be adequate for regulating fuel quality, perhaps complemented with a duty differential or a facility for trading credits. This approach was successful, in phasing out leaded petrol in many countries, including the US and those in Europe, as well as in reducing the sulphur content of petrol and other harmful emissions.

Standards also seem to be adequate for regulating tailpipe emissions: standards can become increasingly stringent over time, and they can regulate emissions of nitrogen oxide, hydrocarbons, carbon monoxide and particulate matter. They could also serve for regulating CO2 emissions, although this is a much more problematic area, where the inefficiency of CAC becomes evident. A carbon tax, or, at least, a fuel tax would be better suited as an instrument for reducing CO2 emissions. We return to this point below. Fuel economy standards for vehicles, not too dissimilar from CO2 caps per km, reduce fuel consumption but also cause a rebound effect.

The Low Emission Zone (LEZ) in London is a CAC policy. However, its cost effectiveness may not be as poor as that of other CAC instruments, mainly because it is being phased-in. In reality, the vehicles affected by the regulation are few, as the increasing standards, required for circulation inside the LEZ, kick in years after the EU-standards, on which they are based, come into effect.

Bans on vehicle circulation, especially of the types imposed in Athens and Mexico City, do not ensure that the most valuable trips will be made. Worse yet, they may result in more, rather than less, driving. This is because motorists may acquire second hand cars, or forge licence plates to be able to drive every day of the week.

Parking restrictions pose a similar problem to a ban on vehicle circulation: the trips that are made may not be the most valuable ones. Thus, an inefficiency will occur. Also, drivers may contribute more to congestion while looking for an available parking space.

IB policies do not impose any choices, but rather leave consumers and producers to make decisions according to their costs and benefits, preferences and constraints.

The use of revenues generated through IB road transport policies that correct externalities has impacts on social welfare and equity and is an important determinant of the political acceptability of these policies. In general, using these revenues to reduce distortionary taxes increases the efficiency of the system, and investing them in the transport sector (on roads, public transport, R&D) increases equity.

Except for the inter-refinery averaging and banking of credits instituted by the US EPA in the 1980s to facilitate the phase-out of leaded petrol in the US, cap-and-trade schemes have not been implemented in the road transport sector in any country or region to date.

The main candidate for a permit system in the road transport sector is CO2 emissions. One important difficulty is that emission sources in road transport are small, dispersed, and mobile, which makes enforcement difficult. This obstacle could, however, be overcome by implementing a system that would work at the pump (and in that case it would essentially be similar to a fuel tax) or one at upstream level, designed either for fuel producers or car manufacturers.

Although tradable permits would in principle incentivise the development of fuel-efficient and alternative fuel vehicles, there are a number of potential problems relating to their implementation: (a) auctioned permits would awake public and political opposition, whereas grandfathered permits would deprive governments of revenues, which could be allocated to reducing distortionary taxes or for investment in R&D of new technologies; (b) permits with shorter life would have higher tradeability, but also transaction costs, and permits with longer life would have greater uncertainty about future prices; (c) including road transport in a much larger scheme, such as the EU ETS, would achieve reductions in emissions, although not necessarily in road transport. Designing a separate system, would guarantee emission reductions in road transport, but not necessarily in other sectors with lower abatement costs.

Taxes and charges have been widely implemented in the road transport sector around the world, both in developed and developing countries. Their effectiveness as corrective instruments, however, depends on the link between the externality they are targeting and the tax or charge itself.

Vehicle purchase and ownership taxes can be designed to take into account the road damage they cause, so that heavier vehicles pay higher taxes. However, these taxes are unlikely to influence travel behaviour. Travel behaviour can be altered with congestion charges, PAYD insurance, and fuel duties.

Congestion, which varies with vehicle type, road and time of day, is best targeted with a congestion charge than with a fuel tax. A fuel tax may reduce demand for travel, but may not reduce it (enough) during congested periods. Examples of congestion charging are limited, with only three urban schemes, the ones in Singapore, London and Stockholm, and a few High Occupancy Toll lanes in the US. The most fine-tuned system is the Electronic Road Pricing scheme in Singapore, which charges different rates to different vehicle types, at different locations and times of the day.

Congestion charging is an IB policy which has and continues to face fierce public and political opposition. Concerns are usually linked to pervasive distributional impacts. In reality, however, car users are not willing to pay for something they have always had for free. As anecdotal evidence, implementation of the Manchester congestion charge was halted after the results of a public referendum. The scheme did take into account a number of potential negative effects and proposed solutions. These included exemptions for low-paid workers, additional school buses, and significant improvements in public transport and Park & Ride facilities.
these compromises were enough to make the scheme more acceptable.

