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Motor Vehicles and the Environment

WINSTON HARRINGTON AND
VIRGINIA McCONNELL



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Motor Vehicles and the Environment

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Executive Summary

Motor vehicles play a conspicuous role in the modern industrial economy—and in shaping our natural and built environment. Cars and light trucks offer rapid, reliable, and convenient mobility on demand to an ever-growing number of people in countries throughout the world. But for all their positives, automobiles carry with them many negatives. No one disputes that motor vehicles collectively contribute to a number of important and pressing social problems. Vehicles are a major source of both air pollution and congested roads, particularly in urban areas, where vehicle concentration is the greatest. They also contribute to global warming, accounting for a large and growing share of greenhouse gas emissions worldwide. In addition, many hold the rapid increase in vehicle ownership and use responsible for the decentralization of urban areas and the negative effects of the resulting urban sprawl. Rapid growth in numbers of vehicles worldwide ensures that these issues will remain important for the foreseeable future.

This report reviews what we know about the negative effects of vehicles and vehicle use on the environment and assesses the policies available to manage those effects. Economic issues are at the heart of debates over how best to design policies to mitigate these effects. We first review the methods for measuring the external costs of vehicle use and then summarize approaches for evaluating policies, including cost-effectiveness and cost-benefit analyses. We focus on the important role of modeling and estimating motorists' responses to changes in the cost of owning and operating vehicles. We then examine three of the major ways vehicles adversely affect the environment—air pollution, climate change, and urban sprawl—and explore policies to solve these problems. Finally, prospects for the future of transport, including an examination of the environmental and economic aspects of the array of possible future technologies, are examined.

Private and Social Costs of Vehicle Use

A compelling indicator of the value of driving is the high costs motorists are willing to pay to own and operate a motor vehicle: 30 to 50¢ per mile in the United States, and more in countries with higher fuel and vehicle excise taxes. It is important to differentiate, however, between the fixed and variable costs of owning and driving a vehicle. The fixed costs, those that don't vary with the number of miles driven, are a large share of overall costs—40 to 70%—depending on the age of the vehicle. Because the per-mile costs are only a fraction of the cost of owning and using a vehicle, higher per-mile costs are unlikely to have a large effect on motorists' decisions about how much to drive. This is an important issue for predicting the outcome of policy instruments on motorists' behavior, and we return to it in the modeling analyses described below.

In addition to private costs, there are many social or external costs of driving that are not currently being paid by motorists. Much effort has gone into monetizing, with estimates ranging from 13 to 67¢ per mile (/mi.). The magnitude of some of these externalities is relatively small, such as vehicle effects on climate change and water pollution, while others are large, including the effects on conventional air pollutants. However, measurement of these effects poses considerable difficulties, and there remains a lack of consensus about the best estimation method-

ologies and even about what should be included as an externality. For instance, there is considerable controversy about whether costs attributable to such expenditures as infrastructure and parking should be included in external cost estimates.

Evaluation and Modeling Tools

The report reviews the standard tools, cost-effectiveness and benefit-cost analyses, that are frequently used to compare vehicle emission control policies. The difficulties that arise in applications of these methods are discussed, including using appropriate baseline comparisons, accounting for joint pollutants, and factoring in behavioral responses, including responses to price changes, when evaluating different policies.

Evaluating motorists' responses to policy changes is particularly important for designing and assessing policies. Economists have devoted considerable effort over the past 40 years to estimating the responsiveness of vehicle fleets and vehicle use to changes in fuel prices, vehicle prices, and household income. Gasoline demand is moderately responsive to both price and income, but there is convincing evidence that fuel price elasticities have declined recently. Fuel use is less responsive to price changes in studies after 1995 than it was in earlier studies. This could be caused by low fuel prices in recent years or by market interventions, especially the fuel-economy regulations in the United States, which set a floor on acceptable vehicle fuel economy and inhibited one of the major responses to changes in fuel prices. Because nearly all policy interventions will affect fuel or vehicle use, understanding how price and income elasticities are measured is important. We review two types of models: those using aggregate national or regional data and those using household microdata. The latter models are more complex but are essential to understanding the response to policies that affect household choices about both vehicle use and the types of vehicles motorists will own.

Conventional Pollutants

Vehicles are a major contributor to air pollution around the world. Vehicles account for most of the carbon monoxide (CO), and a large share of the hydrocarbons (HC), nitrogen oxides (NO_x), and particulates in major urban areas. Much of the effort to reduce pollution from vehicles to date has been in the form of increasingly strict emissions standards on new cars sold in the developed countries. These controls have reduced emissions of CO, HC, and, to a lesser extent, NO_x despite large increases in the number of vehicles and miles driven. Although new cars have become dramatically cleaner over time, many highly polluting vehicles are still on the road, including trucks, busses, motorcycles, and older cars.

Control of gasoline engines and their impact on ambient ozone pollution (NO_x and HC) has been the focus of regulatory efforts in developed countries in the last 20 years, but attention is now turning to emissions from diesel engines. Diesel engines have higher emissions of NO_x and they have significant emissions of fine particulates. Recent evidence shows that fine particulates may be the most serious threat to human health in urban areas. The United States and Europe are each in the process of mounting new regulatory efforts to reduce emissions of both NO_x and particulates from diesel engines, although the approach is different on the two sides of the Atlantic. Europe will rely on more market-based approaches, and the United States on more uni-

form regulations on fuels and engine standards. Diesel emissions are the largest and most serious contributor to urban pollution in developing countries, because of the large share of trucks and diesel buses in vehicle fleets.

We focus on the use and possibilities for market-based policies for reducing vehicle pollution. One of the great success stories of market-based policies in the United States is the use of a credit trading and banking policy to reduce lead in gasoline in the 1970s and 1980s. Other examples of policies that make use of markets are old car scrap policies and differential pricing on vehicle fuels. Many studies have examined the possibilities and pitfalls of fees on vehicle emissions. Narrowly targeted fees on emissions would be the most direct way to provide incentives to reduce emissions, but the measurement and transactions costs of such fees are still too large. We review possible alternatives to direct emissions fees, including policies that target smaller numbers of upstream sources and combinations of regulatory and more feasible fee policies.

Mobile Sources and Global Warming

The transport sector is a major contributor to greenhouse gases. In the United States, about 20% of carbon dioxide (CO₂) emissions come from motor vehicles, and in developing countries emissions are growing apace with motorization. Reducing such emissions from vehicles means reducing fossil fuel use. That means that a tax on the carbon content of fuel is an almost ideal policy instrument against global warming. However, raising fuel taxes to reduce carbon emissions is an instrument with varying political prospects around the world. Past experience suggests that opposition to higher fuel prices will be particularly fierce in the United States and perhaps in other countries with a history of low fuel prices.

In the United States, the favored approach is mandatory fuel-economy standards for new vehicles. Since 1979, the Corporate Average Fuel Economy (CAFE) standards have probably been the main reason why, between 1979 and 1997, the average fuel economy of the car fleet has increased from 14.5 to 21.5 miles per gallon (mpg), and from 11.9 to 17.2 mpg for the truck fleet. However, CAFE may have had a number of other effects. By reducing the cost per mile of travel, CAFE caused an increase in vehicle use, reducing the effectiveness of the policy by about 10–30% below what would be expected from consideration only of fuel-economy improvements. CAFE is also held by many to have encouraged the development of sport utility vehicles (SUVs) and the shift in passenger fleet composition toward trucks, which have much lower fuel-economy standards. The changes induced in the fleet — both the increase in the number of trucks and, within the car sector, toward lighter and more fuel-efficient cars — are argued to have caused an increase in the number and severity of accidents, CAFE's most controversial effect.

The CAFE standard for cars has not changed since the 1991 model year, and that of trucks not since 1996. Political pressure to raise the CAFE standards is growing in the United States, and the issue will continue to receive attention from Congress, where much of the discussion will center on CAFE's unanticipated consequences. To some extent, the adverse consequences have resulted from some particular features of the standard, notably the division between trucks and cars and the loose definition of what constitutes a "truck." Future policymaking efforts will center on these unanticipated consequences and what can be done about them.

Vehicles and the Urban Environment

There is no doubt that increased vehicle ownership and use are associated with more dispersed and less dense development in urban areas around the world. What is more uncertain is how serious a problem this is, and what can be done to mitigate it. Because motorists do not pay the full social costs of driving, vehicle use and the associated sprawl of urban areas is too great. In addition, there are complex interactions of vehicle use and urban structure. Do certain types of development patterns lead to more driving and more decentralization? And, conversely, to what extent can alternative development patterns, such as mixed-use and more-compact residential patterns, reverse this trend?

A number of studies have attempted to look at how urban development patterns affect vehicle ownership and use, and the evidence is somewhat mixed. Most empirical studies do not find that land-use variables have a large effect on vehicle miles traveled (VMT) compared to other factors that influence vehicle use. In addition, urban-density levels appear to have a greater effect on vehicle ownership than on vehicle use.

What is clear is that the cost of driving has decreased over time in most urban areas. This alone can cause more driving and more decentralization. In addition, with more decentralization comes more congestion along with increased demand for road building, which is much less costly at the outer edges of cities. More road building leads to more development, in a continuing cycle that results in large, low-density urban areas.

Policies to mitigate these effects can be grouped into those that are regulatory or those that rely on prices or markets to change behavior. The most common regulatory policies in the United States have been growth controls and growth boundaries. Market-based policies have included parking fees, high-occupancy vehicle lanes on major roadways, and vehicle-ownership taxes. Perhaps the most promising policy on the horizon is electronic pricing of roadway use.

Overarching Policy Issues

There are many policy options for managing the effects of motor vehicles on the environment. However, several issues are central for the design of policies: Is there a role for market-based policies? Where in the production or consumption stream should the policy intervention be targeted? And, how can policies be designed to avoid unintended consequences? We address each of these issues throughout the report but summarize them briefly here.

Market-Based Policies. Market-based policies for reducing environmental effects from vehicle use have been suggested both in the economics literature and in the policy arena for years but have been used very little in practice. This is because of the sheer magnitude of the numbers of vehicles, the difficulty of measuring individual emissions, and the multiple effects of vehicle use make some market-based policies infeasible and, in some cases, too expensive.

Nevertheless, some market-based policies are feasible and offer the promise of better incentives to households and producers to take actions that will improve environmental goals. We highlight cases where simple market-based instruments, such as fuel taxes, may yield comparable results to more complicated policy instruments. We also review examples where combinations of policy instruments might provide the best outcome. Economic instruments often meet

with political opposition because they tend to be viewed as tax increases, but there are ways to make them revenue neutral. Taxes on diesel fuel in Europe — where high-sulfur fuel is taxed at a higher rate and lower-sulfur, cleaner fuel is taxed a lower rate — represent a case where the policy can be both efficient and revenue neutral.

Policy Intervention: Production or Consumption? A crucial design issue is where to target policy instruments: upstream or downstream in the manufacturing, sales, or consumption process. If ideal policies that directly target pollutants at the point of damage to the environment are difficult or costly, upstream policies may be a more effective approach.

Even if economists and policy analysts could agree on the damages from pollutants and their direct source or sources, targeting those sources may be very difficult, given the millions of vehicles and drivers. The transactions costs of finding the polluting vehicles and then enforcing regulations on their drivers may be prohibitively high. An alternative is to direct policies further upstream in the pollution process. Reducing lead in gasoline in the United States was done by allowing credit trading of lead upstream at the refinery level, rather than on fuel at the pump.

Reducing emissions of greenhouse gases from gasoline could follow a similar approach. Taxes or regulations on carbon content can be levied either at the pump or upstream on distributors or refineries. The upstream option may permit easier enforcement and have lower overall costs. In another example, vehicle inspection and maintenance programs in many areas have not been as successful at reducing emissions as many had hoped. The costs of enforcing compliance on so many drivers are high, but a policy that shifts responsibility upstream, requiring vehicle manufacturers to maintain the pollution equipment over the life of the vehicle, may be more effective and implemented at a lower cost per vehicle. Of course, such a policy would also reduce motorist incentives to maintain vehicle emissions systems, so a policy change of this sort would have to weigh the administrative and likely technical advantages of the upstream approach against its potential behavioral implications.

The Importance of a Comprehensive Approach. We also observe one of the greatest dilemmas for policymakers—the unintended consequences that often result if the policy is targeted too narrowly. In case after case, policies have focused on only one pollutant or even a set of pollutants, ignoring impacts in other environmental areas. For example, diesel-fueled vehicles get better mileage than gasoline-fueled vehicles, which make them better with regard to curbing greenhouse gas emissions. But they also currently generate higher emissions of particulates and nitrogen dioxides (NO₂), and may therefore add to local air pollution. New regulations proposed for diesel in the United States and Europe will dramatically reduce the local pollutants from new engines in the years ahead, but emissions from existing diesel fleets will remain high in much of the rest of the world into the foreseeable future. Clearly, vehicle policy on global warming may have unintended consequences in other environmental media and such consequences must be anticipated and considered.

In addition to the use of fuel taxation to raise revenue, a variety of fee instruments can be used to address different policy goals, including pollution reduction, congestion mitigation, and reduction in greenhouse gas emissions. Rather than a piecemeal approach, it makes sense to consider these instruments in a comprehensive framework.

The importance of a comprehensive approach extends to the use of regulatory instruments. For example, the CAFE standard of the early 1980s, when fuel prices were low, created a hardship for vehicle manufacturers, who were in the untenable position of making vehicles that consumers had little interest in purchasing. Manufacturers responded by taking advantage of the more lenient CAFE standard for trucks by building vehicles that counted as trucks for regulatory purposes but appealed to consumers as personal vehicles. Trucks—mainly SUVs and minivans—now account for over 50% of new vehicle sales in the United States. This outcome was an unintended consequence of the CAFE policy.

In another example, the federal government recently provided subsidies to manufacturers if they produce “dual-fuel” vehicles that can be operated on either ethanol or gasoline. But consumers have been given no incentive to use ethanol. Ethanol prices remain higher than gasoline, and consequently, drivers use only gasoline. While the subsidy has resulted in more dual-fuel vehicles, there will be no benefit to the environment unless the vehicle and appropriate fuel policies are jointly implemented.

Continuing Importance of Advanced Technology

We have also looked to the future in this report, to see what is on the horizon for vehicles and their effects on the environment. Since 1970, virtually all emissions reductions in motor vehicles have come about because of technology-based emissions standards imposed first in the United States and soon afterward in Europe and Japan. Regulation forced the development of new abatement technologies that reduced emissions rates of new vehicles by two orders of magnitude by 2000. These regulations may also have had an indirect role in introducing technology that improved overall vehicle performance. For example, fuel injectors were found on only a few high-performance vehicles in 1970. Emissions-control systems, which require precise distribution of fuel to the engine, helped hasten the diffusion of fuel injectors to all vehicles. A similar story could be told with respect to the spread of digital technology for monitoring every aspect of engine performance.

The pace of technological improvements, influenced by regulatory pressure, continues to grow. Efforts are underway around the world to develop vehicle propulsion technologies that reduce the use of fossil fuels, either by improving fuel economy or by switching to renewable fuels. And research is underway to make what is likely to be the next great technological leap—replacing the internal combustion engine with fuel cells.

Technology to reduce the environmental footprint of motor vehicles will be even more vital in the future. Worldwide, the number of motor vehicles in use is expected to double in the next 25 years. By 2015, China is expected to have 60 vehicles per 1,000 people, over five times as many per capita as it has today. And in the United States, VMT continues to grow at a faster rate than both population and economic activity. Even after 100 years, the revolution in personal mobility has barely begun.

Introduction

One hundred years ago the new horseless carriage was hailed as a clean technology for urban transportation. And so it was, at least compared to the technology it eventually replaced, namely horses and horse-drawn carriages. Since then, the growth in the number and use of motor vehicles, together with the ramifications of that growth, has been among the most conspicuous features of the modern industrial economy, as well as one of the most influential forces on the natural and built environment.

Motor vehicles bring rapid, reliable, and convenient mobility on demand to those lucky enough to have access to them. And increasingly, even in some developing and transitional economies, the lucky ones are not just the elites. The automobile is truly a mass transportation medium, in precisely the same way radio or television is a mass communication medium.

And yet, as one acute observer wrote several decades ago, “Today, everyone who values cities is disturbed by automobiles” (Jacobs 1961). For all their advantages, automobiles, especially in large numbers, bring with them an array of negative effects. Throughout the world motor vehicles are a major source of pollution, particularly in urbanized areas, where vehicle concentration is the greatest, and where pollution from all sources is most severe. They cause congestion and accidents, although compared with their predecessor technology, it is not clear that motor vehicles are any less safe. Certainly there are more traffic deaths today than a century ago, but there is vastly more traffic. Motor vehicles are collectively a significant contributor to greenhouse gas emissions because they run on fossil fuels. And finally, by reducing the cost of transportation, motor vehicles have contributed to the decentralization of urban areas, which is generally thought to have negative consequences, although there is no consensus on the reasons why.

Our purpose here is to review what we know about how vehicle use affects the environment and to assess the policies available to manage those effects. This is clearly an enormous area of study, and though we would like to be comprehensive in our treatment, we focus our analysis of these issues in a number of ways.

Because our perspective is on the economic aspects of vehicle use, we review the literature on a range of economic issues, including the pricing of vehicle ownership and use, household responsiveness to prices and other economic signals, and the evaluation of alternative policies. We

*For all their advantages,
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numbers, bring with them
an array of negative effects.*

first look at the current evidence about the social costs of vehicle use, the implications of those costs, and current tax policies to see whether vehicles are “paying their way.” Then, we examine a range of economic models for evaluating the effect of vehicle use and vehicle policies on the environment. We review the potential for cost-effectiveness and cost-benefit analysis for evaluation of alternative policies. A key component of such analyses is the ability to accurately predict behavioral responses to different factors, so we pay particular attention to behavioral models of transportation choice. We focus primarily, though not exclusively, on the household sector, since the behavioral responses of consumers about how many cars to own and how much to drive are so important to policy outcomes.

The study addresses, in some detail, three environmental issues and reviews policies to reduce the influence of vehicles on these problems. The three are: conventional urban pollutants, such as ozone, particulates, and nitrogen oxides (NO_x); greenhouse gas emissions; and urban

In case after case, we observe that policies tend to be focused on only one pollutant or even a set of pollutants, ignoring impacts in other environmental areas.

decentralization or urban “sprawl.” These are not necessarily the most important environmental problems associated with motor vehicles but, arguably, they are ones that have received the most attention from politicians and policy analysts in recent years. In each of these areas, a range of policy tools may be used to manage the environmental effects of vehicle use. Some of these serve as better solutions than others. We identify policy designs and policy targets that are likely to be most effective.

First, we explore market-based policies, which have the potential to provide the right incentives for reducing use but are subject to high costs of implementation in some cases. Market-based policies for reducing environmental effects from vehicle use have been suggested both in the economics literature and in the policy arena for years, but they have been used very little in practice. This is because of the unique characteristics of vehicles and vehicle fleets that complicate policy solutions and make certain market-based policies infeasible. These characteristics include the sheer magnitude of the number of vehicles, the difficulty of accurately measuring individual vehicle emissions, and the multiple environmental effects of vehicle use. For example, the difficulty of measuring emissions and then enforcing emissions fees for vehicle use indicate high transactions costs for direct prices on emissions. High transactions costs can more than wipe out any possible savings from a market-based policy. New technologies may lower these transactions costs for some policies, making them more feasible in the future, as we discuss below.

In general, there are clearly opportunities for market-based policies to mitigate vehicle effects on the environment. We examine a number of such cases, both in the theoretical literature and in practice. We highlight cases where simple market-based instruments, such as fuel taxes, may yield comparable results to more complicated policy instruments. We also review examples where combinations of policy instruments might provide the best outcome.

Another key design issue is how and where to target policy instruments: upstream or downstream in the manufacturing, sales, or consumption process. When the transactions costs of enforcement are high for downstream policies, targeting the policy to a production point upstream can offer an improvement. For example, because emissions of certain pollutants cannot be easily measured from the tailpipe, a policy that affects fuel use may be more effective and less costly.

We consider cases when upstream policies, even if they do not target pollutant levels directly, may yield the best results.

Finally, in assessing policies, we pay particular attention to how vehicles contribute to a broad range of environmental impacts. In case after case, we observe that policies tend to be focused on only one pollutant or even a set of pollutants, ignoring impacts in other environmental areas. For example, diesel-fueled vehicles get better mileage than gasoline-fueled vehicles, which makes them better with regard to curbing greenhouse gas emissions. However, they also currently generate higher emissions of particulates and NO_x , and may therefore add to local air pollution. New regulations proposed for the United States and Europe will dramatically reduce the local pollutants from diesel in the years ahead, but diesel emissions will continue to be high in much of the rest of the world into the foreseeable future. Clearly, vehicle policy on global warming may have unintended consequences in other environmental media. Such consequences must be anticipated and considered along with the stated objectives in making policy decisions.

■ ■ ■

CHAPTER TWO

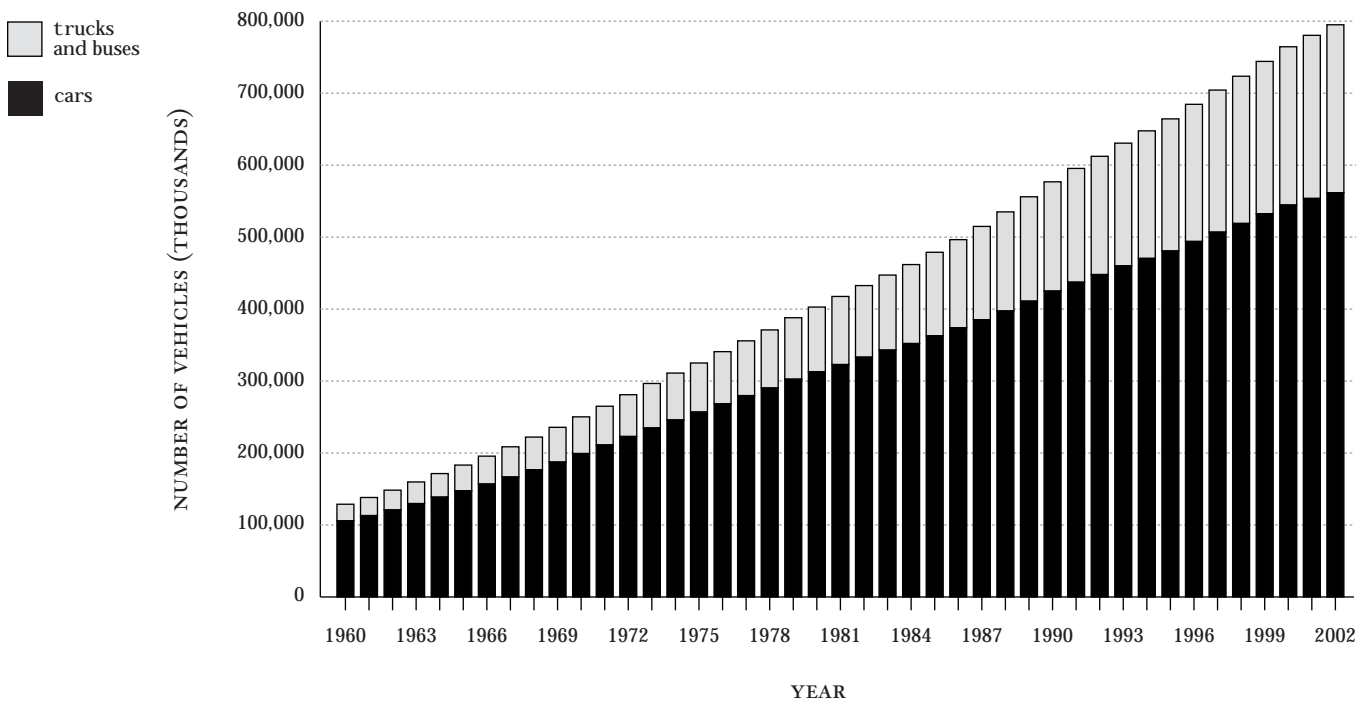
Vehicles in Context

Vehicle Holdings

The total stock of vehicles in the world is growing at about 3% a year. Figure 1 tracks the growth rate of cars and trucks and buses separately since 1960 and shows that the total stock of vehicles is close to 800 million, up from just over 100 million in 1960. Although holdings of both have been increasing, the growth rate of trucks and buses has been higher than that for cars in recent years, and is expected to continue to grow at rapid rates worldwide. In Europe alone, the European Commission forecasts truck traffic to increase by more than 50% in the next 10 years (Walsh 2002).

