In this chapter, we shall review the literature on the demand for transport and transport externalities, focusing on the issues relating to road and rail transport. We first deal with transport demand, especially in the context of inter-city traffic movement.

The Demand for Transport

Three broad classes of models have characterised development and progression in the field of travel demand (Fischer, 2000). The first class is the traditional model associated with large urban transport studies, involving aggregate and descriptive use of data. This type of model conventionally includes four types of sub-models: trip generation, trip distribution, modal split and trip assignment. The second class of demand model is the microeconomic approach of travel choice behaviour emphasising individual-level behaviour and in some cases making use of random utility theory. The third class is the activity-orientated approach viewing travel behaviour as daily or multi-day pattern of behaviour, related to and derived from differences in life-styles and activity participation among the population.

We shall concentrate on the second class of model, viz. the model of travel choice behaviour, since the current study is concerned with the choice between the rail and road modes in inter-city travel. It is at the level of the individual that the need to travel arises and at this level choices about travel are important (Fischer, 2000). Individual choice refers to the selection decision by an individual among commodities which are discrete in nature, such as the mode of travel to be chosen. Among other things, travel behaviour is reflected by pre-trip decisions consisting of destination, mode, route and departure time choices, and en route decisions such as relating to diversion of alternative routes or rescheduling of intended trips. In this framework, travel choice is principally concerned with two factors: (i) the individual in question with his needs, travel experience, preferences, perceptions and attitudes, influenced by both the socio-demographic environment (e.g. age, car ownership) and the environment of social customs and norms; (ii) the physical environment (e.g. the
transport network infrastructure) determining objective travel opportunities and their characteristics.

Lave (1969) develops and tests a modal split model for urban journey to work. The consumer's probability of choice is expressed as a function of his own socio-economic characteristics and the system characteristics of the alternative transportation modes available. The model concentrates on the choice decision of individuals rather than searching for regularities in the aggregated statistics associated with zones or neighbourhoods.

The author first discusses some previous models of modal split. The complex of factors influencing an individual's choice of mode is divided into two main categories: (i) personal characteristics of the individual (income, tastes, auto ownership, competing family needs for the car) and (ii) characteristics of transportation alternatives available (relative time, cost and comfort). Most of the modal split models developed so far had tended to rely on only either the above personal characteristics of the individual or characteristics of transport alternatives available. On account of the highly aggregated nature of the data used and the high degree of collinearity among explanatory variables, the models yield little insight into the possible results to be expected from changes in the alternatives available to commuters. However, a few attempts had also been made to disaggregate the data and approach the problem from the standpoint of the individual decision-maker.

Three such studies are discussed in this context by Lave. In the first study [Warner, S.L. (1962), ‘Stochastic Choice of Mode in Urban Travel: A Study in Binary Choice’, Northwestern University Press, Evanston], use is made of a survey containing specific information on commuters' current travel modes and the characteristics of the alternative modes which they perceived to be available to them. A model is formulated to explain observed choice of travel mode as a function of the commuter's characteristics and the relative characteristics of alternative travel modes. The final model involves the ratio of auto cost to transit (i.e. public transport) cost, the ratio of auto time to transit time, family income, sex and age of the commuter, car availability and distance to work. Warner estimates his model by use of logit and discriminant analysis. His estimates indicate that the price and time
elasticities of demand for transit (public transport), for these commuters who have a choice between transit and auto, is about one. The second study [Moses, L.N. and H.F. Williamson (1965), 'Choice of Mode in Urban Transportation', Unpublished Report, Northwestern University] develops a rigorous indifference curve model which incorporates the effects of relative time, relative cost and the commuter's income. The model is calibrated to obtain estimates of the amount by which the cost of a transport mode should be changed to divert a specified percentage of commuters from one mode to another. The authors find that it would require a reduction of bus fare by as much as 30 to 40 cents to divert half of the current auto users to transit in the city of Chicago.

The same problem is taken up in another study in the context of measuring the value of travel time to commuters [Lisco, T.E. (1967), 'The Value of Commuters' Travel Time: A Study in Urban Transportation', Unpublished PhD dissertation University of Chicago]. Data is used from a 1964 Chicago area survey to estimate a model utilizing auto cost minus transit cost, auto time minus transit time, family income, commuters' age and sex, and a variable involving alternative uses for the family auto. Probit analysis is used to fit a relationship between travel choice and the above variables. The author's primary concern was to estimate the value of travel time, which he finds to be $2.50-2.70 per hour.