Pay-as-you-drive (PAYD) insurance could reduce the number of trips made, which would also have a welcome impact on congestion and emissions. This would also increase the efficiency of insurance systems because the link between distance driven per year and the premium would be strengthened.

The variation in fuel duties across countries is large. Although these were originally introduced as revenue-raising instruments, they are now also increasingly being regarded as corrective taxes to internalise road transport externalities.

Bearing in mind that these are excellent instruments (only second to carbon taxes) to internalise the global warming externality the very fact that fuel duties are so different indicates the presence of an inefficiency. This inefficiency is of particular concern, given that the externality is global, regardless of where the emissions originate.

Harmonisation of fuel duties across countries in a region would support coordinated policies to reduce CO2 emissions, with just one (implicit) shadow price of carbon. One important problem regarding fuel tax harmonisation is that countries would need to agree on the tax rate.

Virtually all countries have mechanisms in place for collecting fuel duties. Thus, in addition to being fairly effective and internalising the global warming externality, a carbon tax on motor fuels, or at least blunter fuel duties, have the advantage of already being place. In addition, they are easy to monitor and enforce, inexpensive to collect, and guarantee some level of price stability. Permits are more costly in regulatory terms, in the sense that they require an institutional setting, definition of property rights and initial allocation, as well as enforcement procedures. However, there seems to be some preference, both from politicians and from consumers and producers, for tradeable permits. This is evident in the trends regarding initiatives to tackle climate change.

The EU ETS was designed and implemented after proposals for a European carbon tax failed in the 1990s. In the US there are currently proposals for cap-and-trade systems, rather than carbon taxes, let alone substantial increases in fuel taxes.

Since cap-and-trade systems seem to be better accepted, perhaps in the hope that they will be grandfathered following persistent lobbying from affected parties, the time may have come to start thinking globally on how to design and implement new systems that will be compatible with each other. Better yet, would be a unified carbon market. This would entail international agreements on country-level quotas and long and costly negotiations, but may finally result in the world-wide institution of a truly global carbon market.

6.2. Policy recommendations

On the basis of our survey, which considers economic theory and policy implementation of instruments to ameliorate road transport externalities, we offer the following policy recommendations.

Regulations and standards are not efficient from an economic perspective, but they are excellent instruments under the following circumstances:

- if the level of an activity, such as emitting a lethal substance, needs to be drastically reduced or altogether eliminated;
- when the constraints faced by the regulator, such as public and political acceptability of incentive based measures, are severe;
- as complements to incentive based policies, as long as the two instruments - the command-and-control and the incentive based one - are not designed to achieve the same reduction in activity (because otherwise the reduction would be beyond optimal).

In general, regulations on fuel composition and vehicle emissions have worked well, and are both feasible and effective instruments.

Given that incentive based policies are efficient from an economic point of view, but in reality, do not always fulfill their cost minimisation potential, we do not recommend widespread adoption of this type of instrument blindly. However, taxes, charges and permits are excellent instruments in the following cases:

- when the revenue generated from taxes or permit auctions by the government can be used to reduce distortionary taxes in the economy, such as income taxes, and/or be returned to the road transport sector in the form of investment in public transport or R&D of cleaner technologies;
- as a driver to change economic agents’ behaviour, such as driving fewer km or increasing the use of public transport;

When faced with the choice between permits and taxes, governments may find that taxes are more practical and administratively easier to implement in the case of road transport. However, given the general current trend throughout the world to consider permits a refreshing alternative to taxes, the time may be optimal to move towards tradeable permits to internalise the external costs of CO2 emissions, if only because they seem to be more acceptable. It should also be highlighted that, if a global carbon market emerged, with the participation of most countries and economic sectors, the price signals would make decisions easier at all levels (production and consumption, as well as investment) , as well as targeting them towards lower carbon choices.

Taxes and charges, on the other hand, should be used when there is a strong link between the tax or charge and the externality in question. Congestion charging and PAYD insurance, for instance, are good examples.

Governments can choose from a variety of excellent policy instruments, which reduce carbon emissions from road transport and address the problem of climate change – “the greatest and widest-ranging market failure ever seen” (Stern, 2006, p. i). Implementing these will require political will and commitment. But given the high costs of postponing these decisions, the best policy is to act now.

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