FIGURE 1

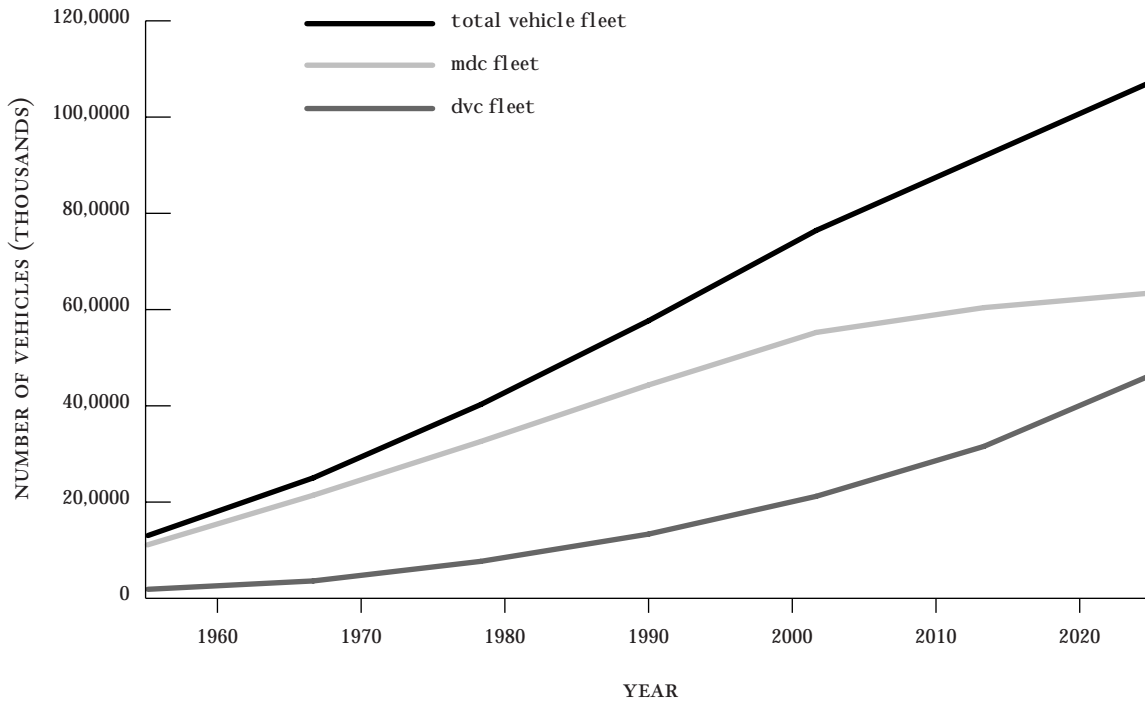
Growth in the World Vehicle Fleet



Source: Max Pemberton Associates, London.

FIGURE 2

Vehicle Fleets in More-Developed and Developing Countries



MDC: More-developed countries include Western Europe, North America including Mexico, Australia, Japan, and New Zealand. DVC: Developing countries include all others.

Source: Max Pemberton Associates, London.

The aggregated nature of these fleet data conceals the dramatic differences in vehicle holding and use among countries and regions of the world. Figure 2 and Table 1 illustrate some of these differences. Figure 2 shows vehicle holdings broken down by more-developed and developing countries, with forecasts of fleet holdings to the year 2020. The rate of growth in the fleet in the more-developed countries is predicted to decrease over time because average per capita vehicle holdings in 2002 are already very high. Although developing-country total holdings are much lower currently, they are predicted to grow rapidly in the next 20 years.

Table 1 provides more detail for selected countries. The first column shows that average vehicle holdings per 1,000 population for developing countries are many times smaller than they are in more developed economies. Holdings in the United States, Canada, and most of Europe were above 500 vehicles per 1,000 population by 1998, while holdings in China and India were less than 10 vehicles per 1,000. Overall, the developed countries hold about 78% of the cars and 66% of buses and trucks (AAMA 1996).

The last column of Table 1 shows that the average age of the vehicle fleet varies a great deal across countries. The age of the fleet is determined by the rate of new and used additions to the fleet and the rate at which vehicles are scrapped. Japan has the lowest average age, at just over five years, in part because of the high costs of getting older vehicles through a tight inspection

TABLE 1

Vehicle Holdings, Gross Domestic Product (GDP), and Vehicle Age in Selected Countries

Country	Average vehicle holdings per 1,000 population 1980	Average vehicle holdings per 1,000 population 1998	Percent change in vehicle holdings 1980–1998	GDP per capita US\$ 1999	Average age of the car fleet* (years) 2000
SOUTH AMERICA					
Argentina	155	176	14 %	10,300	11.0+
Chile	61	110	80	12,500	—
NORTH AMERICA					
Canada	548	560	2	22,400	6.7
United States	660	783	20	31,500	6.8
ASIA					
China	2	8	300	3,600	—
India	2	7	250	1,720	15.0*
Indonesia	8	22	175	2,830	9.4
Japan	323	560	73	23,100	5.3
EUROPE					
Finland	288	448	56	20,100	10.8
Italy	334	591	77	20,800	8.1
Poland	86	273	217	6,800	10.0
MIDDLE EAST					
Turkey	23	81	252	6,600	9.3
AFRICA					
Nigeria	4	26	550	960	12.0+

Sources: World Development Indicators, World Bank (2000), CIA World Factbook (1999), Eurostat Statistical Compendium (2001).

*Data from Pemberton Associates, International Automotive Research, Analysis, Forecasting and Consultancy, London.

— data not available. *Data obtained informally through research organizations.

system. In contrast, India is estimated to have a fleet with an average age of about 15 years. One factor that causes developing countries to have older fleets is that they import many used vehicles from countries in the developed world. Another is that vehicles tend to be driven until they are much older before they are scrapped. Older vehicles tend to have higher emissions of both greenhouse gases and conventional pollutants.

Over 60% of vehicles in the world are held in the developed countries, and their sheer numbers in places like the United States and parts of Europe mean their influence on the environment is pervasive. However, the rapidly growing size of fleets in developing countries and their relatively older age indicate the growing role of vehicles and policies toward the vehicle in these countries in the coming years.

The Special Characteristics of Mobile Sources

Few real-world situations fit perfectly the conditions of the standard economic approach to environmental externalities learned in one's first graduate environmental economics course and presented, for example, in Baumol and Oates (1988). However, motor vehicles are quite likely an especially poor fit. Their distinctive features offer opportunities and—more commonly—create difficulties for fashioning approaches to environmental policy, especially policy respecting conventional pollutants.

The users are primarily households, rather than firms. This means there are millions of potential polluters, rather than thousands as in the case of stationary sources. This obvious fact has had several ramifications that have profoundly shaped policy. First, emissions and emission reductions cannot be directly measured for every vehicle, nor, for that matter, can vehicle use. Vehicle use and vehicle emissions can only be determined indirectly by inferences from surveys of households or (for emissions) of vehicles.¹ Planners and regulators have had to rely heavily on modeled rather than measured emissions. In the United States, for example, the U.S. Environmental Protection Agency's (EPA) MOBILE model is now used not only for planning but for enforcement as well, a development that has had unfortunate consequences both for the model itself and for policy. Second, these millions of motorist-polluters vote, which has made it very difficult to implement effective emissions regulations directly on vehicles. Instead, most of the work in reducing emissions has been done by emissions standards for new vehicles.

Responsibility for emissions is shared among a number of parties. The actions of many different parties can affect motor vehicle emissions. Manufacturers design vehicles of different sizes and types, and install the engine and emissions-control system. Refiners can produce fuel with highly variable emission potential, particularly with regard to hydrocarbon (HC) emissions for gasoline engines and sulfur for diesel fuel. Actual emissions, of both conventional pollutants and greenhouse gas pollutants, vary greatly with how the vehicle is driven. Responsibility for vehicle maintenance and emissions-system repair is shared between mechanics and motorists.

The upside of this diffusion of responsibility is that there are many potential regulatees, and many possible interventions in the system. On the other hand, the more policies, the greater the difficulty of choosing among them. Various actors can more easily point to others as the real problem. And actions taken by one actor can be undone by the behavior of others. For example, motorists' failure to maintain their vehicles can compromise the integrity of manufacturer-installed emissions-control systems. The observed pattern of responsibility is in part a product of the institutional structure that governs motor vehicle production, ownership, and use. Thus, one possible approach to motor vehicle emissions policy is to consider ways to alter this structure. For example, manufacturers could be required to maintain pollution control over the lifetime of their vehicles; or in-use vehicle emissions control could be bid on and contracted by one party, who would then be responsible for a target level of emissions reductions. These approaches have been suggested (Harrington and McConnell 2000 have reviewed alternative property rights structures) but have not been tried.

Variation in emissions rates. The variation in emissions rates of conventional pollutants, such as nitrogen oxides and hydrocarbons, shows up in many different ways.

Among various models and vintages. In the United States, because of the gradual tightening of emissions standards between 1973 and the present, emissions from new vehicles in 1995 were less than 5% of the average emissions of uncontrolled vehicles from the early 1970s. “Engine out” emissions (i.e., emissions produced by the engine prior to abatement) have declined by about 70%. The emissions control system reduces the remaining emissions by about 85–90% (Ross et al. 1995). Light-duty truck emissions standards have not been as strict as car emissions requirements but are required to be equivalent by the middle of this decade. However, heavy-duty engines have hardly been regulated at all through this period in the United States. Recent legislation has changed this, and huge reductions in both NO_x and particulate emissions from heavy-duty diesel engines will be required by the 2007 model year.

Emissions rates also increase with vehicle mileage, probably a consequence of the gradual deterioration of the emission-control equipment and other systems on the vehicle that affect emissions. In addition, emissions appear to vary by manufacturer (Ross 1994, Ross et al. 1995, Bishop and Stedman 1996). Also, for some makes, emissions rates of the more expensive models are lower on average than in less expensive models. It is not known whether this variation arises from differences in manufacture or differences in owner characteristics (Harrington and McConnell 2000).

Among vehicles of the same model and vintage. Emissions rates of vehicles of the same model and year of manufacture can vary. Emissions systems break down in some random pattern over time, and some vehicles are better maintained over time than others. As evidence of this variation, randomly selected models at emissions inspection stations in Arizona during 1995–96 were found to have a range of emissions rates (Harrington et al. 2000, Wenzel 2001). At the time Arizona used the “IM240” emissions test, which probably gives the most accurate and replicable test results among those feasible for use in emissions inspection programs.

Under different operating conditions for the same vehicle. Emissions vary widely depending on speed and acceleration. At very high rates of acceleration, in fact, gasoline vehicles enter an “enrichment” cycle that sends more fuel to the engine than can be burned, and more than can be oxidized by the catalytic converter. The resulting emissions of carbon monoxide (CO) and HC are extremely high, perhaps hundreds of times greater than under normal operating conditions (Ross et al. 1995).

At different times for the same vehicle and same operating conditions. Some vehicles, especially those with underlying emissions-control system malfunctions, have emissions that vary substantially even under the same conditions. Bishop and Stedman (1996) call these vehicles “flippers” and estimated that they were responsible for a substantial share of fleet emissions in Colorado in the early 1990s.

It has long been well understood (and recently formalized by Newell and Stavins 2000) that emissions variation across sources strongly increases the potential benefits of economic incentive approaches to environmental policy. If so, then mobile sources ought to offer a promising application of economic incentive programs. Despite this potential, few market-based policies for motor vehicles have been implemented. Several have been suggested, as we discuss below.

The Private and Social Costs of Automobile Ownership and Use

Every year in the United States, the American Automobile Association (AAA) publishes an estimate of the cost of driving. In its most recent publication,² the AAA estimate for a late-model mid-size vehicle (2001 Ford Taurus) traveling 15,000 miles per year is 51¢/mi. Note that the AAA estimate is for a new vehicle; for older vehicles the average driving cost is much less. For example, we estimate the cost of driving a 10-year-old Ford Taurus to be 20 to 30¢/mi. In older cars, lower depreciation is countered by higher repair and maintenance costs. For a given age, cost declines with use, because there are more units over which to spread the fixed costs. These costs are, of course, only the private costs and don't include the costs imposed on others, which are rarely paid by the motorist.

Before turning to these "external" costs of auto use, we will take note of a couple of interesting aspects of private costs. First, the cost of owning and using an automobile is evidently quite high. So high, in fact, that vehicular transportation now accounts for 18% of expenditure among American households (BLS 2001). But no one forces people to buy cars, and the fact that the average American household owns 1.9 cars suggests that the benefits of auto ownership and use are also very high. In fact, the average benefits per mile must exceed the average costs.

For a new vehicle, about 50–70% of the average private costs are fixed costs, which are costs that do not vary with miles traveled. The importance of fixed cost declines with age and intensity of use, but even for the 10-year-old Taurus, fixed costs still compose 38–55% of our estimate of overall costs. The variable costs of using a Taurus are about the same, 11–15¢/mi., regardless of age. For other vehicles, the variable cost may differ (it depends strongly on fuel economy), but except for extremely old vehicles, the variable cost of vehicle use is much less than the average total cost, and *a fortiori* much less than average benefit as well. This disparity suggests what the review of empirical studies below will bear out: that the short-run elasticity of vehicular travel with respect to its marginal cost of additional mileage is very low.

Motorist costs include fuel taxes, vehicle excise taxes, registration fees, and other payments to governments. These vary tremendously in structure, magnitude, and purpose from one jurisdiction to another (Sterner 2002, chapter 21). In the United States, nearly all vehicle-related taxes are revenue-raising instruments earmarked for roads and other transportation investments. According to AAA estimates, the average annual registration fee is only about \$200, or 0.6 to

TABLE 2

Lifetime Ownership and Fuel Taxes in Europe and the United States

	<i>Ownership taxes as a percentage of net vehicle price</i>		<i>Fuel taxes as a percentage of net fuel cost</i>	
	DIESEL	GASOLINE	DIESEL	GASOLINE
Belgium	43%	35%	188%	311%
Finland	155%	127%	184%	347%
France	25%	24%	293%	428%
Germany	32%	22%	234%	334%
United Kingdom	29%	49%	509%	487%
United States	18%	18%	21%	33%

Sources: European countries: Mayeres and Proost (2001). U.S.: authors' calculations based on data in Ward's *Motor Vehicle Facts and Figures* 1999 and U.S.DOE *Transportation Energy Data Book* Edition 21 (2001). Fuel prices used in calculation: gasoline \$1.13, diesel \$1.36.

1.2¢ per kilometer.³ Gasoline taxes also vary some from state to state, but in all cases are very low, averaging only about 1¢ per kilometer. In Europe, both fuel taxes and vehicle taxes are much higher and not only pay for transportation facilities but also contribute revenues to the general fund. There are also great differences among European countries. Both points are shown in Table 2, which compares lifetime fuel and ownership taxes for the United States and selected European Union (E.U.) countries. The low diesel fuel taxes relative to gasoline taxes in Europe reflect a desire to keep commercial transport costs low.

External costs. Since the emergence of the automobile, motorists have been expected to pay for the cost of roads. In the United States, ownership and fuel taxes have historically been the method of paying for road infrastructure. In Europe, as suggested by Table 2, taxes were high enough to cover infrastructure costs plus make a substantial contribution to general tax revenues. This mode of roadway finance has come under steadily increasing scrutiny.

It has become clear that a tax that is proportional to fuel use is steadily becoming decoupled from road use, and still more from the costs imposed on the road system. Not only have vehicles become more fuel-efficient, some are using other as yet untaxed fuels. In addition, road use is itself an increasingly imperfect proxy for imposition of costs. For example, one of the most important costs of driving is the congestion cost that each additional user imposes on every other user. These congestion costs vary enormously by location and time of day, neither of which can be captured by a fuel tax. Roadway congestion was first studied extensively by Vickery (1963) and his work led to an extensive literature on congestion externalities and congestion pricing. Road wear is another reason why fuel taxes do not line up well with costs imposed. In a U.S. study, Small et al. (1989) pointed out that vehicles cause road damage at a rate (and cost) that is a sharply increasing function of the weight per axle, so that virtually all damage is attributed to heavy-duty commercial vehicles. They recommend replacement of the current fuel taxes and weight-based license fees by a mileage fee sharply increasing in tandem with vehicle weight.

Moreover, awareness is growing that driving also can have substantial off-road effects. Since about 1990 transportation researchers as well as transit and environmental advocates have been trying to describe and if possible quantify both the on-road and off-road components of the “ex-

ternal” costs of driving (e.g., Delucchi et al. 1996, Litman 1994, Lee 1995, McKenzie et al. 1992, Miller and Moffet 1993, and Kågesson 1993). A survey of cost-of-driving studies commissioned by the Metropolitan Washington Council of Governments identified nearly 40 such studies (K.T. Analytics 1997). An excellent introduction to the issues involved can be found in several of the papers in Greene et al. (1997). Although some of these studies are careful to define “external costs” as economists might understand the term,⁴ others seem to use it to include all costs not associated with vehicle ownership and operation, such as infrastructure costs and parking.

These studies have been undertaken primarily with at least three issues in mind. One concerns the marginal costs of transportation use: Are ground transportation modes properly priced, so that the socially optimal level of transport is demanded and supplied and is distributed properly among the various transport modes? To meet the efficiency objective these prices must equal the marginal social costs. The second issue is concerned with total costs, which touches not only on efficiency, but on cost recovery and fairness as well. Are the total social costs of transport fully paid by the users? Are transportation users unfairly subsidized? Third, it is recognized that transportation markets and behavior interact with other markets, notably housing and land use but also the labor market. Government policies, including fees and subsidies, have been developed in transport, land use, housing, and other policy areas without taking these interactions into account. In Europe, transport cost studies are increasingly being used to examine policy interactions as part of the E.U. drive toward harmonization of national policies (Quinet 1997).

The scope and variety of transportation impacts within this framework are quite large and include parking costs, losses from accidents, national expenditures associated with defending international oil trade, and such environmental impacts as air pollution, greenhouse gas emissions, and damage to wetlands and other sensitive areas from road construction. Table 3 provides the range of estimates in each of these categories from U.S. and European studies. As is common practice, researchers express costs in cents per passenger or vehicle (or kilometer) mile traveled. This is true regardless of whether the cost category is truly a component that varies with distance or is a fixed-cost component that is averaged over some standard distance.

The estimates differ so much both because they reflect different assumptions about the kinds of trips and mileage, and because different methods are used in evaluation. In some cases there is controversy about whether the effects cited are truly externalities, for example with infrastructure, accidents, or parking.

In addition, the differences in objectives affect what is included, and things that may be inappropriate to the discussion of whether transportation is priced efficiently may be quite germane to the discussion of whether it is priced fairly or whether road transportation covers its costs. On this last issue, the studies cited broadly agree that the revenues from motorists—primarily gasoline taxes and vehicle registration fees—approximately cover costs. However, if ancillary costs of operating the highway system—road police, signals, and the court system—are included,

TABLE 3

Range of Reported External Costs in cost-of-Driving Studies^a (cents per mile)

	LOW ^b	HIGH ^b
Infrastructure	3.0	7.0
Congestion	4.0	15.0
Air pollution	1.0	14.0
Climate change	0.3	1.1
Noise	0.1	6.0
Water	0.1	3.0
Accidents (external)	1.0	10.0
Energy security	1.5	2.6
Parking	2.0	9.0

a: Combination of the studies surveyed by K.T. Analytics (1997) and Gomez-Ibanez (1997)

b: “Low” and “High” are respectively the second lowest and the second highest estimates reported in the articles surveyed.

many authors find that the system cannot cover costs.⁵ Gomez-Ibanez (1997) reviews the differences in assumptions and definitions in some of the studies and shows how they can affect the estimates.

As comprehensive as the social-costing framework appears, it is actually incomplete in two ways. First, the external cost calculation only begins when the vehicle rolls out of the showroom and ends when it is retired. But plainly, there are also environmental impacts associated with vehicle and fuel production and of vehicle/parts disposal. The *internal* costs of these activities are of course included in the owner's costs, but any external costs are omitted. To get at external production and disposal costs, researchers have begun to apply *life-cycle analysis* (LCA), a systematic attempt to catalogue and value the full womb-to-tomb range of impacts of consumer products, to the motor vehicle. LCA was developed initially for analysis of the fate and toxicological effects of hazardous substances, and guidelines for producing and using LCA have been adopted by the International Standards Organization (ISO 14040-3). So far, however, LCAs of motor vehicles have been largely limited to construction of inventories of impacts, with few if any attempts at valuation.⁶

Research into the life cycles of conventional gasoline- and diesel-powered vehicles indicates that by far the majority of environmental impacts occur during vehicle use (MacLean and Lave, forthcoming). Thus, results of LCA for conventional vehicles would not give results much different from those in Table 3. The situation is very different for vehicles with different fuel or propulsion systems, a point to which we return in Chapter Eight, when we consider alternatives to current motor vehicle technologies.

The social cost framework also does not integrate the analyses of various impacts, but instead treats them independently. Individual cost components are computed and simply added together. Furthermore, displaying the results in costs per mile of travel encourages the interpretation that these are marginal costs, but in many cases they are average costs of elements where the marginal cost of an additional mile of travel is essentially zero. The estimates are best viewed as a rough estimate of the *current* costs of vehicle use under particular assumptions; they are not very useful for designing environmental policy to reduce pollution from vehicles.

Nonetheless, some of the various estimates of social costs have been used as inputs to more comprehensive studies that put all these externalities in a consistent optimizing framework. One such model is the TRENEN model developed by researchers in Europe (Proost and van Dender 1999). TRENEN is a welfare-maximization model of the urban transport sector in the presence of environmental and congestion externalities. In a partial-equilibrium framework, it solves for a set of welfare-maximizing policy instruments, such as externality fees or quantity restrictions, allowing for various political, practical, or technological constraints. TRENEN has been calibrated for a number of different European cities and used to examine local transport issues. For examples of its use in policy analysis, see Proost and van Dender (1999), Calthrop et al. (2000), and Roson (1998).

Proost and van Dender (1999) compare a range of possible vehicle policies for Europe using a partial-equilibrium model calibrated for Brussels in the year 2005. They report net benefits for a number of policies including fuel policies, external cost pricing, and mandates for improved emission technology. They find, not surprisingly, that the full external cost pricing policy provides the greatest net benefits. This policy has motorists pay the marginal cost of driving, in-

cluding congestion, pollution, accidents, and noise costs. They acknowledge that this policy is not feasible to implement and compare it to more feasible policies such as parking fees and cordon pricing. Cordon pricing results in about 50% of the welfare gain of full marginal cost pricing. Another interesting finding of this analysis is that the mandatory emission control technology scenario does not fare well in terms of air pollution reduction compared to the full marginal-cost pricing case. This is because the technology controls assumed reduce VOCs, NO_x, and CO but not the more damaging particulate emissions such as sulfur dioxide (SO₂).