Having discussed these studies, Lave proceeds to develop a behavioural model of commuter's choice from the standpoint of what it motivates an individual to choose one mode or another. A number of possible variables might be included in a study of the factors that motivate a commuter's choice of transport mode: relative cost, relative time, relative comfort, auto ownership, purpose of trip, family size and composition, income, sex and age of commuter, and length of trip. Although each of these has a part to play in determining modal split, only the first three may be considered as instrumental variables useful for implementing some normative goal on the part of the city planner or traffic analyst. The author discusses each of these variables and then proceeds to estimate the modal choice model. The estimation is complicated by the binary nature of the dependent variable. The author uses probit analysis in which the dependent variable is simply an index number produced by the linear combination of various variables. The index number determines the choice

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1 Here the time elasticity of demand for a particular mode is the proportional change in demand for travel by that mode as a result of a proportional change in the ratio of auto time to transit time.
decision of an individual, i.e. when it is above a certain critical level, he decides one way and when it is below this critical level, he decides the other way. The assumption of a normal distribution of individual critical levels of the dependent variables provides the means of interpreting the estimated index numbers. The probit equation is estimated by maximum likelihood techniques. Results are obtained for the choice between bus and auto if relative time or cost characteristics are changed, given the modal split obtained in Chicago in 1956. Reduction of transit time by 5 minutes on bus on the average journey to work would have diverted an additional 3.1 percentage points of auto commuters onto the bus mode. A 10 per cent reduction in the bus fare would have produced approximately the same amount of diversion, i.e. 3.3 percentage points.

A recent study by Wardman, Toner and Whelan (1997) examines the degree of interaction between rail and car in the inter-urban leisure travel market in Great Britain. Disaggregate models are developed to explain individuals' actual choices between rail and car. The estimated models are applied to examine a range of policy measures. The research is done in the background of the very high expected rates of growth of car traffic in Great Britain as well as the potential for improvements in rail services to reduce the volume of car travel. Previous work had produced little evidence on cross-elasticities in the inter-urban travel market in Great Britain. Modal choice models had been developed to a much less extent for inter-urban travel than for urban travel, while aggregate models of inter-urban demand had typically made little allowance for the effect of other modes. However, some previous work had found that for journeys over 50 miles, the cross-price elasticity of demand for car travel with respect to the price of rail is 0.04 [Acutt, M.Z. and J.S. Dodgson (1996), 'Cross-Elasticities of Demand for Travel', Transport Policy, 2(4)].

Wardman, Toner and Whelan (1997) make use of a disaggregate choice model to examine the effect of policy measures such as reduction of fares on travel demand. Such a model is defined by specifying a probability distribution of the unobserved variables affecting utility, given the values of observed variables in a homogeneous market segment (McFadden, 1997). The probability distribution then determines the choice probabilities, i.e. the proportion of the population group choosing a particular mode in accordance with the rule of utility maximization. The assumption of a concrete probability distribution for the unobserved components of utility leads to a concrete formula for the choice probability. However, most distributions of unobserved components yield computationally forbidding choice
probability formulae. An exception, however, is the multinomial logit (ML) model. This model can be interpreted as a disaggregate behavioural model with special assumptions on the probability distribution of the unobserved variables. A multitude of possible disaggregate travel demand models can be formulated in this framework, with the form of the main utility function depending on the application. Besides, the ML model has the mathematical form of share models used in conventional travel demand forecasting system. For the choice between k alternatives, the ML model expresses the probability that an individual chooses alternative i as a function of the utilities (U) of the k alternatives in the choice set:

\[ P_i = \frac{e^{U_i}}{\sum_k e^{U_k}} \]

Utility is in turn related to relevant observable variables \((X_i)\) which include socio-economic and other variables:

\[ U_i = f(\beta, X_i) \]

The point elasticity of demand for mode i with respect to changes in the level of variable X on mode k is next specified. A logit model's elasticities depend not only on market share but also, in general, on the level of the variable for which the elasticity is being calculated. A general specification of the utility function for mode i and m explanatory variables is:

\[ U_i = \sum_{m} \beta_{im} X_{im}^{\lambda_{im}} \]

The function is estimated in an iterative manner across different pre-specified values of X in search of the best fit. In the model, relative valuations of different modes are expressed in monetary terms. For example, the value of travel time savings is expressed as a monetary equivalent of the time benefit. The variables used to explain choice are the cost (including any access and egress costs for rail) and total journey time (including any access and egress time) of each mode, train headway and number of train interchanges. The coefficients of the chosen functional form are allowed to vary by mode. The generalised mode choice model is used to examine the effectiveness of policy measures that could be used to reduce the amount of inter-urban car travel. It is seen that rail pre-reductions have a somewhat larger impact on car demand than the equivalent rail journey time reduction. This is the case for both group and individual travellers. Furthermore, a motorway toll would
have a lower impact on car demand than an equivalent charge applied in situations where car does not perform well. The cross-elasticities of car demand with respect to the characteristics of rail are low for inter-urban leisure travel. In the context of a forecast inter-urban traffic growth of around 4% per annum, even major improvement in the price and quality of rail services can delay the increase in road traffic by only a year or so. In most cases, punitive or restrictive measures to reduce car use are likely to have a substantially greater impact.