Another such model is found in the recent paper by Parry and Small (2001), which attempts to estimate welfare-maximizing gasoline taxes for the United Kingdom (U.K.) and the United States. From a simple general-equilibrium model containing a population of identical households using vehicular transportation and a numeraire consumption good, together with a simple production sector employing labor from the household sector, Parry and Small derive a formula for the optimal taxes to maximize social welfare in the presence of transport externalities, costly infrastructure, and a distortionary tax on labor. The transport externalities they include in the model are congestion, air pollution, and accidents; as shown by the range of estimates in Table 3, except for global warming, these categories are numerically the most significant. These externalities also vary by mileage, whereas global warming varies by fuel use.

What they find is that the optimum gasoline tax for the United States is \$1.01 per gallon, more than double the average combined state and federal gasoline tax throughout the country (37¢/gallon), and the optimum tax for the United Kingdom is \$1.34, less than half its current value. Compared to the respective base cases, these taxes would raise welfare by 7% and 23%. Converting the U.S. number to cents per mile and assuming a fleet average fuel economy of 20 miles per gallon (MPG), the Parry-Small optimum fuel tax for the United States is equivalent to 5¢/mi., compared to the current tax of 1.8¢/mi.

However, Parry and Small also calculate the optimum vehicle miles traveled (VMT) tax, and find it to be 14¢/mile. This tax increases welfare by 28%. The reason why the VMT tax is so much more efficient than the gasoline tax is because VMTs generate most of the external costs of driving, not fuel use. And discouraging fuel use is not a very effective way to reduce VMT. As we discuss further in Chapter Six, most of the response to higher fuel prices, at least in the long run, will involve a shift to more efficient vehicles rather than a reduction in VMT.

One possible surprise from Table 3 is the relative unimportance of externalities affecting the natural environment, except for conventional air pollution. The social cost of global warming attributed to motor vehicles is small relative to other costs, only 0.3 to 1.1¢/mi. It is also surprising to find little mention of urban sprawl, an important part of the environmentalist indictment against motor vehicles in the United States. The low ranking of these environmental effects seems strangely at odds with the high value the public places on environmental matters. For accidents, at least, this disparity may reflect public awareness that this externality is partially

The main implication is that internalizing these social costs requires an increase in the cost of driving. Contrary to what some authors think, however, finding the optimum policy is more than just a matter of summing the costs and imposing a tax.

internalized by insurance payments. Another possible explanation is the difference in the parties affected in the transactions. The congestion and accident externalities primarily affect other drivers; when one uses a vehicle, it is with the understanding that accidents are possible and congestion is likely, and that both are part of the price one has to pay to enjoy the benefits of driving. Environmental effects are in a different category, affecting the general population, not just motorists. Perhaps this is another example of the very different attitudes toward voluntary versus involuntary risk.

A third explanation for these attitudes could be that the actual damages from global warming and the effects of sprawl are poorly understood and not reflected in the damage estimates. In some cases, the estimates of social costs associated with global climate change are based not on damage estimates, but on the estimated marginal cost of emission reductions from stationary sources sufficient for complying with the Kyoto Protocol. It is worth noting that most observers agree that if the concerns about anthropogenic climate change turn out to be justified (as the evidence increasingly suggests they are), the requirements agreed to in Kyoto are but a small first step. The damages associated with climate change are just the kinds that benefit-cost analysis has trouble dealing with: damages that are uncertain but potentially very large and very far in the future.⁷

Likewise, few analysts have ventured an estimate of the social costs of sprawl that are attributable to the automobile. In part this is because there is little consensus on what sprawl is and what its effects are. And whatever they are, the social costs of sprawl cannot entirely be laid at the tires of motor vehicles. Brueckner (2001), for example, puts part of the blame on land-use development impact fees that are below the marginal cost, requiring a taxpayer subsidy. Pietro Nivola (1999) discusses several other factors contributing to suburbanization and low-density development, including home mortgage subsidies, school desegregation, and federal highway subsidies. In any case, because many of the policies now being used or considered to combat sprawl directly affect urban transportation, we will examine the urban transportation and land-use connection below.

To sum up, then, the social costs of automobile use are potentially large and, at 13 to 67¢/mi., comparable in magnitude to the private costs in some situations. The main implication is that internalizing these social costs requires an increase in the cost of driving. Contrary to what some authors think, however, finding the optimum policy is more than just a matter of summing the costs and imposing a tax. For one thing, even though the social costs are customarily expressed on a per-mile basis, they actually affect several different margins, including fuel use, VMT, and emissions. This means that the optimum policy will require several different instruments, one for each margin. Because of the interrelatedness of these externalities, very likely it would not be the optimum policy to set each instrument at the marginal damages. Finding the optimum taxes requires a comprehensive modeling framework like that of Proost and Van Dender (1999) that takes into account the interrelations. Most likely, a policy of this sort would be too complicated to be enacted. For this reason, researchers continue to look for simpler policies, such as Parry and Small's VMT tax, that can capture most of the benefits of an optimum instrument. We return to these points frequently in later chapters.



CHAPTER FOUR

Evaluation and Modeling Tools

Let's say you are a policy analyst charged with examining both the likely consequences and the wisdom of policies at the nexus of environment and transport. These could include, for example, highway or transit investments, policies to encourage or discourage certain technologies, vehicle emissions policies, even land-use policies. What would you like to know? What tools and resources are at your disposal? What are you unlikely to know, given current data and methods?

To illuminate the ways economists have approached issues raised by the automobile, we will center the discussion on three environmental problems in which motor vehicle use has been widely implicated: (1) conventional emissions (particulates and ozone), (2) global climate change, and (3) land use and sprawl. In the first two cases, economic models have been used extensively to examine the properties and likely consequences of various policy options and to make judgments about their relative merits. Detailed economic analysis has made less headway in analyzing sprawl, but it shows considerable promise for doing so in the future.

Before turning to those specific issues, in this chapter we discuss first the techniques that have been used to evaluate policies to reduce vehicle emissions. The standard tools of cost-effectiveness and benefit-cost analysis have frequently been used to compare vehicle emissions control policies. We discuss some of the difficulties that arise in using these techniques, and describe a handful of studies. While a complete review of the extensive literature on cost-effectiveness and benefit-cost analyses of policies to reduce vehicle emissions is beyond the scope of this study, we describe some surveys of the literature and examine several representative studies. We then go on to review an area of the literature that is critical for almost all evaluation of policies that affect vehicle use and vehicle emissions. This is the literature on income and price elasticity of demand for fuels, vehicles, and vehicle use. We look both at the results of models that use aggregate data to estimate elasticities and at models that use household micro-data to estimate the demand for vehicle type and vehicle use.

Evaluation Techniques: Cost-Effectiveness and Benefit-Cost Analyses

As in other areas of economics and policy analysis, cost-effectiveness analysis (CEA) and benefit-cost analysis (BCA) are used extensively to assist policymakers in the evaluation of environmental policies associated with vehicles and vehicle use.⁸ In CEA, the analyst takes an objective as given

and looks for the least costly among the range of options that advance the objective, while in BCA, both benefits and costs of any option can, in principle, be compared, including the “do-nothing” alternative. Estimating benefits in addition to costs under BCA vastly increases the difficulty, controversy, and, if done well, usefulness of the enterprise.

BCA is proving crucial in determining how to best allocate resources for controlling air pollution in the United States. Until recently, policy analysis tended to focus on attaining target levels of air quality for particular air pollution problems such as ozone or carbon monoxide. In the last 25 years, many billions of dollars have been spent on reducing these pollutants (see Chapter Five), while other pollutants such as particulates and sulfur dioxide remained relatively uncontrolled until recently. New evidence about the relative damages from particulates continues to accumulate, making particulates the subject of new regulation both in the United States, Europe and other countries around the world (for example, in the form of new diesel fuel regulations). However, there is still a great deal of uncertainty about the chemical interactions and the health consequences of pollutants such as fine particulates and even ozone (Small and Kazimi 1995, Onursal and Gautam 1997, Harrington and Krupnick 1997). Understanding the health effects and their value to society from vehicle-related pollution is an important area for future work.

In what follows, we discuss some particular issues that arise when CEA or BCA is used to evaluate policies related to motor vehicles.

Baselines. Estimation of costs and benefits requires a counterfactual, credible scenario of what would have happened in the absence of the action being evaluated. In general, the larger and more comprehensive the action or policy under consideration, the greater the uncertainty associated with the counterfactual. For example, determining the cost of the Clean Air Act, which EPA is required by law to provide periodically, is fraught with uncertainty because it requires the analyst to imagine what would have occurred if the Clean Air Act had never been passed.⁹ More specifically, the Clean Air Act mandated the reduction in tailpipe emissions in new vehicles, and estimating costs of meeting the new standards was a major component of the “Cost of Clean.”

Failure to agree on the baseline was a source of discrepancies between EPA’s estimates (USEPA 1984) and those of several other researchers (Kappler and Rutledge 1985, White 1982, Crandall et al. 1986, and Wang et al. 1993). Kappler and Rutledge’s estimates were prepared for the Bureau of Economic Analysis (BEA) in the U.S. Department of Commerce and are referred to as the “BEA estimates” below. The treatment of new technology provides a good example of the baseline problem. After decades of stagnation in motor vehicle technology, the 1980s saw an explosion of innovation in engines, mainly through the application of advances in electronics and information processing (including electronic fuel injection, electronic ignition, and onboard computers). Not only did the new technology help reduce emissions, it also raised performance, improved fuel economy, and facilitated vehicle maintenance and repair. Most observers agree that the emissions requirements hastened the adoption of the new technology, but there is disagreement on when and how much of this equipment would have been developed and adopted without the impetus of the emissions regulations and the new fuel economy standards that were adopted at about the same time (see McConnell et al. 1995 for a summary of these issues).

Jointness. Policy analysts must be aware of two kinds of jointness or inseparability when evaluating the benefits and costs of motor vehicle policies. First, as noted in the preceding chapter, many new technologies simultaneously reduce emissions and improve vehicle quality, and estimating the cost of pollution abatement requires the allocation of these costs of the technology between its public- and private-good aspects.

From 1970 to 1995, for example, tailpipe emissions of vehicles sold in the United States declined by about 98%. About 75% of the reduction was in “engine-out” emissions, that is, the emissions from the engine before treatment in the catalytic converter (Ross et al. 1995). In evaluating the emissions standards that produced this result, only part of the cost of the new equipment should be allocated to the emission-reduction policy. Among the cost studies mentioned above, the EPA analysis assigned half the cost of the new electronics to emission control, BEA assigned 70%, and Wang et al. 1993 assigned 25–33%. Partly as a result, BEA’s cost estimates are much higher than the others.

Second, as noted in Chapter Three, motor vehicle use is associated with several different externalities. Addressing those externalities efficiently would require consideration of all those externalities simultaneously and comprehensively, as discussed by Proost and Van Dender (1999). In reality, policies are made piecemeal, and researchers must approach a policy analysis from a particular perspective, such as to compare strategies for reducing conventional pollutants or for relieving congestion. Nonetheless, the fact that vehicle use is associated with such a broad range of problems means that evaluation of policies targeted at particular effects of motor vehicles must take into account other effects produced jointly. For example, policies that discourage driving not only reduce emissions of conventional pollutants; they also tend to reduce congestion, accidents, and emissions of greenhouse gases. Likewise, emission-reduction policies often result in reductions of multiple pollutants. For example, a three-way catalyst decreases emissions of NO_x, CO, and VOC, and the component parts of the catalyst cannot be allocated to the reduction of different pollutants.

Joint production presents a quandary for calculating cost-effectiveness estimates. In CEA, the criterion for assessing alternatives is “bang for the buck,” or the units of objective gain achieved per dollar expended. If a policy has several effects at once, then arriving at a cost-effectiveness estimate is not straightforward. One approach is to assign weights to each effect, but that just begs the question of what the weights should be. For evaluating policies that reduce multiple pollutants, the most frequently used approach is to divide costs by a weighted average of pollution reduction, where the weights are determined by the relative damages from the different pollutants (see for example, Ando et al. 2000, and California I/M Committee 2000). The results provide estimates of the cost per unit of pollution, where the units are standardized by the relative damages caused. This approach tends to assume constant relative damages from each of the pollutants. For example, in Ando et al. (2000) damages from a ton of NO_x reduced are assumed to be 2.5 times greater than the damages from a ton of HC reduced. In fact, damages are likely to be different in different regions, at different times, and at different levels of control.

Another way to deal with several joint objectives is to use results of other studies to net out the benefits of some of the objectives. McConnell (1990) and Lareau (1994) used this method

Policies that discourage driving not only reduce emissions of conventional pollutants; they also tend to reduce congestion, accidents, and emissions of greenhouse gases.

to net out the benefits of CO emissions from analysis of policies to reduce VOC emissions. But this approach requires that the benefits of at least some of the pollutants are known. If no benefit studies are available, researchers have used “alternative cost,” the least-cost alternative to removing the other pollutant (CO in the example above).

Nonpecuniary Costs. The costs of motor vehicle policies often cannot be measured by expenditures on pollution abatement or on other easily monetized activities. Sometimes they fall directly on motorists, imposing inconvenience or increasing risk. To estimate these nonpecuniary costs, the researcher must rely on indirect or inferential methods of estimating costs that are no different from the methods used to estimate benefits.

For example, the earliest U.S. emissions standards on new vehicles came into force with the 1973 model year. At the time manufacturers had not yet perfected the engine modifications and abatement technology needed to meet the standards with sacrificing performance. These vehicles were difficult to start and had poor acceleration, diminished fuel economy, and higher maintenance expenses. Bresnahan and Yao (1985) showed that these drivability problems were reflected for years afterward in the prices of used cars and were able to use those prices to estimate the additional cost imposed on motorists by the emissions standards. They found that the nonpecuniary costs were very high in the early seventies—\$865 per car in the 1973 model year—but gradually declined as new technologies were developed, so that 1981 vehicles actually enjoyed a drivability credit of \$130.

Another example of nonpecuniary costs can be found in vehicle emissions inspection and maintenance (I/M) programs. Here, a majority of the costs are not out-of-pocket but are denominated in motorist inconvenience. In a study of the Arizona I/M program, Harrington et al. (2000) estimate that 25% of the costs of the I/M program are incurred while motorists are queued up at the testing station.

Probably the most important example of nonpecuniary costs is the suspected effect of the Corporate Average Fuel Economy (CAFE) standards on accidents, which is discussed in more detail in Chapter Six.

Distribution of Benefits and Costs. As noted in Chapter Three, taxes to increase the cost of driving would probably increase welfare, assuming of course that the revenues from the tax increase are used to offset other, already existing taxes. And yet, around the world we can find very few examples of the use of tax instruments to correct environmental externalities, especially where motor vehicles are concerned. There are several reasons for this. One, which we discuss in more detail in Chapter Eight, is that a single vehicle’s contribution to these externalities is difficult to measure, which makes it difficult or impossible to implement policies that rely on such measurements. But even though there have been technological improvements reducing the cost of monitoring, there is still indifference—some might even say outright hostility—to the kinds of tax instruments that would improve welfare, so it would appear.

The distributional properties of economic incentive policies, which are ignored in assessments of efficiency, are the main reason for this anomaly. According to modern notions of welfare economics, efficiency is achieved if *potential* compensation is possible; *actual* compensation is not required. Thus a policy that economists would call “efficient” could end up hurting a great many people, possibly even more than it helped. There is good reason to think that many

efficient transportation policies would suffer from this characteristic.

The reason is not the usual one that is cited when considerations of equity are brought up in policy discussion, namely that the policy in question will adversely affect an already-identified disadvantaged group, such as the poor or the elderly. The concern with the impacts of policies on different social and income classes is what some have called *vertical equity* (e.g., Litman 1999), expanding on the economic concept of the same name. In principle, at least, it is possible to neutralize the adverse effects on vertical equity by redistributing tax revenues in the appropriate way.

Horizontal inequity—the unequal treatment of households that have the same income, race, or other readily identifiable characteristic—may be a more compelling explanation for the failure of economic incentive policies to make more headway in personal transport. Households and individuals differ greatly in their use of motor vehicles and in the emissions characteristics of the motor vehicles they use. Those who use vehicles heavily, either by a taste for travel or for a lifestyle that requires much driving, will be affected by social cost pricing (and its various approximations) much more than a household without those characteristics. Furthermore, unlike vertical inequities, it is much more difficult to devise a compensation scheme, because the household characteristics that help identify those most adversely affected are not easily observable.

In our view, the characteristics of horizontal equity go far to explaining why “welfare improving” policies are so difficult to enact in transportation as well as in other policy contexts.

Behavioral Consequences. The methods used in many studies define costs and benefits very narrowly and often ignore economic responses that could have important effects on cost-effectiveness or cost-benefit results. Because environmental policies cause prices and incentives to change, there are feedback effects on behavior and associated environmental effects. Ideally, these should all be taken into account in the analysis, but often they are not. For example, if a regulatory policy causes the price of new vehicles to rise (e.g., due to stricter pollution control requirements), motorists are likely to respond by buying relatively fewer new cars and holding older vehicles longer than without the policy. If this market response is ignored, the estimated emissions reductions will be too high, and cost-effectiveness of the control policy would look better than it really is. In an analysis of the introduction of electric vehicles in California, Dixon and Garber (1996) found that cost-effectiveness would be between \$5,000–\$845,000 per ton of emissions reduced, depending on the assumptions about how clean electric vehicles are relative to the existing fleet, and most important, on how quickly the new electric vehicles will penetrate the market.

In another example, Alberini et al. (1996) look at the cost-effectiveness of old car scrap programs and find that the cost per ton of HC removed would be higher if the scrap program is large and drives up the price of used cars in the region where cars are being recruited for scrapage. When used-car prices rise, drivers hold on to these vehicles longer, which partially offsets the emissions reductions from the scrapped vehicles.

Because environmental policies cause prices and incentives to change, there are feedback effects on behavior and associated environmental effects. Ideally, these should all be taken into account in the analysis, but often they are not.

Models of Vehicle Ownership and Use

There is a class of models that have been extensively used in the transportation literature to examine how vehicle characteristics and patterns of use change when operating costs or incomes change. For example, when regulations or market-based policies change the relative costs of driving, it is critically important to know how vehicle owners will respond to these changes to fully evaluate the effect of the policy on the environment. Knowledge of price elasticities is essential for understanding both the response to particular policies and their impacts on affected parties (motorists and potential motorists). Knowledge of income elasticities is less important for policy evaluation, but with world population heading for 11 digits, most of whom will live in currently poor countries striving for higher living standards, it is vital for understanding the scale and scope of future problems associated with auto use.

We distinguish between two subclasses of models: (1) models using aggregate data that are primarily (but not exclusively) concerned with fuel demand and (2) models using household data that attempt to model household decisionmaking explicitly.

Aggregate Models

Several fuel price elasticities appear in these models: elasticity of gasoline use (G), vehicle miles traveled (VMT) and fuel economy in miles per gallon (MPG).¹⁰ These three variables are related thus: $G = VMT/MPG$.¹¹ VMT is a function of c , the price per mile of travel, which in turn depends on the price p of fuel and the fuel economy, that is, $c = p/MPG$. If we write this relationship as logarithmic functions of logarithms, we have

$$\ln G = \ln (VMT(\ln c)) - \ln MPG(p), \quad (1)$$

which we can differentiate in order to write the elasticity of demand for gasoline as a function of the elasticity of demand for driving (VMT) and the elasticity of demand for fuel economy.

$$E_{G,p} = E_{VMT,c} (1 - E_{MPG,p}) - E_{MPG,p} \quad (2)$$

In the short run, if fuel economy of the fleet is fixed ($E_{MPG,p} = 0$), this formula says that the price elasticity of gasoline is the same as the elasticity of VMT with respect to its cost. Over time, if the elasticity of demand for MPG is greater than zero ($E_{VMT,p} \geq 0$) and the elasticity of demand for VMT is less than zero ($E_{VMT,c} < 0$), then $E_{G,p} \leq E_{VMT,c} < 0$. Fuel economy can respond to price in the short run by means of a reallocation of trips to more fuel-efficient vehicles in the existing fleet. In the long run it can result from stock turnover, as consumers turn to more efficient vehicles.

Ordinarily we would think of vehicle fuel economy as a dependent variable, but in the United States it has become an independent variable by statute. The Energy Policy and Conservation Act of 1975 mandated for each vehicle manufacturer a minimum "corporate average fuel economy" under which the average fuel economy of the vehicles sold by each manufacturer had to meet minimum requirements, beginning with the 1979 model year. We discuss CAFE again below, but for now we merely observe that the elasticity of a vehicle's fuel use with respect to MPG , with fuel price unchanged, is

$$E_{G.MPG} = -1 - E_{VMT,c} \quad (3)$$

If vehicle use remained constant, then this elasticity would be unity. But with better fuel economy, the fuel cost per mile declines, leading to greater vehicle use. This effect, whereby a part of the effectiveness of CAFE is given back, has been called the rebound effect. As shown in (3), the rebound effect is the elasticity of VMT to the fuel cost per mile, and most studies find it to be rather small, at about 10 to 20%.

The empirical literature on fuel demand models is vast. Ten years ago a review of studies of fuel price elasticity (Dahl and Sterner 1991a and 1991b)¹² found nearly a hundred studies, with more than 300 separate elasticity estimates using a wide variety of econometric approaches and data sources and representing some 20 countries. Subsequently Dahl (1995) surveyed an additional 39 studies that appeared after the earlier review. This later survey was limited to studies using data from the United States.

In all cases, the main dependent variable is gasoline demand. The dependent variables always include gasoline price and some measure of average household income. Nearly all the studies examined in the two reviews are aggregate studies. Several types of models are discussed:

- Static models. Usually based on non-cross-sectional data, these models estimate a single price and income elasticity parameters.
- Lagged endogenous models. Estimated on time series or panel data, these models contain a lagged dependent variable to distinguish between short- and long-run elasticities. The duration of the lag strongly affected the estimates, with short lags (one month or one quarter) producing very low elasticities, especially in the long run. Dahl and Sterner (1991a) discounted these estimates as being confounded with seasonal effects.
- Models with other lags. Other lag structures don't require the same rate of adjustment to price and to income changes.
- Models with vehicles and vehicle characteristics. These models included the number of vehicles or average vehicle characteristics, such as fuel economy.

Examination of the results of these studies suggests that gasoline demand is still moderately responsive to both price and income. However, in more recent years, summarized in the 1995 Dahl study, the elasticities have fallen substantially. This conclusion is illustrated in Table 4, which compares elasticity estimates in the two time periods for simple static models and for lagged endogenous models. The decline in responsiveness extended to other related phenomena. For example, Dahl (1995) reported that the mean price elasticity of average fuel economy (MPG) declined from about 0.4 to about 0.2.

Several explanations have been suggested for the decline, including improved data, rising incomes in many countries in the world, and the much lower fuel prices in the 1980s, which made

TABLE 4

Median Price and Income Elasticities of Gasoline Demand

	PRICE		INCOME	
<i>Static models – fuel use</i>				
Dahl and Sterner (1991a, 1991b)	-0.51		1.24	
Dahl (1995)	-0.18		0.39	
<i>Lagged endogenous models</i>				
	SHORT-RUN	LONG-RUN	SHORT-RUN	LONG-RUN
Dahl and Sterner (1991a, 1991b)	-0.26	-0.86	0.48	1.21
Dahl (1995)	-0.13	-0.65	0.19	0.72

Note: In Dahl and Sterner (1991a, 1991b), few studies contain data after 1980. In Dahl (1995) most studies contain data at least to 1985.

gasoline a smaller portion of the cost of driving. Moreover, the studies surveyed by Dahl (1995) used data from the 1980s, a period when energy prices declined substantially. At the same time, Gately (1992) found that the response of gasoline demand to both price and income is asymmetric, that is, sensitivity to price declines is much lower than to price increases. This asymmetry has suggested to some observers that the CAFE standards have been responsible for the decline in elasticity. When prices decline, mandatory CAFE standards reduce fuel costs as a share of the cost of driving; they have also reduced the availability of larger, less efficient vehicles.

Most of the studies examined in Dahl (1995) and Dahl and Sterner (1991a) were reduced-form models using aggregate data. In some models fuel demand was estimated directly from aggregate data on prices and transportation fuel use; in other models, energy demand is estimated indirectly by estimating the separate effects of fuel price on vehicle use, usually measured by vehicle kilometers traveled (VKT), and on fuel economy. Because fuel use is the product of fuel economy and vehicle use, its elasticity is the sum of these elasticities.

Income Elasticity. It is clear from Table 1 that higher GDP per capita is associated with larger per capita vehicle holdings. Many of the studies discussed above also compute income elasticities, but most have been limited to one or at most a few countries and have a limited range of years and incomes. Recently Dargay and Gately (1999) examined vehicle ownership patterns (measured in vehicles per 1,000 people) in 26 countries with widely varying per capita incomes, over a period that usually extended from 1960 to 1992. They found that income was by far the most important variable explaining vehicle ownership—more important than fuel price, vehicle price, infrastructure, or population density. They also found that the relationship of vehicle ownership to income in each country tended to be nonlinear and in fact S-shaped.¹³ (In no country was the historical record so complete as to see the entire S. In a country like the United States they observed the upper part, and in China the lower part.) Between 1992 and 2015, the long-run ownership elasticity falls from 0.24 to 0.04 in the United States and from 1.92 to 0.40 in a middle-income country like Korea, but increases from 1.34 to 2.16 in China.

Structural Models Using Microdata

Thanks to their high level of aggregation, the reduced-form models considered in this Chapter can be used to simulate responses to changes in policy. However, their use is limited to aggregate outcomes: changes in the total size of the fleet and average vehicle use. Evaluation of environmental policies usually requires more detailed information. Vehicles differ greatly in their characteristics, and these differences can affect both their attractiveness to motorists and their impact on the environment. Often the performance of a proposed policy will have very different effects on cars versus small trucks, old vehicles versus new vehicles, the number of vehicles owned by various types of households, the level of vehicle use, and the implications of all these outcomes on various fuels.

To examine those effects we need more detailed models that explicitly represent individual choices among types of vehicles and their use. In the past couple of decades there have been numerous attempts to build structural models of transportation behavior using microdata from household surveys. These models attempt explicitly to account for all kinds of transportation choices made by households, applying the discrete and discrete-continuous modeling frameworks pioneered by Daniel McFadden and colleagues. McFadden brought together two fairly

new ideas in the social science of the time: the idea of random utility, which had appeared in the psychological literature about 30 years before, and the logit model, which economists, transportation planners and psychologists had borrowed from biostatistics in the early 1960s and begun to use in empirical studies of processes with discrete outcomes. In McFadden (1973) it was shown that under particular assumptions about the structure of the error term, the empirically convenient multinomial logit model applied to individual decisions was equivalent to a random utility model (RUM).

The multinomial logit model was not, unfortunately, a “flexible” functional form; that is, its use imposed strong structural assumptions. In particular, it had the property of “independence of irrelevant alternatives” (IIA). If an additional choice is introduced, the probabilities of choosing each of the previously available choices are changed by the same multiple. An example is the famous “red bus/blue bus” problem. Consider a mode split model that assigns a probability of one-half to each of two modes: driving and taking a red bus. If now a third choice is added, a blue bus otherwise with the same characteristics as the red bus, then the model assigns a probability of one-third to each alternative, an obviously undesirable property.

An important theme in the recent history of discrete choice models has been the efforts to get away from the IIA assumption while maintaining the connection to utility theory. Until recently the most successful and still the most widely used alternative was the nested logit model, introduced by Ben-Akiva (1972) and given a RUM justification by Ben-Akiva and Lerman (1979) and McFadden (1979). A nested logit model consists of a decision tree of multinomial logit sub-models, each linked sequentially by an “accessibility” or “inclusive value” term that indicates the overall attractiveness of the alternatives in the next lower nest. For example, an individual decides whether to take a trip for a given purpose, and if so, to which destination, and finally, by which mode. Nested logit is a natural fit for hierarchical decisionmaking, which is common in transportation decisions.¹⁴

Discrete choice models are well suited to models of household decisionmaking, and not least those decisions affecting transportation, because so many transportation choices are discrete. In particular they have been used in the following ways.

Urban Travel Demand. Predicting urban travel demand, and particularly mode choice, was one of the earliest applications of the random utility model (Domencich and McFadden 1975; McFadden 1978). Today, transportation planners generally use nested (and non-nested) logit models to predict mode choice and, less commonly, trip destination parameters for use in detailed, spatially disaggregated urban travel models. (For example, the Washington, DC, metropolitan area model has 2,100 zones and 18,000 travel links.) However, commercially available transportation planning packages still rely on a mix of behavioral and heuristic devices. For example, many use a “gravity” model for distributing trips among possible destination zones. Rather than rely on assumptions about individual behavior, a gravity model bases its estimate of trips between zones A and B on aggregate productions and attractions in each zone, plus some measure of the distance or travel time between them. The distance parameter is then altered to fit the observed number of trips between the zones.

Locally, these models now produce *official* estimates of total travel, levels of congestion, mode share, and, when coupled with a mobile-source emissions factor model, aggregate vehicle emissions in U.S. metropolitan areas over the duration of the transportation planning cycle, which

is generally 20 years or more. These uses have made transportation planning models local sources of controversy among planning officials and interested parties, especially the environmental and business communities.

Empirical models of urban travel behavior that are fully based on individual behavior have been developed but are still largely research tools (Ben Akiva and Bowman 1998). These models are entirely “bottom up” in their reliance on individual behavior. In addition, recent work has attempted to be more explicit about the manner in which transportation contributes to household utility. For example, recent work has moved beyond “trips” as a unit of analysis. Newer models are “activity-based,” with utility-producing activities outside the home organized into “tours.”

Integrated Land Use and Travel Models. These models aspire to a fully integrated model of urban transportation and land use. Unlike the pure transportation model, in which housing and employment locations are taken to be exogenous, these models endogenize housing location decisions. For example, Watterson (1993) attempts to link commercially available transportation and land-use modeling tools in an interactive framework. Ben-Akiva and Bowman (1998) and Eliasson and Mattsson (2000) link models in a coherent microeconomic framework. These models could help sort out the causal relationships between highways and land use (discussed in detail in Chapter Seven), provide insight into the social costs of various types of land-use development, and assist in evaluation of land use and transportation policies intended to curb suburban sprawl.

Vehicle Ownership and Use Decisions. RUM models have been extensively used to study vehicle holdings and use by households. These models generally rely on the discrete-continuous model developed by Dubin and McFadden (1984) and, in the motor-vehicle context, Train (1986).

This type of model has been used in numerous environmental policy studies requiring detailed characterization of vehicles and their use. For example:

- Demand for alternate-fuel vehicles: Train (1986), Bunch et al. (1993), Brownstone et al. (1996), Leiby and Rubin (1997), Rubin and Leiby (2000).
- Effect of density, transit availability, and other land use characteristics on vehicle ownership: Kockelman (1997), Schimek (1996), Walls et al. (2002), Bento et al. (2002).
- Simulations of CAFE policy: Goldberg (1998).

The discrete-continuous models of vehicle ownership and use also produce elasticities of vehicle ownership and use with respect to vehicle and fuel price, income, and fuel economy. Short-run fuel price and income elasticities can be determined directly from the conditional VMT equation; in a log-log specification they are simply the coefficients. However, determination of the long-run elasticities is more complex, requiring computation of expected vehicle use and ownership using the conditional probabilities estimated in each nest of the conditional logit model.

When the short-run elasticities are compared with the elasticities estimated using aggregate data and the simpler methods in this Chapter, the RUMs tend to have, with some exceptions, higher fuel price elasticities. A possible explanation for the disparity is that many of the studies using RUMs have used cross-section household survey data, including those by Train (1986),

Berkovec (1985), Walls et al. (1993), and Bento et al. (2002). The elasticity estimates in cross-sectional RUMS are probably a blend of the short and long run—short-run because vehicle holdings and other household characteristics are effectively held constant, but long-run because there are some adjustments such as in household location and commute length. Therefore they are likely to have higher elasticities than estimates using time-series aggregate data.¹⁵

Notwithstanding the hundreds of studies of fuel and vehicle demand that have been done over the last several decades, there is still no consensus on the responsiveness of fuel use to price changes. There are still questions about whether the data and various methods have identified the true underlying elasticities.



CHAPTER FIVE

Conventional Pollutants

Vehicle emissions contribute to a range of local and regional air pollution problems. We summarize the major pollutants and what is known about their damages, and then present evidence about the extent and causes of these air quality problems in regions around the world. Although vehicle ownership and use have been increasing in all parts of the world and there is some commonality in pollution issues, the variations in the levels and causes of pollution, and the extent of controls among countries are striking. There are differences in the pollution problems themselves, in the use of policy tools, in enforcement and institutional settings, and in the penetration of new technologies. We briefly review the experience with regulatory and control policies in developed and developing countries and then focus on the opportunities for market-based policies.

Vehicle Pollutants, Air Quality, and Damages

Some of the damaging pollutants from vehicles come directly from fuel that is not completely combusted, others are formed by chemical processes during combustion or other vehicle processes and then released, and still others are formed by chemical processes that occur in the atmosphere. The processes are complex and can vary with types of fuels and engines, and other factors including temperature conditions and natural background levels of chemical compounds.

The emissions patterns of gasoline and diesel engines are quite different. For gasoline engines, the pollutants of most concern are carbon monoxide, volatile organic compounds, oxides of nitrogen, and airborne lead. From diesel engines emissions of CO and VOCs are low, but NO_x emissions are comparatively higher than with gasoline engines. Diesel engines are also major emitters of fine particulates.

The link between vehicle emissions, pollution formation, and damages to human health, vegetation, and materials are fairly well understood for some pollutants (ozone and CO) and not for others (fine particulates). We briefly describe the major air quality problems associated with vehicle emissions.

Ozone (O₃) is not discharged from vehicles but is formed on hot sunny days through a series of complex chemical reactions involving many atmospheric contaminants, in particular NO_x and VOCs. Vehicle emissions of both hydrocarbons and NO_x can contribute to ozone formation, but

the effects on air quality vary with weather conditions and total emissions of both HC and NO_x from all sources, including natural sources. In general though, ozone, or urban smog, is one of the most prevalent vehicle-induced pollution problems and can form locally or down-wind of the sources of the emissions. Damages to human health include changes in pulmonary function, especially during exercise, and impaired defense against bacterial and viral infections. Short-term effects can include eye, nose, and throat irritation; coughing; and chest tightness. Asthmatics appear to be particularly affected (Romieu 1992, Krupnick et al. 1990, NRC 2002b).

Total Suspended Particulates (TSP) from vehicles fall into two categories of concern: coarse particles (PM₁₀) are those with a diameter of 10–2.5 μm, and fine particles (PM_{2.5}) with a diameter of 2.5 μm or less. PM_{2.5} has more serious health consequences than PM₁₀ because it can reach lung tissue (PM₁₀ tends to be deposited higher in the respiratory tract) and can remain imbedded in the lungs for long periods. Most of the PM emitted directly by diesel vehicles falls in the latter category of fine particles, with a diameter of less than 1 μm.

PM_{2.5} can also be formed indirectly by chemical reactions in the atmosphere involving other pollutants such as SO₂, NO_x, and HC. Sulfate aerosols, particulates formed from SO₂, appear to be especially harmful and have been linked to such respiratory diseases as pneumonia, asthma, and bronchitis. In general, long-term exposure to PM_{2.5} in the United States has been associated with higher mortality rates in epidemiological studies (Dockery et al. 1993). However, there is still not complete understanding of how vehicle emissions contribute to various types of PM 2.5 formation, and which is more damaging (Onursal and Gautam 1997).

Lead is added to gasoline to raise octane levels. When gasoline engines use leaded fuel, much of the lead is released through the exhaust and forms fine particles (PM₁₀) in the ambient air. Lead is absorbed in human tissues and organs, and in levels found in urban areas where leaded gas is the primary fuel, has been shown to have adverse health effects on both children and adults (Romieu et al. 1992). For example, epidemiological studies have found evidence linking higher blood levels of lead and decreased IQ test performance for children (USEPA 1990). The share of leaded gasoline in total gasoline fuel has declined in many parts of the world in recent years (in the United States and some countries in Europe it is no longer used), but in some countries it remains a major component of the gasoline fuel supply.¹⁶

Of these air quality problems, particulates are emerging as one of the most pervasive and serious health threats, in both developed and developing countries (Small and Kazimi 1995; Cropper et al. 1997). Lead and later ambient ozone have been the focus of past regulatory attention and concern over health issues, but particulates, which have not been as heavily regulated in the past, are a growing focus of regulatory policy around the world.

Table 5 summarizes what is known about the severity of various pollution problems for many of the world's largest cities. Total suspended particulates appear to be the most severe problem. More than half of the cities have levels that exceed World Health Organization (WHO) guidelines by more than a factor of two. In recent years, it has been clear that health damages from exposure to particulate levels are relatively greater, on average, than exposure to ambient ozone levels. Of course, it depends on the city and current exposure levels, but a recent study concluded that across a sample of U.S. cities, acute PM₁₀-related mortality appears to be about four to five times greater than ozone-related mortality (TRB 2002). Lead exposure presents a serious health

TABLE 5

Air Pollution Levels in Selected World Megacities

	Population [†]	SO ₂	TSP	Lead	CO	NO ₂	O ₃
Bangkok	10.3	*	***	**	*	*	*
Beijing	11.5	***	***	*	—	*	**
Buenos Aires	13.0	—	**	*	—	—	—
Cairo	11.8	—	***	***	**	—	—
Calcutta	15.9	*	***	*	—	*	—
Jakarta	13.2	*	***	**	**	*	**
London	10.8	*	*	*	**	*	*
Los Angeles	10.9	*	**	*	**	**	***
Manila	11.5	*	***	**	—	—	—
Mexico City	24.4	***	***	**	***	**	***
Moscow	10.1	—	**	*	**	**	—
New York	16.1	*	*	*	**	*	**
Rio de Janeiro	13.0	**	**	*	*	—	—
Seoul	13.0	***	***	*	*	*	*
Shanghai	14.7	**	***	—	—	—	—
Tokyo	21.3	*	*	—	*	*	***

Source: ARIC (1996).

† Estimated population in 2000 in millions

— Inadequate data

* Low pollution, WHO guidelines are normally met (short-term guidelines may be exceeded occasionally)

** Moderate to heavy pollution, WHO guidelines exceeded by up to a factor of two (short-term guidelines exceeded on a regular basis at certain locations)

*** Serious problem, WHO guidelines exceeded by more than a factor of two.

TABLE 6

Contribution of Motor Vehicles to Urban Air Pollution

Percent of total air emissions by pollutant

	SO ₂	TSP	CO	HC	NO _x
By Country					
United States	-	-	66	48 ^a	43
Germany	6	-	74	53 ^a	65
United Kingdom	2	-	86	32 ^a	49
By City					
Budapest	12	-	81	75	57
Cochin, India	-	-	70	95	77
Delhi	13	37	90	85	59
Lagos	27	69	91	20	62
Mexico City	22	35	97	53	75
Santiago	14	11	95	69	85
Sao Paulo	64	39	94	89	92

Sources: Small and Kazimi (1995); World Resources Institute (1997).

a: Data are for VOCs (a component of HC)

problem in several of the cities in Table 5, because lead gasoline is still used exclusively in most parts of Africa and in the Middle East. Los Angeles, Mexico City, and Tokyo have the most serious ozone problems.

For each of these pollution issues, vehicles are one of many factors influencing local air quality. Emissions of the underlying pollutants can come from transport, power, industrial, or residential sources. In fact, the contribution of vehicles to total emissions varies considerably. For example, the percentage of NO_x contributed by vehicles is higher in developing than developed countries for a number of reasons: diesel vehicles make up a large share of the fleet; there are large numbers of two-wheeled vehicles such as motorcycles that tend to have high NO_x emissions; and the stock gasoline vehicles tend to be older and therefore without catalysts that reduce NO_x emissions. Table 6 shows the shares contributed by motor vehicles to total emissions of the pollutants listed for a range of countries and cities.

Vehicle Emissions: Experience and Policy

We briefly examine vehicle emissions policies in the United States, Europe, and selected developing countries.

Developed Countries

The United States was the first country to take serious steps to reduce air pollution from motor vehicles, and billions of dollars have been invested trying to reduce emissions of certain vehicular pollutants since the mid-1970s. California, often acting independently, has led efforts to impose strict controls on vehicle emissions. For the most part, the regulations in the United States have focused on meeting uniform standards for new-vehicle emissions rates.

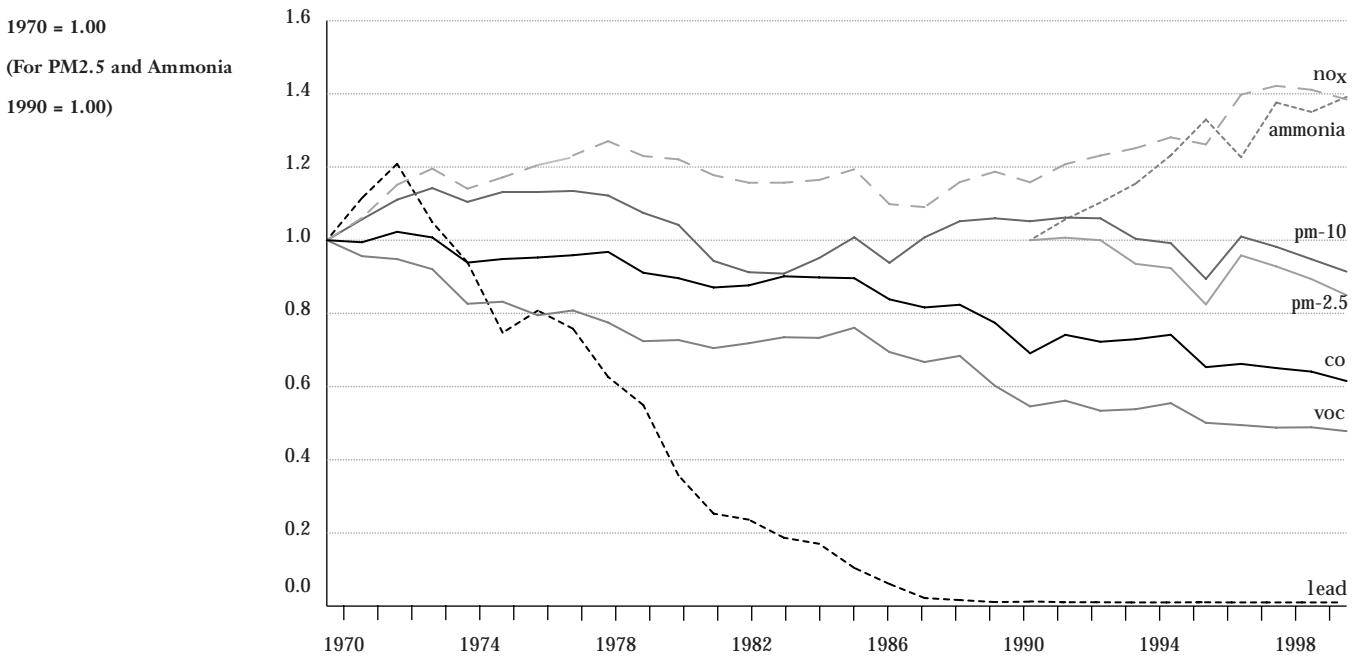
The focus in the 1970s and 1980s was on a handful of pollutant emissions considered in the Clean Air Act of 1970 to be the most important. These included HC (VOCs), lead, particulates, NO_x, and CO.

First in California and then in the rest of the country, the primary policy has been to require increasingly stringent new-car controls on light-duty vehicles, focusing initially on HC and CO, and, in more recent years, on NO_x.¹⁷ The requirements were in the form of new-car standards that had to be met by all vehicles in the fleet, with light-duty trucks having more lenient standards than cars (reductions of 80 to 95% over uncontrolled levels of 1970 vehicles were required). Figure 3 shows that since 1970, emissions of several pollutants have declined despite an average annual increase in VMT of more than 3%. Lead has been phased out completely, and CO and VOC emissions have fallen. However, these controls have been uniform across vehicles and regions of the country and target only emissions from new vehicles, and as such, they may have relatively high costs and some unintended consequences.

For example, Gruenspecht (1980) argued that imposing strict controls on new cars may create a “new-source bias” that could actually increase emissions in the short run. If new vehicles cost more—by some estimates up to \$1,000 more per vehicle, or 5% of the cost in the early 1980s (McConnell et al. 1995)—then motorists are likely to hold on to their old vehicles longer, thereby increasing the average age of the fleet. Depending on the elasticity of demand for new vehicles and the rate of substitution of old for new vehicles, emissions could initially increase as a result of this policy. Gruenspecht (2001) and Kazimi (1997) have both argued that such a short-

FIGURE 3

Index of Air Pollutant Emissions from Transportation Sources: United States



Source: USEPA Office of Air Quality Planning and Standards, *National Air Pollutant Trends*.

term emissions increase could occur when zero-emission vehicles or alternative-fuel vehicles are introduced, depending on the cost of the new vehicles relative to the existing gasoline fleet.

The Clean Air Act (1970) and its initial amendments (1977) focused primarily on pushing manufacturers to produce cleaner new cars. By the late 1980s, however, there was growing evidence that the vehicle emissions problem would not be solved by new-car controls alone. The standards for emissions levels of HC, CO, and NO_x were not being maintained throughout the life of the vehicle, and in fact, average emissions of the fleet on the road were roughly two to three times higher than the new-car standards (Crandall et al. 1986; Harrington et al. 2000). Further, evidence began to accumulate that emissions from the on-road fleet were skewed, with a small number of high-emitting vehicles contributing a disproportionate share of the total emissions (Beaton et al. 1992).

To deal with in-use emissions, vehicle emissions I/M programs were required under the 1990 Clean Air Act amendments for the most polluted urban areas. However, these programs have met with mixed success over the past decade. Motorists have little incentive to comply with the requirements, and enforcement is difficult because of the large number of vehicles. Further, the most polluting vehicles appear to have the highest repair costs, giving their owners the greatest incentive to avoid compliance. Nevertheless, I/M has been found relatively cost-effective (Harrington et al. 2000) and could be made more so by better-targeted efforts to find and repair (or

scrap) only those vehicles that are likely to have the highest emissions (Harrington and McConnell 2000; NRC 2001).

Under recent regulatory changes, I/M program requirements vary across urban areas depending on the severity of local pollution levels. This is consistent with the costs and benefits of additional controls and how they vary by region (Oates and Schwab 1988) and reflect recognition of steeply increasing marginal costs once substantial reduction has been achieved. For many regions, the first 80% of vehicle emissions reductions has been reached, and the marginal cost of reducing the last 20% may be quite high. Additional reductions in HC should be targeted to areas where the associated benefits are high—in areas that have high pollution or exposure levels.

Another early focus of the Clean Air Act of 1970 was leaded gasoline, which interfered with efficient working of the new pollution control equipment.¹⁸ Starting in 1974, lead was to be phased out of gasoline. The costs of the phase-out were high, so the policy evolved into a market-based program in which refineries could trade or bank “credits” for exceeding some prescribed level for lead reduction, with required total reductions in lead increasing over time (Kerr and Maré 1997; Kerr and Newell 2001). Lead was completely phased out by 1989, and the trading mechanism was estimated to have saved \$226 million of the transition cost.

After the intense efforts to reduce lead in fuel and remove HC and CO from vehicle exhaust, attention has gradually turned toward other pollution issues. There was growing evidence that early efforts to reduce ozone were too simplistic, and reductions in HC in certain areas were said to have actually increased ozone levels when NO_x was the limiting pollutant (NRC 1991). Policies in the 1990s have focused more on NO_x reductions for ozone control. And, recent years have witnessed growing concern over emissions of particulates and toxics, which, until now, have been virtually ignored in regulatory policies. Diesel-fueled vehicles are the major source of PM_{2.5}, but they have been relatively unregulated compared to gasoline-fueled vehicles.

Recent legislation in the United States, however, has focused on diesel engines and diesel fuels. New regulations on both heavy-duty trucks and on light-duty gasoline vehicles require the use of catalysts to reduce NO_x (and particulates from trucks) to very low levels. However, to operate effectively, these controls require fuels with very low sulfur levels (MECA 2000). The engine technology and the fuel type must thus change in tandem.¹⁹ This is, as in past regulations of vehicle emissions in the United States, a uniform technology-based regulation for new engines. It calls for a 90% reduction in particulates and 95% reduction in NO_x over current levels by 2007 models, and requires an 80% conversion to low-sulfur diesel fuel in all regions by 2006 (sulfur levels must be less than 15 parts per million (ppm) compared with 500 ppm currently).²⁰ We discuss alternative, more market-based policies to achieve the emission reduction goals of the diesel program below.

In Europe, policy developments have paralleled those of the United States. The primary tool has been strict new-car standards, which were implemented coincident with U.S. policy. European countries have also relied heavily on new-car standards, and although they have been slower to implement requirements with equivalent stringency, standards are now being gradually harmonized between the United States and the European Union.

There are some important differences across the Atlantic in transport policies, however. Countries in Europe have had a larger public-sector role in transportation, with much higher tax rates in general, and higher taxes on fuels in particular, as shown in Table 2 above. Over the years,

the higher taxes have been used, in part, to subsidize rail transport. Nevertheless, rail use has been declining as a share of transportation in kilometer miles traveled (KMT), for both passengers and freight, and in some countries it has been declining in absolute levels (USDOT 1996).

The use of diesel-fueled vehicles has increased in Europe in recent years. Although much of this increase has been for freight transport, diesel passenger-car registrations rose from 14% of all car registrations in Europe in 1990 to 21% by 1993 (Schipper et al. 2001). In part, this growth reflects lower tax rates for diesel fuel than for gasoline in many countries, and as a result diesel fuel use has grown at a much faster rate than gasoline use through the 1980s and 1990s. The problem is that diesel engines, with current technology, have both higher particulate and higher NO_x emissions than gasoline engines. Mayeres and Proost (2001) have argued that the relatively low taxes on diesel fuel and the higher taxes on gasoline are inefficient because these external effects are not taken into account.

In the European Union as in the United States, recent regulatory efforts have been directed toward making diesel engines and fuels cleaner. Diesel engines manufactured after 2006 in the United States and 2005 in Europe are mandated to be more than 90% cleaner in emissions of NO_x and particulates, but these levels can be achieved only if diesel fuel is very low in sulfur content. Countries in Europe have taken the lead on moving to low-sulfur diesel by implementing differential environmental taxes on low- and high-sulfur fuels (Arthur D. Little 1998). Sulfur levels in Sweden and Finland have been reduced from about 300 ppm to 50 ppm in more than 90% of the fuel, through the introduction of differential taxes of about 5¢ (euros equivalent) per liter from the early to mid-1990s. Sweden's tax was designed to be revenue neutral, but to date revenues have decreased because the conversion to low-sulfur fuel has been more rapid than expected (Arthur D. Little 1998). The United Kingdom, Germany, and Belgium are now following suit, and introducing their own versions of the differential tax.

In part, the willingness to initiate new policies is prompted by the increasing levels of NO_x and TSP levels in many European cities over the past decade (USDOT 1996). Some cities have used innovative policies to try to reduce transport-related urban pollution. Stockholm uses a toll system to steer traffic around the city and away from the central district. Milan restricts driving in the downtown area on days when air quality exceeds certain levels. Other cities are considering similar policies, in which only certain vehicles can drive in the downtown areas (only vehicles equipped with a specified level of pollution control equipment), or all except certain exempt vehicles are restricted from entering on high-pollution days. In various forms, these so-called low-emission-zone policies have already been implemented in Germany, Sweden, and Switzerland (Rapaport 2002). Sweden currently has four cities with environmental zones, including Stockholm, where diesel trucks and buses of a certain size are not allowed to travel in the zone without advanced emissions and noise control systems (Rapaport 2002). Analyses of such policies in Stockholm (Rapaport 2002) and London (Carslaw and Beevers 2002) find that they are likely to have only very small effects on ambient nitrogen dioxide (NO₂) concentrations, in part because of the non-linear chemistry of NO₂ formation, and also because traffic will be displaced to other parts of the city where pollution levels may become worse.

As in the United States, the type and severity of pollution problems in western Europe vary regionally. The southern countries tend to be poorer, with a somewhat older vehicle fleet, less stringent control policies, and weather conditions that make certain pollutants (such as ozone) more severe.

Developing Countries

The nations referred to here as the developing countries actually differ a great deal in income and level of economic development. They include countries as economically diverse as Brazil and Bolivia in South America to Senegal and South Africa in Africa. As the data above show, many of the world's most polluted cities are in the developing world. The general policy tools available and the problems of regulating such large numbers of emissions sources are the same as in the developed world, but the developing countries have a different fleet mix—larger shares of older vehicles, of diesel buses and trucks, and many more motorized bikes. Although data on vehicle emissions in most developing countries are scarce, there is some evidence that vehicle emissions of particulates and SO₂ are a significant source of air pollution problems in major cities and present the most serious health hazards (see Table 5 above). Lead in gasoline remains an important health issue in the relatively small number of countries that still use leaded gasoline.

The average annual rate of growth of vehicle ownership in these countries was about 6.5% through the last three decades, compared with 2.5% in the United States. During this period, the share of the world's vehicle fleet held by countries outside the Organization for Economic Co-operation and Development (OECD) increased from 14% to 24%.²¹ Most of this growth in vehicle holdings comes from imports of used vehicles. As a result, the average age of the fleet is 12 to 15 years in many developing countries, compared with 6 to 8 years in the OECD (USDOT 1996). Emissions per kilometer traveled tend to be high in developing countries, both because older fleets mean that more vehicles were built to lower design standards and because vehicle emissions tend to increase with vehicle age. The share of motorcycles in many developing countries, particularly in certain countries of Asia, is very large (WBCSD 2001), and the two-stroke engines on most motorcycles have high emissions of particulates.

Fuels are a major issue in the developing world. Although leaded gasoline is being phased out in most countries, it is still the primary gasoline fuel in some areas such as the Middle East and Africa. Sulfur in fuel, particularly diesel fuel, poses one of the greatest health risks for many developing countries (Onursal and Gautam 1997). Sulfur levels in diesel fuels were increasing in many parts of the world during the 1980s and early 1990s (USDOT 1996), and diesel fuel is heavily subsidized in many places (such as in Manila; see Shah and Nagpal 1997). If policies to introduce lower-sulfur fuels result in higher prices or reduced supplies, drivers may switch to even higher-sulfur heating fuels to power vehicles and emissions could actually go up (Harrington and Krupnick 1997). In general, it is not yet well understood how policies to promote low-sulfur fuel in some markets or in some grades may simply shift the sulfur in fuel to other markets and other grades.

Developing countries have implemented many policies to cope with worsening vehicle pollution in urban areas. Some countries have attempted to subsidize retrofit of diesel trucks and buses with particulate traps and other abatement devices. However, the success of current retrofit

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programs for diesel engines apparently depend on the sulfur content of diesel fuel. At sulfur levels between about 30 and 150 ppm, ceramic particulate filters reduce emissions only slightly, and at high sulfur levels (2,000 ppm and higher, the levels of sulfur in diesel fuel in most developing countries), sulfate emission levels actually increase substantially with these traps (US DOE 2002).

In Mexico, ambient ozone levels are among the highest in the world, and particulate levels are also very high.²² Most of the PM₁₀, HC, and NO_x emissions come from cars, trucks, buses, and taxis (Onursal and Gautam 1997). Average vehicle emissions rates are high, primarily because an estimated 50% or more of the vehicles were built before 1991 and lack catalytic converters. The combination of older vehicles remaining in circulation and high growth rates in new vehicles (recently close to 10% a year) has created a fleet of more than 3 million in Mexico City (OECD 1999).

Mexico has tried an array of regulatory policies to deal with its severe air pollution, from incentives for retrofitting truck and bus engines, to fuel taxes and fuel regulations, to transportation management policies. One of the most well known and controversial is the *boy no circula* (don't drive today) policy implemented in 1989, under which each nonexempt vehicle is barred from the city on one day of the week.²³ Exemptions from this requirement have provided an incentive for retrofitting some truck and bus engines with pollution-control equipment, but the policy has also had unexpected consequences. Many households purchased a second (often older) vehicle to circumvent the restriction, and there is evidence that vehicles kilometers traveled may have actually increased as a result of the policy (Eskeland 1997).

Singapore has imposed high fees on vehicle ownership and use (the fees are often greater than the cost of a vehicle). Despite these high prices, vehicle ownership continues to grow rapidly, so Singapore is now considering a quota system to limit vehicle ownership (see Chapter Four). The introduction of mass transit systems has also not been as effective for reducing vehicle use as many had hoped in developing countries. For example, the new underground rail system in Seoul, South Korea, has not attracted as many riders as expected (OECD 1995). It has also been argued that policies in some countries have tended to promote auto use and discouraged non-motorized and low-cost forms of private transit (Hamer and Linn 1987).

Market-Based Policies for Pollution Control in Urban Transport

Market-based policies for vehicles have long been suggested, starting with White (1982) who proposed an ideal system of effluent fees, or an alternative marketable emissions rights system. However, few market-based policies to reduce vehicle pollution have been implemented anywhere in the world. Difficulties of implementation and enforcement, and political resistance to price or tax policies, at least in the United States, might explain why such instruments are missing from the regulatory toolkit for fighting urban air pollution. Interest in such policies is increasing around the world, however, and some initial applications look promising.

Although the main attraction of emission fees or tradable permit systems is their efficiency properties, they have other useful properties that should be noted. One of the most important is cost revelation (Johansson-Stenman 1999). In a regulatory or command-and-control system it is very difficult to determine the true cost of environmental regulations, but with economic incentives the marginal cost of compliance is the tax rate or the permit price.²⁴ Thus, the cost

of control at the margin is known with certainty and provides useful information for comparing sources and for assessing cost trends over time.

Market-based policies can be either broadly applied or narrowly targeted. Policies narrowly targeted to a specific environmental problem will be the most efficient if there are no monitoring and enforcement costs, since they provide the most direct incentive to reduce emissions. If the goal were to reduce NO_x emissions, for example, a tax on each vehicle's on-road NO_x emissions would be the most efficient. However, it is difficult to monitor vehicle-by-vehicle NO_x emissions for millions of vehicles,²⁵ so broader tax policies have been suggested, such as a tax on fuel, or a tax on vehicle type that varies with pollution-control equipment. These taxes are more feasible, but not as efficient. They provide the incentive to use less fuel or install pollution-control devices, respectively, but do not tax the vehicle's total NO_x emissions over time. Various studies have considered how broader taxes could be designed and how policies could be combined to yield results that mimic efficient outcomes; we discuss their results below.

Market policies can also be implemented as price-based or quantity-based instruments (the latter are often referred to as cap-and-trade programs). Weitzman (1974) showed that price and quantity instruments give equivalent results only when abatement costs are known with certainty. If costs are uncertain, then taxes are more efficient if and only if the marginal benefits of abatement are elastic. When marginal benefits are inelastic—for example, when increasing levels of pollution cause little damage up to a threshold beyond which damages suddenly become very large—a quantity instrument is preferred. Weitzman's results further suggest that a quantity-based command-and-control standard could easily be more efficient than an emissions fee if the possibility of catastrophic damages exists.

The costs of controlling most conventional pollutants, such as NO_x, HC, and particulates from diesel engines, are probably known with a fair degree of certainty. However, the lifetime effectiveness of controls on, for example, diesel engines is uncertain at this point, suggesting some uncertainty in the costs per unit of control. The marginal benefit of controlling particulates is widely found to be relatively elastic, suggesting that price controls may be preferred to quantity-based controls for diesel vehicles. Europe has moved toward price-based policies for reducing sulfur in diesel fuel as a way of reducing NO_x and particulates from diesel vehicles (see discussion above). The United States has opted for a quantity-based regulatory approach, requiring refineries to convert a certain percentage of their fuel to low-sulfur by a target date (note that this is not a pure quantity standard), with some allowances for trading fuels by sulfur level among refineries (USEPA 2000).

Tax Policies

Direct taxes based on the damages from vehicle emissions would be an efficient instrument for pollution control. Polluters (drivers) would respond efficiently to the direct signal about the external costs of the pollution they cause. However, taxing emissions directly is both technically and politically very difficult. Emissions vary by individual vehicle (size and type of vehicle), and

Policies narrowly targeted to a specific environmental problem will be the most efficient if there are no monitoring and enforcement costs, since they provide the most direct incentive to reduce emissions.

over time (control equipment ages or breaks down), by how a vehicle is maintained, by fuel use and type, and by miles driven (see Chapter One). There is no proven technique yet to monitor emissions as vehicles are driven.²⁶ Even if emissions could be measured and taxed, enforcement would be difficult and unpopular given the millions of vehicles in every urban area.

However, there has been a good deal of theoretical attention to defining an optimal tax, and then examining second-best alternatives to it. Eskeland (1994) derives a first-best emissions tax, and examines combinations of possible taxes and mandates that can mimic this tax. Eskeland and Devarajan (1996), in an application of the analysis to Mexico City, show how combining a vehicle I/M program and new-car controls with certain taxes can get very close to the optimal emissions tax. They also show that the mandated controls then being suggested for Mexico City could be 25% more expensive if not implemented in tandem with a gasoline tax.

Innes (1996) and Fullerton and West (2002) both use general equilibrium models to derive an optimal emissions tax and explore alternatives. Fullerton and West (2002) examine a range of tax schemes, including a pure emissions tax, a gas tax, and a tax on engine size and pollution-control levels. They find that a gas tax that varies at the pump with the type of vehicle (engine size and pollution characteristics) can yield results similar to the optimal emissions tax.

Harrington et al. (1998) compare emissions fees with alternative policies, focusing on different aspects of the problem. Although some of the analyses described above allow for variation in engine size and pollution-control equipment, they do not account for variation in emissions or emissions measurements within these categories. In fact, on-road emissions vary a great deal across vehicles, even for vehicles with the same pollution-control equipment. Harrington et al. (1998) compared policies currently in use, specifically I/M programs to emissions fees, and included uncertainty in how well emissions are measured and in how well repairs reduce emissions. They find that fees are significantly less costly than I/M alternatives when drivers are well informed about the effectiveness of potential repairs. However, when there is uncertainty about repair effectiveness and any associated improvements in fuel economy, fees fared little better than the mandatory I/M program.

Harrington et al. (2000), using a different model, examine the relative importance of factors that might cause emissions fees to look either more or less similar to current mandatory policies, such as I/M. They look at the effect of factors such as transactions costs, consumer myopia, and uncertainty about the effectiveness of repair on the outcomes for different programs. Oates et al. (1989) derived more general results and showed that the efficiency differences between command-and-control policies and market-based regulation are quite complex, and may depend on a number of different factors. For example, command-and-control policies often result in greater emissions reductions than market-based policies, and may therefore yield additional benefits.

Sevigny (1998) examines alternative emissions tax policies, in which the emissions tax depends on measured emissions rates of the vehicle and miles driven. She simulates the effect of an emissions tax on total regional emissions, government revenues, miles driven, and the vehicle fleet. She looks at the effects of replacing part of the gas tax with an emissions tax, and finds such a policy to be welfare enhancing. Harvey (1994) compares distance and emissions-based pollution fees with a gasoline tax in a study of alternative pollution-reduction policies in the Los Angeles and San Francisco regions. He finds that the distance-based policies do not have much effect on emissions, and the emissions-based policies do not have much effect on miles driven.

In another applied welfare analysis, Jansen and Denis (1999) broaden the scope of the analysis to examine tax and other policies for reducing both carbon dioxide (CO₂) emissions and conventional pollutants in an application to the European case. They find that the best policy for CO₂ emissions is a fuel tax, with a supplementary differential fee on vehicles of different sizes (the latter tax is to correct for myopia on the part of drivers about the value of better fuel efficiency). For conventional pollutants, they find that the best combination is an emissions-based kilometer tax and a new vehicle purchase tax based on the emissions equipment of the vehicle. Their results show substantial joint benefits for reducing both pollutants for some policies, such as the straight fuel tax, and emission-based road pricing. Considering joint benefits and synergistic effects of different policies on the different external margins of vehicle use appear important for comparing among alternative policies.

Another European analysis, by Mayeres and Proost (2001), compares actual and optimal taxes on gasoline and diesel vehicles. They find that taxes on current diesel vehicles are too low and conclude that the tax mix is best altered through a revenue-neutral change in diesel and nondiesel vehicle ownership taxes.

Sterner and Hoglund (2000) also make the case for two-part pricing instruments. With a tax on the pollution-causing input—fuel, for example—polluters can reduce their tax liability if they can show that they are using a cleaner fuel than some average required level. Eskeland (1994) suggests such policies in developing countries to reduce vehicle emissions. For example, fleet operators would be required to pay a tax, but if they use compressed natural gas vehicles, their tax is reduced or eliminated. This type of two-part tax provides incentives for pollution reduction and may lower administrative costs by shifting the reporting responsibilities to the polluters. Harrington et al. (1998) find that a two-part tax on emissions, where the tax on some baseline allowable emissions is zero, results in welfare levels very close to those under a pure emissions tax. Two-part taxes and combined quantity- and price-based instruments offer promising opportunities and merit further exploration.

Trading Policies

It is obvious that it would be very difficult to devise a pure quantity instrument for individual vehicles, in which emissions could be monitored and traded across vehicles. Even the emissions standards that have done most of the work of emissions reduction in the past 30 years fall far short of being true quantity instruments for two reasons: they affect only new vehicles, and they affect only emissions rates, not total emissions. Fixing the quantity of total emissions would require restrictions on individual vehicle use, which would be difficult to implement and enforce. Instead, quantity instruments for vehicles take two forms.

What comes closest, perhaps, to a pure quantity instrument for individual vehicles is the complex system of vehicle-use permits suggested by Goddard (1997) for Mexico City. The three types of permits in Goddard's scheme are a base permit that allows the driver on the road on any given day of the week, an interruptible permit that can be revoked on high-pollution days, and a visitor permit for a vehicle that is temporarily being driven in the city. These permits

Even the emissions standards that have done most of the work of emissions reduction in the past 30 years fall far short in controlling emissions: they affect only new vehicles, and they affect only emissions rates.

would be distributed and could then be bought and sold in many locations in and around the city. The issuance of additional permits or the buy-back of permits would give regulators another tool to achieve pollution-reduction goals. The scheme also has the advantage of being able to address severe short-term pollution problems, which most other policies do not have.

Other quantity instruments have been directed at vehicle emissions rates or the number of vehicles. For example, Kling (1994) compared a system allowing emissions-certification trading with the current uniform certification requirements in the United States. Under the trading policy, vehicles can have different emissions certification rates as long as the sales-weighted average of each manufacturer's fleet meets the required emissions rate. Manufacturers can reach the target rate by averaging within their own fleet or by trading with other manufacturers. The model is applied using data from California on vehicle fleet and emission rates, and cost func-

tions for different levels of control. Allowing emissions trading in lieu of the existing uniform requirements results in cost savings of 1 to 18%, depending on the functional form of the cost function, the pollutants considered, and certain technology assumptions. These cost savings are relatively small because the existing vehicle mix under the current policy was already weighted in the direction of small vehicles, so allowing trading from this command-and-control baseline does not have much impact. The results might be different in today's market, which includes many sport utility vehicles and light-duty trucks.

Singapore uses a quota policy for the purchase of new cars. To limit the growth in vehicles, the government has set a quota on the number of new cars that may be brought into the country. The quota of new vehicles is auctioned off each year in a sealed-bid auction, allowing for a maximum fleet growth rate of 4% (Koh and Lee 1994).

It is also possible to devise quantity instruments "upstream" by establishing markets for commercial fleets or for fuel constituents. The leaded-gasoline phase-out discussed earlier in this chapter is an example. Another example is provided by the regional clean air incentives market (RECLAIM) in Los Angeles. This program was established in 1994 to reduce the high cost of compliance with clean air requirements in the Los Angeles region. It allowed sources to trade emissions of either NO_x or SO₂. Private companies in the Long Beach area were allowed to buy old vehicles around the Los Angeles basin and retire them in exchange for having the right to emit greater levels of NO_x at their plant sites. Such scrappage programs have been examined by Alberini et al. (1995, 1996), who found that the vehicles scrapped tend to be older and in worse condition than other vehicles in that same age class. Although emissions from these vehicles are high, their remaining useful life on the road is low. The cost-effectiveness of such policies ranges from \$3,500 to \$6,500 per ton of HC removed, which makes them attractive in some contexts and not others. Dill (2001) provides a summary of the studies on old-car scrap programs.

Accelerated vehicle scrappage is an example of trading mobile source for stationary source pollution, which until recently has not been allowed by EPA. It is not a cap-and-trade program as considered by Weitzman (1974), but an "offset" program, in this case allowing regulated sources to substitute emissions reductions from existing unregulated sources for emissions reductions at their own facilities.

Efficiency may be best served by policies that target upstream agents, such as manufacturers, fleet operators, and in some cases fuel producers, or by policies that have low transactions costs.

Another upstream policy was established in the United States under the Energy Policy Act of 1992. Fleet operators must purchase a proportion of their new vehicles as alternative-fuel vehicles to reduce dependence on foreign energy supplies. Under the rule, government or private-sector fleet owners can obtain credits for buying more alternative-fueled vehicles than required and sell those credits to other fleet owners. Winebrake and Farrell (1997) discuss the program and its potential to reduce emissions, but there has been no analysis of its potential or actual cost savings to date.

In summary, there is a large literature now on market-based policies for reducing conventional pollutants from vehicles. The unique characteristics of vehicles make designing policy instruments different from other sources of pollution. There are many, many sources; there is great heterogeneity across sources; and emissions cannot be easily or accurately observed. Thus, efficiency may be best served by policies that target upstream agents, such as manufacturers, fleet operators, and in some cases fuel producers, or by policies that have low transactions costs.

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CHAPTER SIX

Mobile Sources and Global Warming

Growing concern about global climate change has directed the attention of policymakers and analysts worldwide to all major sources of greenhouse gases (GHGs). The transportation sector is one such source, and within this sector motor vehicles are now coming under particularly careful scrutiny. In the United States, which leads the world in motor vehicle use both in total and per capita, motor vehicles account for about 20% of CO₂ emissions. In other countries motor vehicle use is growing rapidly, especially in the developing world. Accordingly, the search is on for efficient and equitable policies to reduce emissions of greenhouse gases from motor vehicles. Reducing CO₂ emissions essentially means reducing fossil fuel use in vehicles, and there are only three ways to do that:

1. Reduce the amount of vehicular travel.
2. Improve fuel economy in vehicles.
3. Switch to alternative fuels with lower greenhouse gas potential.

Fuel Taxes versus Fuel Economy Standards

As in the case of conventional pollutants, the question of price versus quantity instruments is relevant for global climate change. Newell and Pizer (forthcoming) examine price and quantity regulation in this context and find that an optimal tax policy generates welfare gains that are several times higher than a permit policy. Because carbon is a stock pollutant, they argue that the marginal benefits of abatement are relatively elastic. However, their analysis does not focus on the transportation sector specifically.

The relevant price instrument is a fuel tax. Nearly all the carbon in gasoline is emitted as CO₂, and most of the rest is emitted as CO, an even more potent GHG. This, together with the fact that the location and timing of GHG emissions do not matter, means that a tax on the carbon content of fuel would be an almost ideal instrument against global warming—“almost” because other GHGs are present in vehicle emissions, including methane, and because a fuel tax would provide no incentives for abatement technology. It would, however, provide incentives to reduce emissions in the three ways mentioned above.

The familiar gasoline tax, which is in use in nearly every country, could be easily converted to a carbon tax. Since motor fuel is already taxed at varying rates, achieving a reduction in fuel

use would require even higher fuel taxes. In Europe, there appears to be an acceptance of high tax rates, although the civil disturbances in the summer and fall of 2000 may have shown there are limits to high fuel prices even in Europe. At any rate, the European approach to reducing greenhouse emissions in the long run is to rely on alternative propulsion systems. In the short run, it is apparently to encourage the use of diesel-powered engines, which are considerably more fuel-efficient than spark-ignition engines of comparable power.

In the United States, however, gasoline taxation to mitigate global warming has very little purchase with politicians, and little wonder, considering how unpopular gas taxes are with the general public. These taxes are widely perceived as unfair to the poor and to those whose circumstances and life choices have locked them into a high-mileage lifestyle. And their effectiveness is challenged, not only by the public but also by some economists, who argue that the low price elasticity of motor fuel will require very large tax increases to have the desired effect (e.g., Greene 1991). As noted in Chapter Three, recent studies find the elasticity of motor fuel to be low, especially since 1980.

Resistance to high fuel prices would make it just as difficult to implement a quantity instrument as it would a price. A pure quantity instrument for GHG emissions from vehicles would very likely be an upstream instrument, whereby refineries would need carbon permits to sell fuel. At the retail level, prices of fuels would rise and fall depending on the availability of permits.²⁷

Instead, the favored approach in the United States has been mandated fuel economy standards for new vehicles powered by fossil fuels. Since 1979 motor vehicles in the United States have been subject to sales-weighted CAFE standards. At the time of enactment, the principal justification was concern about scarcity of motor fuel and fear of reliance on imported oil. Today these concerns have abated somewhat, but the policy is still strongly favored by environmentalists as a way of curbing emissions of greenhouse gases. Since 1991 the CAFE standards have required fuel economy in new cars and trucks to be 27.5 and 20.7 mpg, respectively. Pressure is growing to raise these standards substantially. In 2001 the National Research Council issued a report examining the cost and technical feasibility of raising the CAFE standards, and at this writing there are bills before Congress to raise the standard for cars and trucks to 36 MPG by 2013. These deliberations will be guided in part by the past performance of the CAFE policy.

Because CAFE stands out as the principal alternative policy to higher fuel prices for controlling greenhouse gas emissions in the transport sector, and because it offers so many examples of the unintended consequences of policies, it will be the focus of the discussion below. In addition, other countries are considering policies that resemble CAFE. For example, the United Kingdom has recently imposed on new vehicles a variable excise duty based on CO₂ emission rates.

CAFE Effectiveness

Between 1978 and 1991, the CAFE standards increased from 18 to 27.5 MPG for cars and from 17.2 (in 1979) to 20.7 mpg for trucks. Over that same period, the fuel economy of new vehicles sold in the United States increased from 19.9 to 25.1 MPG (USDOE 2001). Most observers

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agree that this increase was caused by CAFE (NRC 2002), but for dissenting views see Nivola and Crandall (1995) and Sykuta (1996). One of the points of contention is the “rebound effect,” which prevents an increase in fuel economy from causing a proportional decrease in fuel use. The idea is that as fuel economy improves, the cost per mile of driving declines, causing an increase in the demand for travel. The size of the rebound effect thus equals the elasticity of VMT: $E_{G.mpg} = -1 - E_{VMT.cpm}$. As noted earlier, the elasticity of travel is well studied, and the consensus estimate of the size of the rebound effect is between -0.1 and -0.2 (NRC 2002a). The rebound effect is real but fairly small.

There is less consensus concerning other effects of CAFE, including its effect on highway safety and its role in several profound changes in the U.S. motor vehicle market since 1980. These controversies are due partly to problems inherent in fuel economy standards in general, and partly to the details of the particular CAFE standards adopted.

Details of CAFE Policy

The most important of these details was that separate standards were specified for cars and light trucks. And the timetable of gradually increasing car standards was specified in the legislation itself. For trucks, standards were established later by regulation. At the time, most trucks were

commercial and farm vehicles, and business and agricultural groups argued successfully that severe restrictions would adversely affect profits and productivity. Federal policy also favored light trucks by exempting vehicles exceeding 8,500 pounds from any CAFE standards and by exempting trucks from the “gas-guzzler” tax imposed on cars. The upshot was that the CAFE standards for trucks were much more lenient and remain so today.

The difference between car and truck standards was rendered especially important by another aspect of the CAFE policy, little noticed at the time: the definition of “car” and “light truck.” Manufacturers managed to get trucks defined in a very liberal way, such that a vehicle was considered a truck if it had no hump behind the front seat and if its rear seats could be removed without the use of tools.

The fuel economy standards were also “fleet-weighted” by manufacturer. This approach allowed manufacturers much more flexibility than a set of model-specific standards (and hence lowered the costs) for meeting a particular fuel economy standard. Thus a manufacturer could still sell gas-guzzlers as long as their sales were offset by sales of enough subcompacts that the average fuel economy met the standards. To prevent manufacturers from shifting production (and employment) abroad, where small-vehicle capacity and expertise were high, a manufacturer’s imports (from outside North America) were considered separately from domestic production.

*Effects on Market Structure*²⁸

Several writers have pointed out that the CAFE policy, in principle at least, creates some unusual and even perverse incentives. Vehicle manufacturers have three general strategies for meeting

The difference between car and truck standards was rendered especially important by another aspect of the CAFE policy, little noticed at the time: the definition of “car” and “light truck.”

the CAFE standards: (1) adopt fuel-saving innovations in new vehicles, (2) modify vehicle characteristics to reduce fuel use (mainly, reduce vehicle weight), and (3) use pricing to affect the mix of vehicles sold. That is, a manufacturer can improve its CAFE rating by raising the price of big cars and lowering the price of small cars. This third alternative was not much discussed during the legislative deliberations over CAFE, but it is the only alternative available in the short run. Kwoka (1983) observes that such a fleet-mix strategy will affect not only the mix of vehicles but also the number of vehicles sold, and if sales increase then aggregate fuel use will rise even as average fuel economy improves.²⁹

In Kwoka's model the firm is a monopolist. Kleit (1990) develops a similar model for a competitive firm, giving conditions under which output and energy use will rise when a CAFE standard is imposed. Then, using plausible assumptions together with empirically determined initial conditions and parameter values, he solves a market simulation model with five types of vehicles. He finds that a CAFE standard below 28.1 mpg causes fuel consumption to increase. At higher rates, CAFE does reduce fuel use, but very inefficiently. He concludes that even modest changes in the fuel economy standards could be costly.

Greene (1991) calibrates a multinomial logit model of consumer vehicle choice with 1986 sales data and uses it to estimate the efficient set of price changes required to achieve a given improvement in fuel economy. He asserts that the consumer surplus change approximates the maximum cost to firms, assuming that firms were made to bear the entire cost of the policy. This gives him an upper bound on the cost to manufacturers, which turns out to be small for small changes in the CAFE standard, but which increases rapidly as the standards become successively tighter. Compared with the estimated cost of new technology and design changes, Greene (1991) concludes that a pricing strategy is not economically attractive except for short-run changes.

A more serious market-structure question involves cars and light trucks. Between 1980 and 1998, the sales of new light-duty vehicles that were classified as trucks increased from 21.4 to 47.3%. Part of the growth could be attributed to the growing popularity of pickup trucks in both commercial and household applications. But far more important was the introduction of minivans and sport utility vehicles (SUVs)—new families of vehicles that were classified as trucks for regulatory purposes but had many of the characteristics and appeal of passenger cars. By 1990 what had been only a farm or commercial vehicle had become a household vehicle as well.

The growth of the light-truck market is a fact; the role of CAFE in that growth is less certain. The disparity between car and light-truck CAFE standards is certainly a strong incentive for manufacturers to look for ways to sell trucks to car buyers, and the loose definition of a truck certainly created opportunities to do that. However, other events were occurring simultaneously. As the recent NRC report points out, during the 1980s the full-size light-duty truck category was dominated by U.S. manufacturers, and they naturally sought to expand sales in that category.

Thorpe (1997) constructed a CGE model consisting of a light-duty vehicles sector, with the rest of the economy treated as a single remaining sector. Thorpe's motor vehicle model has 17 vehicle classes, differing by vehicle type, vintage, and country of origin. His conclusion was that CAFE reduces fuel economy by encouraging motorists to shift to less fuel-efficient vehicles (e.g., from Japanese small cars to American cars and from big cars to light trucks).

Godek (1997) examined the trend in the light-truck share of the passenger-vehicle market and found that the abrupt rise that began in 1981 could be explained by a CAFE dummy variable and a CAFE variable interacted with a time trend, results consistent with an upward rota-

tion of the trend line. During this same period, the small-car market declined slightly while the large-car market declined substantially—it was a mirror image of the truck trend. On the other hand, the truck share of the market was growing in the decade before 1978, though not quite as fast as the two decades after 1981, and it declined drastically, from 29 to 26%, between 1978 and 1981. Because of these puzzles in the trend line, the National Research Council committee report refrained from assigning all the blame for the shift to trucks to the CAFE policy.

Effects of CAFE on Conventional Pollutants

The conventional wisdom is that CAFE policy has no effect on conventional pollutants. This is by design, for the U.S. emissions standards for light-duty vehicles are written in terms of grams of pollutant per mile, regardless of fuel economy. However, various observers have noted several disparate effects of CAFE on emissions. First, the CAFE standards and the vehicle emissions standards interact at the design stage, but not in a simple way. Certain technologies, such as electronic fuel injection, both improved fuel economy and reduced vehicle emissions. But emissions controls add weight to vehicles, and stringent emissions standards have also precluded the use of lean-burn engines, which are desirable from a fuel-economy standpoint, but which cannot meet emissions standards for NO_x (NRC 2001).

Second, to the extent that CAFE reduces fuel use, it also reduces all the VOC emissions from upstream petroleum industry operations. Delucchi et al. (1994) have estimated these emissions and found them to be large. Perhaps not coincidentally, the two American cities with the worst air quality—Los Angeles and Houston—both have substantial refinery operations.

Third, although gas-guzzlers may *discharge* pollutants at the same rate as more efficient cars, they *generate* pollutants at a much higher rate. To avoid higher discharges they must have more efficient emissions control systems. If, over time, the systems' performance deteriorates, then the vehicle's emissions profile will grow to resemble its generation rate. Using 1990 remote-sensing data, Harrington (1997) found that CO and VOC emissions did depend on fuel economy. The effect was small for the first five years or so but increased with vehicle age, so that at age 12, a 20-mpg car had 60% more emissions of VOC and HC than did a 40-mpg car. With more recent vintages, however, that effect is likely to be attenuated because of the improvements in the durability of control equipment.

Finally, the rebound effect implies that higher CAFE standards will induce greater vehicle use, which in turn will raise emissions of conventional pollutants (Khazzoom et al. 1990). As noted above, this effect will be small—a 10% increase in CAFE will generate at most a 2% increase in travel in new vehicles subject to it—and will in any case be countered by the improved emissions rates over time as vehicles with improved technology make up a larger share of the fleet.

To our knowledge no one has examined the net impact of these effects.

CAFE and Highway Safety

Probably the most important and controversial issue involving CAFE is its putative effect on highway safety, an issue discussed at length by two National Research Council reports (2002, 1992). The mechanism is weight. Numerous studies, reviewed in both reports, have found a

significant negative correlation between vehicle weight and the probability that an accident will result in serious injury or death. Crandall and Graham (1989) connected these results to CAFE in a quantitative way. They developed a model explaining vehicle weight as a function of the expected prices of gasoline and steel four years hence (the estimated lead time for design and production) and estimate it on vehicle weights for new vehicles in 1970–85. Over this period the elasticity of weight to fuel price is only -0.14 . When restricted to 1970–77, however, they found an elasticity of -0.54 . They interpreted this result as evidence that CAFE was binding after the fall in oil prices in 1982, and use their results to construct counterfactual vehicle weights in the 1978–85 period. They estimate that CAFE reduced vehicle weights by an average of 18%, or about 500 pounds. Using Evans's (1985) study of the role of vehicle size in traffic fatalities, Crandall and Graham estimate that the 500-pound reduction in average vehicle weight caused a 14 to 27% increase in occupants' fatality risk. As Godek (1997) points out, however, the increased accident incidence may have been counteracted by the shift from cars to trucks.

Crandall and Graham's work was criticized by Khazzoom (1994) and Ledbetter (1989), primarily for equating vehicle size with vehicle weight. These authors argued that vehicle length or volume might be the crucial variable, but because weight and volume are highly correlated in vehicles, volume is usually discarded in studies of vehicle characteristics and accidents. This issue is still largely unresolved. Two physical principles are at work. When two vehicles of different weights collide head-on, the deceleration is proportionately lower in the heavier vehicle.³⁰ Deceleration is also lower in more spacious vehicles, because the greater volume provides greater "crush space"—the ability of vehicle components to absorb the energy of impact and not transmit it to the driver. The simple physics implies that in one-vehicle collisions mass doesn't matter but crush space does and, in multivehicle collisions, it is not mass *per se* but the disparity in the masses of the vehicles that kills.

The realization that weight disparity was important gave new significance to the observed shift in fleet composition toward trucks. Whereas vehicle safety studies had hitherto concentrated on the safety of the occupants of the vehicle, concern was growing over the fate of occupants of the other vehicle in a crash. The recognition of this externality, together with the controversial article by Crandall and Graham, motivated new work by the National Highway Traffic and Safety Administration on the question (NHTSA 1997). In this study, the effects of weight on accident severity were categorized by vehicle type. A 100-pound weight reduction increased fatalities by about the same amount for cars and trucks (actually, slightly more for trucks) in accidents involving stationary objects, a confirmation of intuition. A 100-pound reduction in car weight increased the fatality risk by 2.63% in a collision with a light truck. However, a similar reduction in trucks *reduces* fatality risk by 1.39% in a collision with a car. Taking all types of accidents and their incidence into account, the study found that reducing car weight increases fatality risk by 1.13% per 100 pounds, while reducing truck weight reduces fatality risk by 0.26% per 100 pounds. These results remain controversial, and NRC was not able to achieve unanimity on this point.

In its discussions of safety, the NRC committee considered only the effects of differences in weight. But the rebound effect also has obvious safety implications. Indeed, the rebound effect

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may look small when only fuel consumption is considered, but once its effects on conventional pollutants, accidents, and traffic congestion are brought into the discussion, its effects might not look so small any more.

Looking Ahead

As far as we know, the CAFE policy is in use in only one country, the United States. Nonetheless, the importance of the United States as a fuel consumer — 27% of the world's motor vehicles and 22% of the world's petroleum consumption in 1996 — means that the CAFE policy is of interest to policymakers throughout the world. Not only does CAFE influence vehicle design throughout the world, but it also offers a policy alternative to countries that wish to reduce transportation fuel use without raising transportation fuel prices. Also, as mentioned above, countries that do not adopt CAFE may nonetheless levy taxes on the potential CO₂ emissions of vehicles.

To that end it is worth noting that in addition to its apparent effectiveness in improving fuel economy and reducing the rate of growth in fuel use, CAFE may have had several unanticipated consequences. (We say “may” because confounding variables make it difficult to draw conclusions.) Early on, its effectiveness may have been compromised by manufacturers' use of pricing policies that changed fleet mix to improve fuel economy, yet led to greater vehicle sales and possibly higher fuel use. CAFE may also have caused the down-weighting of vehicles, which in turn may have increased the severity of accidents and fatalities. Finally, CAFE may have encouraged the shift in vehicle markets from cars to light trucks and SUVs.

It is important to keep in mind, however, that except for the use of pricing strategies, the unanticipated consequences of CAFE could be traced to the details of policy design; they were not inevitable consequences of the policy. Chief among these details was the disparity between truck and car standards and the loose definition of trucks. In the next few years, the U.S. Congress is likely to revise and render more stringent the CAFE policy, which has remained unchanged for a decade. Much of the discussion will center on these unanticipated consequences and what can be done about them.

■ ■ ■

Vehicles and the Urban Environment

For some time it has been clear to urban planners and policy analysts that (1) patterns of land use exhibit very large and measurable differences from one urban area to another, and (2) some urban forms are associated with high levels of vehicle ownership and use, others with low levels, even controlling for household income. Low-VMT areas tend to be densely settled, have good access to transit, and offer a mix of residential and commercial uses, so residents can walk to work and to shop. In high-VMT areas, density is lower, transit access is poor and inconvenient, the “jobs-housing balance” (ratio of jobs to workers in a neighborhood) is far from unity, and land uses are segregated, so nearly every errand requires a car trip. Many if not most U.S. households choose to live in suburban areas when they have a chance, and sprawl is just a reflection of these preferences and, in fact, a solution to congestion (Gordon and Richardson 1999).

Theory shows that the failure of motor vehicles to pay either the full or the marginal social costs of travel is a potential cause of sprawl, which is defined for these purposes as urban development that is less dense than the optimum. Using the traditional model of urban form, Brueckner (2001) has argued that one of the major market failures that lead to sprawl is unpriced congestion externalities from vehicle use. Because the congestion externality is not accounted for in road pricing, the cost of driving is too low, resulting in urban areas that are too dispersed. Using a simulation, Wheaton (1998) finds that developed areas could be roughly 10% too large. In an earlier paper, McConnell and Straszheim (1982) examine the case with both unpriced pollution and congestion externalities from vehicle use in a city. They simulate an urban area using parameter values from the literature and find that the joint externalities cause the city to be too dispersed, and that the congestion externality dominates the pollution externality.

The evidence is strong that lower-density development is associated with higher VMTs. Newman and Kenworthy (1989) have made this case with data assembled from a number of cities around the world. Figure 4 shows the association with data from 1960 and 1990, using average household VMT and average density for 19 cities. Over time density has fallen and VMT has increased. This evidence of the inverse relationship between aggregate VMT and average urban density has led researchers to search for, and in some cases claim, a causal relationship.

These results have found their way into the policy realm. Under the banner of “smart growth” or “transit-oriented development,” policies to promote transit-friendly, compact development are making their way onto the agenda of numerous planning boards and city councils in

developed countries, especially in the United States. In part, such developments are advocated for their own sake. Environmentalists and “new urbanists,” believe they are more pleasant and livable for residents, and the fact that some such developments have succeeded in the marketplace provides a bit of supporting evidence. But they are also advocated because they are thought to further other environmental goals. Such as, if more compact development reduces VMT, then it will *pari passu* reduce emissions of conventional pollutants and greenhouse gases.

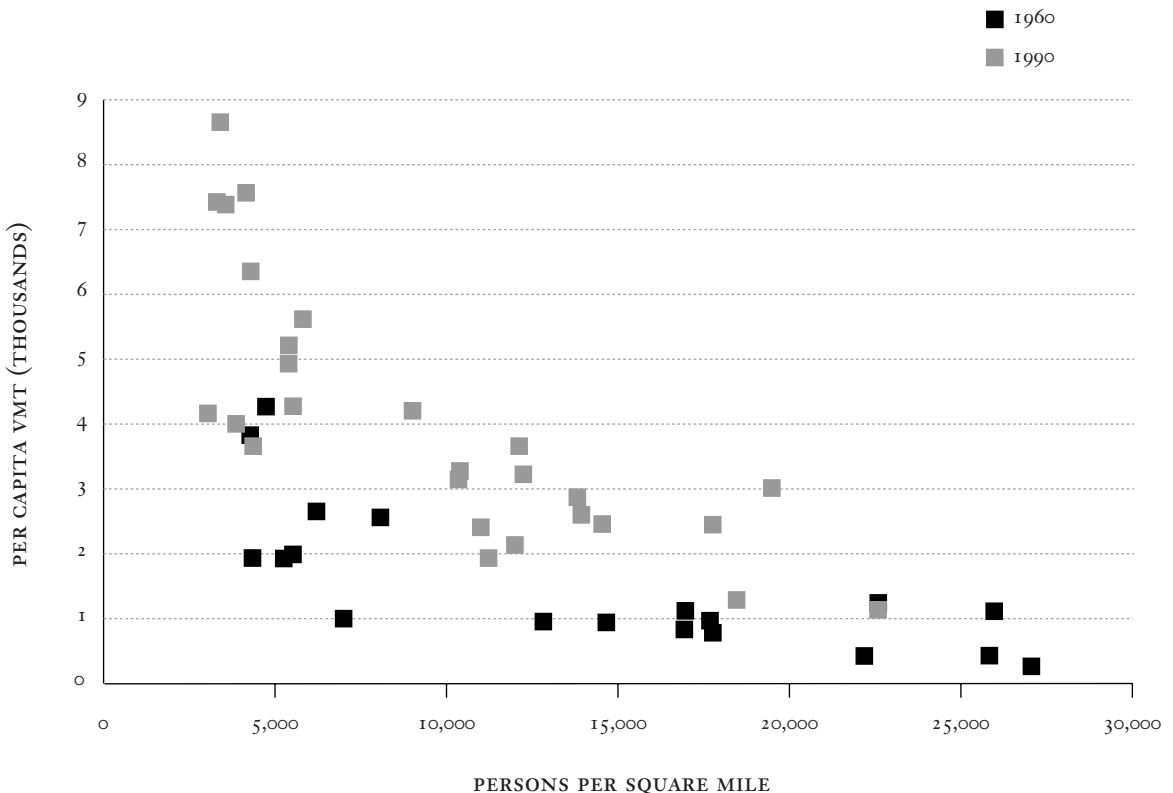
Do Land-Use Patterns Affect the Demand for Travel?

Much of the evidence invoked to support the proposition that land-use patterns reduce travel demand is descriptive. We won’t review these studies here (see Crane 1999 for an excellent review), but many are based on hypothetical maps of land-use developments, showing that some designs are much better than others at reducing the distance residents must travel to get to employment or shopping opportunities. In addition, besides Newman and Kenworthy’s (1989) aggregate study of density, transit availability, and auto use, other researchers have observed the same tendencies in microdata.

Holtzclaw (1994) added detail to Newman and Kenworthy’s work by using data from specific neighborhoods in four California cities. His estimation results show vehicle ownership and VMT

FIGURE 4

Per Capita VMT (Cars) versus Urban Population Density for 19 Cities



Source: Newman and Kenworthy (1989).

to be a function of density, public transit availability, neighborhood shopping availability, and a constructed measure of “pedestrian friendliness.” For example, his results suggest that a doubling of density should lead to a 25 to 30% drop in average VMT per household. Holtzclaw did not have individual household-level data, however, and he omitted prices, income, and other factors besides the land-use measures that might explain vehicle use.

Using the U.S. Nationwide Personal Transportation Survey (NPTS),³¹ Dunphy and Fischer (1996) found evidence of greater transit use and lower VMTs in autos in higher-density communities. However, they also found that underlying demographic characteristics influenced both. Residents of higher-density communities tended to be those with less need for autos and with greater dependence on transit systems. This also raised the issue that location itself was endogenous and that both travel demand and location were affected by other underlying variables. For example, those with an aversion to auto travel choose residential locations that cater to their travel tastes and capabilities. This endogeneity is a serious obstacle to sorting out the relationship between land use and travel demand.

Several other studies using microdata have failed to find clear empirical evidence for a large land-use effect once other factors are accounted for. These studies are based on discrete-continuous models of travel choice, as described in Chapter Four. Individuals or households are assumed to make decisions based on an underlying utility function, in which the demand for travel is derived from the demand for activities and goods that require travel.

Cervero and Kockelman (1997) used travel diary data for individuals in San Francisco neighborhoods to examine the link between travel decisions, land-use variables, and other socioeconomic variables. In separate equations, VMT and mode choice (car versus transit) were regressed on such land-use variables as population and employment densities, the extent of mixed-use development, and street-design measures, and such socio-demographic variables as education, gender, and age. Some of the land-use variables were significant but tended to have smaller effects on the travel measures than the socio-demographic variables. In a similar analysis, Kitamura et al. (1997) found that the personal attitudes of the diary respondents about driving, the environment, and other factors were more important in explaining travel behavior than the land-use variables.

Bento et al. (2002) estimated both a commute-mode choice model in which commuters chose to drive, walk, or take some form of transit, and a model of vehicle ownership and use. This study is distinguished in part by its careful attention to the definition of exogenous land-use variables—measures of land use that are clearly exogenous to individuals making commuting, vehicle ownership, and driving decisions. The measure of density was citywide (more than 7,000 households drawn from 119 Metropolitan Statistical Areas from the 1990 NPTS), and the transit supply measure is of transit miles supplied normalized by city area. The land-use variables are found to be significant, but to have relatively small effects on mode choice and on vehicle ownership. There is little effect on miles traveled per vehicle. While sensible, the search for exogenous land-use variables comes at a cost, for if the important land-use variables are local, then analysis will overlook them.

Walls et al. (2002) also estimated a model of vehicle choice and use with data from the 1990 NPTS, but they took the opposite strategy on the land-use variables, accepting the possibility of endogeneity but offering more nuanced land-use variables. To the extent that the data permitted, their land-use variables were local: population density in the respondent’s zip code and

self-reported transit availability. The paper specified a sequential choice model, in which respondents first chose how many vehicles to own (one, two, or three) and the type of vehicle, and then how many miles to drive each vehicle. The paper found that the major effect of the land-use variables was on vehicle ownership, not on use per vehicle, and that the effect on vehicle ownership was very small except at high densities, above roughly 5,000 people per square mile (they estimated that 72% of urban households lived at lower densities). Finally, they found that proximity to a transit stop reduced the likelihood of vehicle ownership only if transit stops were less than a quarter mile away from the respondent's residence.

In an earlier study using multivariate techniques, Schimek (1996) found similar results: that residential density had a significant effect on household vehicle use and vehicle trips, but only when density levels reached 4,000 people per square mile. At 7,000 per square mile, the effect was more pronounced (the mean density of the urban areas in his study was 2,500 per square mile).

Other researchers have examined directly whether the jobs-housing balance affects commute times or the journey to work. This question has in fact been studied in urban economics for 20 years, ever since Hamilton's (1982) "wasteful commuting" article, a comparison of actual com-

mutte times in Baltimore and the efficient level of commuting based on a monocentric model. Hamilton found that actual commuting exceeded "efficient commuting" by a factor of 5 to 8. Giuliano and Small (1993) also examined commuting patterns in travel analysis zones in Los Angeles, focusing on the jobs-housing balance throughout the city. They found that jobs and housing were reasonably well balanced, but as in Hamilton's results, hypothetical minimum commutes are much lower than actual commutes. These results lend support to the notion that in U.S. cities, long commutes are attributable

to other factors, such as two-worker households, frequency of job changes, and the demand and supply of particular worker skills at different sites.

Crane and Crepeau (1998) estimated several equations using data from travel diaries in San Diego. The dependent variables for the first equation were the number of nonwork trips, and for the second, choice of mode (walk, vehicle, or other). These travel measures were assumed to depend on economic variables such as price and income, on tastes and other household variables, and on various land-use measures. The trip costs were found to be highly significant in the number of nonwork trips: if trips were longer or slower, there were fewer of them. Also, the greater the share of commercial land use, the greater the number of trips; however, VMT could rise or fall depending on the length of the trips.

Boarnet and Crane (2001) extended that analysis and found conflicting evidence in the results. Some of the land-use measures had an impact on the travel demand variables only at the larger zip-code levels (comparable to census tracts) and only when residential location was treated as endogenous. They also examined whether trip costs had a separate impact on travel decisions. The results show that when land-use variables have an impact on the number of trips, it is through the effect on trip price (speed and distance in their model). They found that for areas with a higher proportion of commercial land use, people have both shorter nonwork trip distances and slower trip speeds. The net effect on trip costs was ambiguous. Overall, these authors

Lower transportation costs tend to disperse employment locations, particularly if jobs follow the outward movement of workers.

found very little conclusive evidence about the effect of land uses on travel behavior because of the complexity of the interrelationships.

In summary, what do we know about the effect of land uses on travel demand from this research? Decisions about travel are complex and are made jointly with other decisions about residential and employment location. It is not clear that the existing studies have fully captured these decisions and identified the role of economic variables relative to others. Several conclusions do emerge, however:

- Most studies find either none or only small effects of land-use variables compared with other variables on travel measures, such as VMT or vehicle ownership. As with any negative finding, it is impossible to say whether the effect really does not exist or whether the land-use variable has not been modeled correctly.
- There is fairly consistent evidence over a number of studies that measures of urban form tend to have more impact on vehicle ownership than on VMT, and the effect on vehicle holdings tends to be at high density (Walls et al. 2002; Bento et al. 2002).
- Even though the effects of certain land uses on travel measures are small and significant in these studies, it is not clear that we can take the next step and assume that policies to change existing land uses over time would have the predicted effect.
- Finally, some evidence suggests that differences in land uses have an impact through the effect on the price (time and distance) of trips, at least for nonwork trips. This suggests that it is prices and costs that drive travel decisions, and changing costs directly would have more impact on urban travel than more indirect methods of changing urban design and land use.

Transportation Cost and Urban Travel Demand

Accessibility and the cost of transportation have always played a central role in theoretical models of urban form. Households are assumed to trade off higher commuting costs for housing cost savings at locations distant from employment centers (see classic articles by Muth 1969 and Mills 1972). As the cost of transportation falls, whether because of improvements in vehicles themselves or improved transport infrastructure, households are able to purchase more housing farther away, causing the density of development to fall. In general, lower transportation costs tend to disperse employment locations as well, particularly if jobs follow the outward movement of workers. Firms' locations also depend on the costs of transporting raw materials and finished goods, and thus transport technology, which varies by industry and type of product, has affected industry dispersion. For a good review of this literature, see Pickrell (1999).

For the United States, Pickrell (1999) argues that travel time per mile fell by 50% in the early part of this century and has fallen tenfold in the past 200 years, since the time many eastern cities were first developed. Little wonder urban densities have also fallen with each subsequent decade (McDonald 1989). A decline in costs of urban mass transit did not occur, so these results also provide some explanation for replacement of transit by motor vehicles as the principal transportation mode in most urban areas of the United States. McDonald (1989) and Jordan

et al (1998), in empirical studies of density patterns and the determinants of density across cities, conclude that transportation cost differences have contributed to density differences.

In a study of motorization and road provision in countries with widely varying income levels, Ingram and Liu (1999) argue that another explanation for urban decentralization is the combination of congestion and the high cost of road building in existing cities. They find that at the national level, vehicle ownership and roadway length both tend to increase in proportion to income. In contrast, in most urban areas, the amount of roadways tends to increase much more slowly than vehicle ownership or income. The resulting higher congestion levels tend to increase pressure to build more roads, but new roads in already urbanized areas are expensive to build. It is less expensive to build new roads at the outer edge of cities, leading to more decentralization.

Noland and Lem (2002) have examined whether increasing urban road infrastructure increases VMT and therefore congestion, and possibly urban decentralization. They conclude that the induced travel effects of road building are real, but some empirical questions remain. It is not clear, for example, that the endogeneity of road capacity has been fully accounted for in these models on induced travel demand. In addition, the implications for urban form and infrastructure investment need to be drawn out.

One final point we wish to make is that transportation costs have not fallen everywhere. In downtown Manhattan, for example, vehicular transportation costs are very high, even though the costs of vehicles and fuel are not much different than in other places. But other transportation costs in New York City are very high: principally traffic congestion and the cost of vehicle storage (which can run hundreds of dollars per month and helps explain why the observed effects of density on VMT operate through the number of vehicles and not the level of use of each vehicle). In addition, there is a cheap, reliable, and fast transit system. Note that the traffic congestion and high vehicle storage costs in New York City are inevitable, given its density. But that makes vehicular transportation costs endogenous also, if density levels are high enough.

Vehicle Policies and the Urban Environment

The competing hypotheses presented above generate competing policy strategies for reducing private-vehicle use. One strategy aims to reduce the need for car travel by more compact development that mixes housing, shopping, and employment so that the same errands can be accomplished with less travel, or by improving alternatives to private vehicles, such as pedestrian or bicycle-friendly development or cheaper and improved mass transit. Policies that implement this strategy can be carrot-based, such as subsidized transit fares, investment subsidies for desired commercial and office development near transit stops, and provision of bicycle or pedestrian facilities; or they can be stick-based, such as zoning, use of “growth boundaries,” or limits on the number of building permits issued in a given period. The goal of these policies is not to reduce vehicle use directly but to encourage development that makes it easier to get by without a vehicle. They cannot be evaluated simply by their effects on vehicle use.

Casual evidence suggests that these land-use strategies may work better if transportation prices are already reasonably high. At least in Europe, where fuel prices are high, cities appear to have made more effective use of transit and have been better able to control land use than U.S. urban areas. Also, car ownership and use are much lower in Europe. Multicollinearity makes it difficult to sort out the relative strength of fuel prices and land-use measures in achieving these

objectives, though. In the United States, there have been several periods since the 1920s when a consensus emerged favoring the reining-in of suburban growth, and we are apparently in one now. Thus we find smart-growth initiatives, first in Maryland but now spread to other states, that use a mix of regulation, taxes, and government investment to channel growth into favored areas. And then there is Portland, Oregon, the cynosure for those attempting to apply land-use tools to control development and reduce dependence on automobiles. Portland has implemented a growth boundary that severely limits all development outside until development within the line reaches a certain level.

It is too early to evaluate these programs, but the results so far of Portland's growth boundary have in some ways fulfilled the hopes of its supporters and the fears of its opponents. Despite rapid economic and population growth in the region, the rate at which rural land is converted to urban uses has slowed, a new transit system is thriving, and the city has enhanced its reputation for livability. On the other hand, traffic congestion and housing prices have increased impressively.

The other strategy is to discourage auto travel directly by making it more costly to users. The policies that implement this strategy include higher fuel taxes, mileage-based registration fees, high-occupancy vehicle lanes, congestion or time-of-day pricing of roads, and parking fees. Fuel taxes are used almost universally but only as a revenue-raising instrument. Although congestion pricing is highly regarded by economists for its efficiency properties, it faces formidable political barriers (Giuliano 1992; Goodwin 1994) and is not often used in practice. Interest in high-occupancy toll lanes is high, and some experiments are under way. Only in Singapore and a few Norwegian cities (e.g., Trondheim) is it used extensively to ration access to scarce roadway capacity.

Far more common are regulatory methods to restrict vehicle use. The auto-free zone, which prohibits motor vehicles in the city core (except perhaps for residents), is an example of a regulatory policy that directly discourages vehicle use. Another is "traffic calming," the use of barriers and speed bumps to reduce the impact of traffic in residential neighborhoods. These measures may have beneficial local effects, but whether traffic is shunted to other areas is not well understood.



CHAPTER EIGHT

Toward More Sustainable Transport

Trends

Predicting the future is always difficult, but there are some predictions that inspire more confidence than others. One thing that appears so likely that it is almost considered a fact is that world population will be much greater in the future. The United Nations' medium projection of world population in 2030 is 8.1 billion, nearly a 35% increase over the present. Substantial per capita income growth is also expected, and combined population and income growth will very likely produce unprecedented increases in vehicle ownership and use. Dargay and Gately (1999) applied their model of income elasticities (see Chapter Four) to World Bank GDP growth estimates, extrapolated to 2015, and found that the world vehicle fleet would nearly double between 1992 and 2015. Similarly, a doubling of worldwide vehicle travel was forecast by Schafer (1998), using travel budgets. Even then there will be plenty of room for further growth. China, for example, is both the world's most populous country as well as an economic dynamo, with current and projected growth rates exceeding those of almost every other country. By 2015, vehicle ownership in China, for example, is projected to increase from 2 to 60 vehicles per 1,000 population, for a total of 78 million vehicles (Dargay and Gately 1999).

At the other end of the certainty spectrum, there will almost certainly be surprises: new and unanticipated developments in markets, technologies, and environmental impacts. Motor vehicles have encountered their share of such surprises. When tetraethyl lead was introduced into fuel for octane enhancement in the 1920s, no one dreamed that the health of three generations of children would be put at risk. Just recently something similar has happened for methyl tertiary-butyl ether (MTBE), which serves as both an octane enhancer and an oxygenate in gasoline. More soluble than gasoline, MTBE has begun to show up in community drinking water, presumably from groundwater supplies contaminated by leaking underground storage tanks. California has recently announced plans to outlaw MTBE in fuel; if the oxygenate is to be replaced, ethanol will have to be used. That MTBE was put into fuel initially for environmental reasons adds irony to environmental distress.

The possibilities for problems of this sort are always present, given that fuels and additives are complex mixtures of organic compounds, few of which have been completely tested for health or ecological effects. But even the common air pollutants we think we are bringing under control—CO, NO_x, and SO₂—have the potential for causing hitherto unsuspected problems. Re-

cently, scientists have discovered a significant decline in the concentration of hydroxyl radical in the atmosphere. This ion is one of the principal agents for cleansing the atmosphere (Madronich 1993). If hydroxyl levels continue to drop, the lifetime of many atmospheric constituents will increase and could turn smog from a moderately serious local or regional issue into a life-threatening global problem.³²

Putting aside the catastrophic implications of that scenario, the resource and environmental implications of the trends in vehicle ownership and use are sobering enough. For conventional pollutants, the problems are probably manageable. The effectiveness and durability of vehicle emissions-control systems are driving emissions rates to very low levels. Certainly, many dirty cars will remain in world fleets, but over time they will be replaced by much cleaner, if not always state-of-the-art, vehicles.

Even for diesel-powered vehicles, which continue to lag behind the best achievements of gasoline-powered engines, substantial emissions reductions are expected in the new generation of diesel engines to be produced in the United States and Europe by 2006. These engines will require ultra-low sulfur fuel, and policies designed to assure adequate supplies have already been promulgated. Of course, emissions from existing vehicles will remain an issue in the intermediate term, especially since diesel vehicles in commercial applications tend to have long lifetimes.

It is more difficult to be sanguine about emissions of greenhouse gases. Because there is no known abatement technology for CO₂, emissions are determined by the amount of carbon in the fuel. Reductions in GHG emissions from motor vehicles, therefore, would seem to require a large reduction in use of fossil fuels, which in turn requires some combination of reduced vehicle travel, better fuel economy, and alternative fuels that cause no net increase in carbon use.

The trends of urbanization and suburbanization will very likely continue. That is, larger and larger shares of the population will live in urbanized areas, and within those areas, at lower and lower population densities. The resulting spatial arrangement of activities will be more and more difficult to serve by transit, and mode share will continue to move toward the automobile. At the same time, there will be difficulties everywhere, but especially in developing countries, in coping with the increased traffic. If the recent past is any guide, lack of funds, environmental concerns, and difficulty of securing rights-of-way in already built-up areas will make it extremely difficult to build enough highway infrastructure to keep up. This appears to be a worldwide phenomenon. Even in the United States, which has a dedicated source of transportation funds, rates of VMT growth have exceeded rates of highway expansion by a factor of two in recent years (Shrank and Lomax 2002).

Technologies

We briefly consider three technological effects: technology that produces substitutes for travel, technology that improves management and use of transportation infrastructure, and new transportation technology that addresses the environmental implications of increased demand for transportation.

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New Technology and Transportation Substitution

For some time, policymakers and the public have been beguiled by the idea that advances in communications technology will enable people to conduct more of their work and personal business without leaving their homes. “Telecommuting” and more recently “e-shopping” have been put forward as potential replacements for urban travel. Although the latter is too new for researchers to gauge its long-term effect on travel, telecommuting has been a noticeable phenomenon since the 1980s.³³ Most telecommuting is home-based, but some companies and local governments in the United States have set up telecommuting centers offering office space to workers of many companies. For both home-based and center-based telecommuters, it is difficult to identify the determinants of telecommuting and their effect on travel demand.

It is even difficult to estimate how many home-based telecommuters there are. Surveys of home-based work in the United States conducted between 1980 and 1997 (Mokhtarian and Henderson 1998) variously estimated between 2 million and 55 million home workers, with most of the variation due to differences in definitions and the categories included. Only a small fraction of these workers are telecommuters; the rest work out of the home or engage in uncompensated after-hours work. A congressionally mandated study of the effect of telecommuting on transport (US DOT 1992) asserted that 2% of the workforce was telecommuting 1 to 2 days per week in 1992 and predicted that within a decade, 7 to 15% would be telecommuting 3 to 4 days per week. Drawing on evidence from several sources, Mokhtarian (1998) estimated that in California in 1997, 6% of the population telecommuted 1.2 days per week.

Although telecommuting offers flexibility and convenience to those able to take advantage of it, its effects on transportation are likely to remain minor. To be sure, telecommuting does appear to reduce daily VMT. In a survey of California workers, Mokhtarian and Henderson

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(1998) found that among telecommuters, trips and time spent traveling were 18% and 16% less, respectively, than among ordinary commuters. But when we multiply this by the fraction of telecommuters and their frequency of telecommuting (about 6% and 20%, respectively), the effects on travel are small. Similarly, Eash (2001) analyzed the 1995 Nationwide Personal Transportation Survey and found that VMT by home-based workers did not differ from that of on-site workers, but the distribution by time and trip purpose did differ. One optimistic study (US DOT 1992) found that even if 15% of the workforce were telecommuting 3 to 4 days per week, total VMT would be reduced by only 1.4% (4.5% during rush hour).

Finally, Mokhtarian and Salomon (1997) point out that telecommuting cannot be taken out of context because other aspects of the information revolution may encourage more travel, not less. At the very least, cell phones, car faxes, and wireless Internet may make it more bearable to be stuck in traffic, and many workers can now begin their workday while still commuting. These opportunities are not limited to personal travel. Just-in-time inventory systems, made possible in part by computerized management of inventories and electronic links between factories and suppliers, have allowed manufacturers to substitute transportation for inventory. More generally, better communications technology has always meant the expansion of markets and trade over longer distances. No doubt, the rapid growth in world trade in the past few decades is primarily the result of falling

trade barriers, but certainly better information flows about products and developments in electronic banking have played important roles. When all aspects are considered, say these authors, the weight of evidence suggests that the relationship between transport and telecommunications is one of complementarity, not substitution.

Technology and Management of Infrastructure

Telecommunications and information technologies also provide many opportunities for improving travel itself, through what are called intelligent transportation systems (ITS).³⁴ These technologies are being applied in myriad ways, such as better emergency response, incident information to allow motorists to alter routes, information about wait duration at transit stops, and dynamically optimized signal control. Their main purpose, however, is to improve productivity of roads and hence increase the capacity of a given amount of concrete. The environmental effects of these innovations will therefore resemble those of capacity additions. In the short run, we can expect smoother traffic flow, less stop-and-go traffic, and lower emissions. Long-run effects are ambiguous, since capacity expansion will reduce the costs of travel and attract latent demand to the roadways.

From an environmental perspective, probably the most significant application of ITS is to electronic metering and electronic toll collection (ETC). With ETC, vehicles are equipped with a transponder that communicates with a roadside device and allows metering of road use. The first application of this technology has been to freeways, and, worldwide, an estimated 8,800 freeway lanes have been equipped with ETC (Fourchet 2001). ETC systems can reduce congestion by replacing tollbooths on major highways, but their real environmental significance is that they remove most of the technical barriers to something approaching true social-cost road pricing, as discussed in Chapter Three. That is, with electronic road pricing, the toll rates can vary by time of day, type of road, location, and eventually vehicle emissions rate and driver characteristics. So far, these capabilities have only begun to be exploited; with rare exceptions (e.g., congestion pricing in Singapore, Norway, and a few other places), ETC is being used to collect revenue tolls. The Netherlands may be about to take a further step (Pieper 2001). Road pricing is a central element of the Dutch National Transport Plan, and late in 2002 the government will decide whether to implement an ambitious program of social-cost pricing—congestion and air quality—on all roadways, not simply expressways. Rollout of this system would begin in 2004, with mandatory use of transponders by 2006 for the 8 million vehicles of the Dutch fleet.³⁵

Also using transponders, London is implementing a congestion tax. The 250,000 motorists who drive into the eight square miles of the City between 7:00 A.M. and 6:30 P.M. will have to pay £5 (\$7) a day. Those who fail to do so will face an automatic £80 penalty unless they fall into one of several exempt categories, such as taxi drivers or nurses on duty.

Elsewhere, there is more resistance, not only to road pricing but also to electronic toll payments. In the United States, which currently has about half of worldwide ETC lane installations, use of transponders in vehicles is voluntary almost everywhere. Despite the large time savings that the use of electronic tollgates provides, market penetration of transponders is less than 50% for most U.S. systems (ETTM 2002). According to one observer, the ETC market in the United States has “peaked” and is approaching a replacement market (Fourchet 2001).

It is not clear why participation in electronic tolling is so low. In part, it could be the lack of coordination among different tolling agencies, requiring multiple transponders (which can in-

terfere with each other's operation). Significantly, in the Northeast, where the EZ-Pass system is used in the New York City region south to Delaware, transponder penetration is much higher, at 50 to 70%. Concerns about privacy and fraud may also be holding back transponder use. On expressways with ETC in Japan, only about 2% of commuters use ETC transponders.³⁶ One reason is their high cost—about \$300, 10 times or more what is charged in the United States and Europe (where transponder fees are often credited against future tolls).

Despite the slow acceptance of electronic tolling, pressures will be strong for expanding its use in the long run. Given the political, financial, and environmental opposition to new highways, authorities everywhere are on the lookout for methods to raise revenues and ration access to existing roads. These pressures will intensify in the future if gasoline tax revenues, everywhere one of the major sources of revenue for new transportation infrastructure, fall as alternative vehicles and more fuel-efficient conventional-fuel vehicles achieve greater fleet penetration.

Environmentally Benign Vehicles

If alternative vehicles do not achieve more prominence, it will not be for lack of effort, for the environmental problems associated with the current technology of motor vehicles have inspired an intensive search for alternatives. This search has been two pronged, directed at both new fuels and new propulsion technologies. Within limits, it is possible to mix and match these technologies. For example, either compressed natural gas or methanol can be used to power either a spark-ignition engine or a fuel cell. The coupling of an engine with different fuels can have very different environmental characteristics—and vice versa.

As mentioned in Chapter Three, when comparing technologies we must compare life-cycle emissions—emissions during vehicle and fuel production, vehicle use, and final disposal. For motor vehicles, by far the most important stages are fuel production and vehicle use. Numerous researchers have now constructed life-cycle estimates comparing fuel propulsion configurations. These studies cover a wide range of alternative assumptions about future technology, and they consider a wide range of vehicle characteristics and performance. This breadth is useful for some purposes, but it complicates the making of comparisons across studies. Costs are rarely considered, largely because of the uncertainties in costing out products and technologies not yet available on a commercial scale. Energy efficiency and GHG emissions rates are the metrics used for comparison, and some studies also pay attention to performance characteristics, including acceleration, range, and refueling time. For an excellent review of these studies, see MacLean and Lave (forthcoming).

For our brief treatment of the issue, we focus on one such study, conducted by General Motors, Argonne National Laboratories, British Petroleum, Exxon/Mobil, and Shell (GMC/Argonne 2001).³⁷ This study held vehicle characteristics constant—the specimen vehicle was a Silverado full-size truck—and considered 75 fuel pathways applied to spark-ignition (SI) engines (common gasoline engines), compression-ignition (CI) engines (diesel), fuel cell vehicles, battery-powered vehicles, hybrid electric vehicles (HEV), and fuel cell HEV. The full report is accessible on the Internet.

Table 7 summarizes the main results of this study for a representative set of fuel-propulsion combinations. The table displays three performance characteristics: acceleration, fuel economy, and GHG emissions. The fuel economy is a measure of the energy efficiency of the vehicle itself:

TABLE 7

Fuel Economy and GHG Emissions of Vehicle Propulsion Technologies

	Feedstock	Fuel economy		Zero to 60 mph (seconds)	GHG emissions	
		mpg	index		g/mi	index
Conventional gasoline SI	Fossil	20.2	100	7.9	544	100
Conventional diesel CI	Fossil	23.8	85	9.2	472	87
Ethanol (85%) SI	Renewable	20.2	100	7.9	172	32
Gasoline SI HEV	Fossil	24.4	83	6.3	454	83
Diesel HEV	Fossil	29.4	69	7.2	384	71
Gasoline fuel cell	Fossil	27.2	74	10.0	408	75
Methanol fuel cell	Fossil	30.3	67	9.4	371	68
Ethanol (100%) fuel cell	Renewable	28.6	71	10.0	35	6
Hydrogen fuel cell	Fossil	43.2	47	8.4	330	61
Hydrogen fuel cell HEV	Fossil	48.1	42	10.0	296	54

Source: GMC/Argonne (2001) Tables 2.1 and Appendix 3C.

SI: spark ignition

CI: compression ignition

HEV: hybrid electric vehicle

the so-called tank-to-wheel (TTW) efficiency. GHG emissions are more or less an indicator of the efficiency of the entire fuel-vehicle system, the well-to-wheel (WTW) efficiency.

For fuel economy we show both mpg and an index of performance that indicates the amount of fuel in percentage terms required to travel a fixed distance, relative to the baseline, conventional gasoline spark-ignition vehicle. Likewise, we show two measures of GHG emissions: grams per mile and an index of emissions emitted while traveling a fixed distance. As indicated by the fuel economy index, a diesel vehicle provides about a 15% reduction in fuel requirements, diesel plus HEV a 31% reduction, gasoline and alcohol fuel cells about 30%, and hydrogen fuel cells about 53 to 58%.

Now compare the GHG index. For fossil hydrocarbon fuels, the GHG index tracks the fuel economy index quite closely, reflecting the high and relatively uniform conversion efficiency of hydrocarbon fuels at the fuel production stage. For hydrogen-fueled vehicles the two indices diverge, mainly because of the low efficiency of converting fossil fuels to hydrogen. They diverge even more strikingly for ethanol, the only renewable fuel process considered here. As shown, a conventional vehicle with 85% ethanol fuel has the same fuel economy as a gasoline-powered conventional vehicle, yet its GHG emissions are only one-third as high.³⁸ Comparison of the ethanol SI with the various fuel cell vehicles shows that fuel is a more important determinant of GHG emissions than engine type. When renewable ethanol is combined with a fuel cell propulsion system, GHG emissions approach zero.

Considering that these new vehicle technologies have not yet been produced for commerce, and in fact are barely available at a demonstration scale, it should not be surprising that their costs are very uncertain. Toyota's Prius is the first HEV available in the consumer market, and

its purchase price is \$3,000 greater than a comparable vehicle, the conventionally powered Toyota Corolla.³⁹ According to Lave and MacLean (2002), the Prius becomes economically preferable to the Corolla only when fuel prices exceed \$3.55 per gallon. That makes the Prius attractive to buyers in Europe but not in the United States. As for fuel cell technologies, the consensus seems to be that the price premium over conventional vehicles will be even greater for the foreseeable future.

What, then, would constitute a cost-effective GHG reduction strategy for the United States? Taking the Kyoto Protocol as representative of climate policies in the short term, meeting the proposed targets would cost \$30 per ton of carbon: for vehicles, this means about 0.3¢ per mile, or about 7.5¢ per gallon (see Chapter Three). But a 7.5¢-per-gallon gasoline tax would have only minimal effects on vehicle use, purchase patterns, or technology. In other words, motor vehicles have only a small role to play in any cost-effective near-term strategy adopted by the United States, and no doubt other countries as well.

When we turn to long-term strategies, however, motor vehicles can no longer be ignored, especially in light of the prospects for a vast expansion in worldwide motor vehicle ownership and use. Table 7 suggests that in the long run, use of renewable fuels will be essential. Cellulosic ethanol was the most attractive alternative for Lave et al. (2001), for several reasons. First, it provides a very large reduction in net GHG emissions, considerably larger even than a fuel cell vehicle with a natural gas feedstock. Second, the existence of large numbers of flexible-fuel vehicles means that the United States already has an installed user base for ethanol fuel. Third, although ethanol production would require considerable land resources, marginally productive lands that would not impinge on current food production could be used. It would not be cheap, however. Current estimates suggest that ethanol could be produced for the gasoline equivalent of \$2.70 per gallon, equivalent to a carbon price of \$300 per ton.

Meeting the current gasoline demand with ethanol would require 300 million to 600 million acres of land—up to a quarter of the total U.S. land area. Demand, of course, will continue to grow, at least in the short run, but increases in productivity both on the land and in the factory will increase the fuel output per acre. According to Lave et al. (2001), growing energy crops will have much smaller impacts on the environment than other agricultural or urban uses of land. But even if this is true, the ability of the land to service increasing demands for both fuel and food is not guaranteed.

■ ■ ■

Conclusion

Collectively, motor vehicles contribute significantly to a number of important environmental problems. Given that vehicles are here to stay, the question before us is how can we best manage vehicle ownership and use, and the associated environmental effects. The answers will differ with the mix and severity of environmental problems in countries around the world. However, in all cases, economic issues are at the heart of debates over how best to design policies to mitigate the negative effects from vehicles. Such economic issues include ways to measure and add up environmental effects, methods for predicting behavior about vehicle use and response to policies, methods for evaluating policies, and ways to consider the costs and incentives of different policies.

In assessing the evidence about the magnitude of the social costs of vehicle use, we find that at least in the United States, social costs of vehicle use are large and comparable to the private cost of driving, at least in some situations. In addition, vehicle taxes are probably too low and often of the wrong form to reflect the full economic costs to society of vehicle use. However, there are a number of methodological issues about evaluating social costs that have yet to be resolved. Not all external costs are included, and those that are do not usually reflect the additional cost of driving another mile, but rather reflect current or average costs of driving. To design environmental policy to reduce the costs of driving, we would want to use marginal costs as the signal to drivers about the full cost of their actions. In fact, there has been a strong reluctance, not only in the United States but in other countries as well, to use prices to reflect external costs.

Vehicles and their environmental effects are so pervasive in the world economy that there are a range of different types and places for policy intervention. We have focused our review on three important environmental concerns: air pollution, global warming, and urban sprawl. In the context of these problems, some policy designs and targets are likely to be more effective than others. We paid the most attention to the opportunities for market-based policies, because they offer the promise of better incentives to households and producers to take actions that will improve environmental goals. In addition, we identified two other ways policies might be made more effective. One is the importance of the stage of production or consumption where the policy is directed. If ideal policies that directly target pollutants at the point of damage to the environment are difficult or costly, upstream policies may be more effective. And finally, we observe one of the greatest dilemmas for policymakers—contending with the unintended consequences

that often result if the policy is targeted too narrowly on only one issue. We summarize our findings for each of these factors below.

The Possibilities for Market-Based Policies

While collectively significant, individual motor vehicles don't make much of an impact, and yet there are very great differences in those impacts from one vehicle to another. Just as with larger sources that are heterogeneous, efficiency would be served by policies that target those vehicles for which the payoff is high relative to the cost. This sort of targeting is just what well-conceived economic instruments do, but targeting small sources can have very high transaction costs. The role for incentive policies in regulating motor vehicles, we believe, lies in devising instruments and finding applications where transaction costs are likely to be low. For example, gasoline taxes (or more generally, carbon-content taxes) are an ideal instrument for mitigating global warming. They also have very low transaction costs, since they can be (and are) easily collected at the point of sale.

Pollution fees for motor vehicles are more difficult, since they require knowledge of vehicle emissions, which cannot be easily observed. This is, after all, why authorities opted for a policy of increasingly stringent new-vehicle standards: they are much more easily enforced. The technology for identifying vehicle types and therefore pollution-control equipment as part of electronic pricing systems now exists. It is being used in Hong Kong and will be tried in London and in the Netherlands.

Clearly, motor vehicle emissions policies will continue to affect vehicle and fuel technologies. But now, causality may begin to go the other way as well. Rapidly improving remote-sensing and telecommunications technology promises to enable non-intrusive determination of vehicle travel and emissions in ways that protect the privacy of individual motorists. This will finally make it feasible to charge motorists the true marginal social costs of travel—that is, fees based on fuel use, distance traveled, and possibly emissions. However, just having this technology is not sufficient. The technologies and the policy ideas have to be sold to a skeptical citizenry. So far, motorists have demonstrated considerable resistance not only to economic-incentive policies but also to the electronic-metering technologies themselves.

Political opposition to such economic instruments as congestion fees, pollution fees, and gasoline taxes has been noted throughout the report, and is an issue that needs to be revisited and considered in greater detail, we believe. It is possible that much of the opposition arises from the context in which these instruments ordinarily are discussed, where it is understood that there will be a tax increase. If the purpose of the fee is to change behavior rather than to raise revenue, these instruments should be revenue-neutral. That means that the discussion is incomplete without consideration of which existing taxes or fees should be cut. Then, in addition to asking which tax is the most efficient, perhaps it would be equally worthy to ask, "What package of tax reductions would most reduce political opposition?" Tax policies toward diesel fuel in Europe may represent a case where the policy can be both efficient and revenue neutral. High-sulfur fuel is taxed at a higher rate and lower-sulfur (cleaner) fuel is taxed at a lower rate than standard fuel.

Policy Design: Location of Policy Intervention

Even if economists and policy analysts could agree on the damages from pollutants and their direct source or sources, targeting those sources when there are millions of them may be very difficult. Again, transactions costs for finding all of the polluting vehicles and then of enforcing requirements on them may be prohibitively high. In such cases, there are possibilities for directing policies farther upstream in the pollution process. For example, in Europe, differential fuel taxes on high- and low-sulfur fuel have already been effective in bringing about the early introduction of low-sulfur diesel (which reduces particulates both directly and indirectly, in the latter case by enabling the pollution-control equipment that captures NO_x and particulates before they are released). The taxes are levied upstream on fuel refineries, instead of directly on diesel trucks or on fuel at the pump.

A similar case can be made for policies on gasoline to reduce emissions of greenhouse gases. Either taxes or regulations on fuel content can be levied at the pump, or upstream on distributors or refineries. The latter may be easier to enforce and have lower overall costs. In another example, vehicle I/M inspection and maintenance programs have not been as successful in many areas at reducing emissions as many had hoped. This is due, in part, to the high costs of enforcing compliance on so many drivers. But a policy that shifts responsibility upstream, to vehicle manufacturers to maintain the pollution equipment over the life of the vehicle, may be more effective and implemented at a lower cost per vehicle. Of course, such a policy would also reduce motorist incentives to maintain vehicle emission-systems, so a policy change of this sort would have to weigh the administrative and likely technical advantages of the upstream approach against its potential behavioral implications.

The Importance of a Comprehensive Approach

In addition to the use of fuel taxation to raise revenue, a variety of fee instruments can be used to address different policy goals, including pollution reduction, congestion mitigation, and reduction in greenhouse gas emissions. Rather than a piecemeal approach, it makes sense to consider these instruments in a comprehensive framework, such as the TRENEN framework of Proost and van Dender (1999) or that of Parry and Small (2001).

The importance of a comprehensive approach extends to the use of instruments that are not economic incentives. For example, we noted above that the CAFE policy was most likely responsible for the improvement in fuel economy in the United States in the early 1980s and may yet again be used for this purpose. Implementing that policy while fuel prices were low created a hardship for vehicle manufacturers, which were in the untenable position of making vehicles that consumers had little interest in. Manufacturers responded by taking advantage of the more lenient CAFE standard for trucks, for they built vehicles that counted as trucks for regulatory purposes but appealed to consumers as personal vehicles. Trucks—mainly SUVs and minivans—now account for over 50% of new vehicle sales in the United States.

In another example, the government recently provided subsidies to manufacturers if they produce “dual fuel” vehicles that can be operated on either ethanol or gasoline. However, there was no incentive provided to consumers at the same time to use ethanol. Ethanol prices remain higher than gasoline, and consequently, drivers use only gasoline. While the subsidy does result

in more dual-fuel vehicles, there is no benefit to the environment unless the vehicle and appropriate fuel policies are jointly implemented.

Continuing Importance of Advanced Technology

We have also tried to look to the future, to see what is on the horizon for vehicles and their effects on the environment. We will close with a few observations about what will be most important in the coming years as the world grapples with its need for mobility and the effect of vehicles on the environment.

Since 1970, virtually all emissions reductions in motor vehicles have come about because of technology-based emissions standards imposed first in the United States, and soon afterward in Europe and Japan. Regulation forced the development of new abatement technologies that by 2000 had reduced emissions rates of new vehicles by two orders of magnitude. These regulations may also have had an indirect role in introducing technology that improved overall vehicle performance. For example, fuel injectors were found on only a few high-performance vehicles in 1970. Emissions-control systems, which required much more precise distribution of fuel to the engine, helped hasten the diffusion of fuel injectors to all vehicles. A similar story could be told with respect to the spread of digital technology monitoring every aspect of engine performance.

The pace of technological improvements, which are influenced by regulatory pressure, continues to grow. Strenuous efforts are under way around the world to develop vehicle propulsion technologies that reduce the use of fossil fuels, either by improving fuel economy or by switching to renewable fuels. And some researchers are already thinking about the next great technological leap—replacing the internal combustion engine with fuel cells.

Technology to reduce the environmental footprint of motor vehicles will be even more vital in the future. Worldwide, the number of motor vehicles in use is expected to double in the next 25 years. In 2015, China is expected to have 60 vehicles per 1,000 population and an economy growing as fast as it is today. And in the United States, VMT continues to grow at a faster rate than both population and economic activity. Even after 100 years, the revolution in personal mobility has barely begun.



Notes

- 1 Until the advent of universal vehicle inspection and maintenance (I/M) programs, it had proved almost impossible to recruit a random sample of vehicles for emissions testing. Owners of dirty vehicles would have an incentive to avoid testing, for fear that test results might be used to require repair.
- 2 “Your Driving Costs 2001: Figuring It Out,” American Automobile Association.
- 3 These fees are set at the state level so they vary, but some states with high fees have begun to eliminate them. For example, in 1998 Virginia began to phase out an annual personal property tax for household motor vehicles that is allocated to local government revenues. Local governments set the rate as a percentage of Blue Book value. For a new vehicle this tax could easily reach \$1,000 per year.
- 4 An externality is an uncompensated effect of a production or consumption activity. For example, vehicle use causes air pollution, inflicting damages on people and property that vehicle users do not have to pay for. Externalities can be either positive or negative, but with vehicles the focus is on negative externalities.
- 5 There is very little information on the costs of roads and who pays those costs for developing countries. It is even difficult to find data on the number of lane miles and road capacity for most countries.
- 6 For a discussion of the promise and potential pitfalls of life-cycle analysis (LCA), see Portney (1993–94). Among the difficulties that have been encountered in applying LCA is the boundary problem. That is, we may examine the impacts of inputs to vehicle production, but what of the impacts of the processes producing those inputs. Where is the line to be drawn? One approach, developed by researchers at Carnegie Mellon University, is to link the analysis to an input-output model of the national or world economy (Lave et al. 1995). The researchers make available on the Web an input/output model of the U.S. economy, linked to a matrix of environmental effect coefficients (EIO/LCA 2001). The Carnegie Mellon University model is an application of an approach initially suggested by Ayres and Kneese (1969).
- 7 Parry and Small’s (2001) concise survey of damage studies finds a range of marginal damage estimates from a low of \$0.70/ton carbon to \$560/ton.
- 8 For a good summary of benefit-cost analysis principles, see Greenberg et al. (2001).
- 9 In “Environmental Investments: The Cost of a Clean Environment,” EPA estimates the cost of all its programs.
- 10 In the United States, the unit of vehicle use is vehicle miles traveled (VMT) and the unit of fuel economy is miles per gallon (mpg). Note that this measure of fuel economy is the inverse of that used

- in Europe, where fuel economy is typically expressed in units of liters of fuel per 100 km of travel.
- 11 This relationship is obviously true for individual vehicles. It is also used in aggregate studies, where *G* and *VMT* are understood to be total fuel and vehicle use, respectively, and *MPG* is taken to be the average fuel efficiency (AFE) of the fleet. Average fuel economy of two vehicles is calculated assuming each travels the same distance, which requires the harmonic mean of the MPG of individual vehicles, not the arithmetic mean.
 - 12 Dahl and Sterner (1991a) tabulate results of individual studies, categorized by type of study. It also contains an extensive bibliography of studies. Dahl and Sterner (1991b) presents a summary of the review and an interesting and useful discussion of the implications of various studies.
 - 13 Dargay and Gately (1999) used a Gompertz function in their estimation procedure.
 - 14 The focus of travel demand research now appears to be the mixed multinomial logit model (MMNL), which is still more flexible than the nested logit model, and which in fact can approximate any random utility model. MMNL is only now beginning to be used in applications. McFadden (2001) surveys the historical development of qualitative choice models up to the present day. Also, Maddala (1983, chap. 3) provides an extensive and accessible treatment of the logit and nested logit models.
 - 15 Two exceptions can be found in articles by Brownstone et al. (1996) and Goldberg (1998). Using random utility models they find low elasticities, but this is likely because they use transactions on new vehicles rather than vehicle holdings as the dependent variable. Goldberg's data set was the U.S. Consumer Expenditure Survey for the years 1984–90.
 - 16 Reductions in the use of lead were achieved by designing engines to be tolerant of lower-octane fuels and by substituting other additives to raise octane.
 - 17 Although NO_x reductions were required through the 1970s and 1980s, the reductions were not as strict for HC and CO, in part because the technology that reduced HC and CO tended to increase NO_x. With the advent of the three-way catalysts in the early 1990s, reducing all three pollutants became more feasible. Also, the chemical reactions leading to ozone formation were complex and not well understood. VOC had been considered the limiting pollutant for ozone formation through the 1980s, but new evidence—that VOC emissions were higher than originally believed—made it clear that NO_x was the key in many jurisdictions (NRC 1991).
 - 18 Only later did evidence emerge that lead in gasoline was a major contributor to elevated blood concentrations of lead. See USEPA (1990).
 - 19 In the past, policies toward the vehicle and the fuel have been made independently, although there was likely some interdependence, as evidenced by the fact that the fuel companies have always lobbied hard for strict controls on engines, and auto makers were strong advocates of the cleanest, highest-quality fuel.
 - 20 Trading of sulfur credits is to be allowed within each of five major petroleum regions to achieve the 80% reduction.
 - 21 See USDOT (1996).
 - 22 SO₂ and lead pollution were problems in the past, but levels of these two pollutants have improved in the past decade.
 - 23 The program operates by issuing color-coded decals that contain vehicle license numbers. Five colors are issued, and cars must be parked on the day the color is prohibited. Cars from outside the control area do not have to follow these rules.
 - 24 It is telling that economic incentive policies comprise only a small fraction of all regulatory programs in the United States and Europe but nonetheless account for a large share of the *ex post* economic studies of regulatory policies. Perhaps this is due to the fact that economists, who conduct

these *ex post* studies, have an avuncular interest in economic instruments, but surely the relative ease of determining costs also is important. See Harrington et al. 2000.

- 25 Individual vehicle emissions can be monitored, using remote sensors on the roadside, but these readings provide only a snapshot of emissions, not a continuous monitoring. It is important to get a number of readings for each vehicle that are taken while it is in a mode of travel that best reflects its true emissions (mild acceleration). It is also difficult to get readings on an entire fleet of vehicles because the sensors work only for one lane of traffic and in good weather. Finally, remote sensors tend to work better for CO₂ and VOC emissions, and less well for NO_x. Research is under way to develop ways to read vehicle emissions continuously using on-board devices.
- 26 On-board devices that measure emissions may be feasible in the future, but capability does not yet exist. Currently, remote sensors can measure pollutant levels for CO and HC at a specific time and under certain driving conditions.
- 27 Kopp et al. (1997) have resurrected an idea first proposed by Roberts and Spence (1976): a hybrid instrument in the form of a carbon-emission permit market in which sources could purchase additional permits at some ceiling price.
- 28 There were also some major disparities in firms' impacts that are beyond our scope. See Kwoka (1983), Kleit (1990), and Yun (1997) for a discussion and some results. Kleit's simulation, for example, breaks out results for Ford, GM, Chrysler, "Asian," and "other."
- 29 Kwoka (1983) also noted that domestic manufacturers in 1978, CAFE's first year, sharply raised the ratio of big-car to small-car prices, evidence that in that year at least, manufacturers did adopt a fleet-mix strategy.
- 30 Conservation of momentum requires that, for example, if two vehicles, one twice the mass of the other, collide head-on while traveling 45 mph, the velocity immediately after the crash will be 15 mph in the direction traveled by the heavier vehicle. Thus the change in velocity in the heavier vehicle is 30 mph; in the lighter, 60 mph.
- 31 The NPTS is a national survey of daily household travel patterns. It is conducted every five years by the Federal Highway Administration, Department of Transportation.
- 32 A special issue in the popular British science magazine *New Scientist* (22 April 2001) used Madronich's results to develop a scenario of global catastrophe from a runaway reaction involving conventional pollutants.
- 33 Most observers see telecommuting as a desirable trend. However, Safirova (2002) argues that it can also reduce welfare by interfering with agglomeration economies in cities.
- 34 Massive Web resources are devoted to intelligent transportation systems. A good introduction to the possibilities can be found on the U.S. DOT site, www.its.dot.gov.
- 35 From a presentation by M.M.D. van Eeghen, Director-General of Passenger Transport of the Ministry of Traffic, Public Works and Water Management, "Road Pricing in the Netherlands," at the Kickoff Road Pricing Event, March 20, 2002.
- 36 "Prices Keep Electronic Toll Collection System from Catching On." *The Japan Times*, May 9, 2002.
- 37 Examples of other such comparisons are those by Delucchi (1998) and Wang (1999).
- 38 Conventional gasoline engines can use mixed ethanol fuel (15% gasoline) with some minor modifications. In the United States there are now substantial numbers of flexible-fuel vehicles capable of using "E85" fuel. The Alternative Motor Fuels Act of 1988 (P.L. 100.494) provides a CAFE credit for such vehicles, even though gasoline is now used with these vehicles almost exclusively.
- 39 Even at that price, Toyota is said to be providing substantial subsidies for each Prius sold (Lester B. Lave, personal communication June 5, 2002).

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