Recently, the multinomial logit model has been modified to form the random parameter logit model (Hensher, 2001). This is of the same form as the standard multimodal logit model, except that one or more taste weights for the individual decision-maker (including alternative-specific constraints) may be treated as random parameters with the standard deviation estimated together with the mean. The selected random parameters may be embedded in a number of predefined functional forms and are typically assumed to be normally or lognormally distributed. The random parameter model is sometimes called the "mixed logit" model because of the mixture of an extreme value type 1 distribution for the overall utility expression and embedded normality for the distribution of the taste weights across a sample. The random parameter/mixed logit model does not have a closed form expression (unlike the multinomial logit model) and so it is approximated numerically through simulation by the method of simulated maximum likelihood.

Having surveyed some of the literature in the analysis of travel demand, we shall now turn to studies of the relative environmental/social impact of rail and road.

Environmental Performances of Rail and Road

A study commissioned by the French Railways (SNCF) in France (Savelli and Domergue, undated) found that the contribution of rail to various pollutants at the national level was less than 0.8% of the total emissions from transport, the modal shares of road and rail being respectively 70% and 20% for freight transport and 80% and 8% for passenger transport. Road transport contributed between 94% and 99% of the total pollutants arising from transport. In the study, rail was found to be not only energy efficient but also with a lower emission coefficient of carbon dioxide (CO₂) for the same amount of energy consumed. For every hundred passenger kilometres (PKMS), a high-speed train consumes the equivalent of 1.8 litres of oil equivalent as primary energy and emits 0.4 kg CO₂. A private car, on the other hand, carrying an
average of 1.85 people, consumes 4 litres and emits 10 kg of this pollutant for every 100 PKMS. The percentage share of transport as a whole in the total emissions was found to be as follows: sulphur dioxide (SO₂) 13.5%; nitrogen oxides (NOx) 62.7%; non-methane volatile organic compounds (NMVOC), 41.7%; carbon monoxide (CO), 59.4%; and lastly, CO₂, 39.4%. The shares of road in the total transport emissions are as high as at least 90%. Specifically, they are as follows: SO₂, 94.8%; NOx, 93.9%; NMVOC, 95.1%; CO, 99%; and CO₂, 93.6%. The rail shares in the same total transport emissions are the following: SO₂, 0.8%; NOx, 0.8%; NMVOC, 0.2%; CO, 0.1%; and CO₂, 0.6%. The study works out hypothetical savings in pollution loads resulting from a transfer of road traffic to rail. If 1 billion PKMS are transferred from private car to rail travel (taking a mean load factor of 1.8 passengers per car), the resulting reductions in pollution (in tonnes) are of the following order: CO₂, 106860; CO, 3720; volatile organic compounds (VOOC), 620; NOx, 550; and suspended particulate matter (SPM), 55 tonnes. If 1 billion tonne-kilometres are transferred from heavy goods vehicles to rail haulage, the overall decreases in pollution levels are as follows: CO₂, 59450; CO, 420; VOOC, 100; NOx, 580; and SPM, 65 tonnes. It may be concluded, therefore, that any shift in favour of rail at the expense of road has significant implications in terms of savings in energy consumption and reductions in pollution loads.

An OECD study (Wiederkehr, 1988) found that for the OECD countries as a whole, the share of transport in polluting emissions varies from 25% for CO₂ to 90% for CO, with road transport contributing more than 80% of the total transport emissions. The so-called social costs of transport arising from the health and environmental effects of air pollution, noise, congestion, and time losses have been estimated between 5% and 10% of GDP of OECD countries. Road transport and aviation are primarily responsible for these costs, while the rail mode contributes less than 1% of the social cost burden. Estimates of the social costs from transport indicate that for cars the cost is almost 50 European Currency Units (ECU) per 1000 PKMS, whereas the social cost of passenger rail is 25 ECU per 1000 PKMS. The social cost of rail freight transport is 30 ECU per 1000 NTKMS, in contrast to a road freight average of 70 ECU per 1000 NTKMS.

The OECD study considers environmental impacts arising not just from the operation and use of transport means, but also from the production and maintenance of vehicles, the construction of infrastructure, the provision of energy and fuels, and the disposal and decommissioning of vehicles. Life-cycle assessments and material-
balance studies show that rail transport - including high-speed rail - causes considerably less environmental impacts than road and air traffic. The life-cycle emission of NOX due to passenger transport is about 1.1 grams per PKM for car, whereas for ordinary train the estimate is about 0.8 g/ PKM and for high-speed train the figure is about 0.25 g/ PKM. In respect of CO₂ emissions from passenger transport, car discharges about 180 g/ PKM, while ordinary train emits about 60 g/ PKM, and high-speed train sends forth about 70 g/ PKM.

Studies carried out in Australia (Australasian Railway Association, undated) show that road transport consumes nearly 90% of Australia’s transport energy requirements and produces 86% of the country’s transport greenhouse gas emissions. By contrast, rail transport consumes just 3% of the nation’s domestic transport energy consumption and constitutes only 2% of transport greenhouse gas emissions. Rail freight uses only one-third of the fuel required by road transport for the same freight task. Rail is still twice as energy efficient as road even after fuel use has been included for rail line haul, road pick-up and delivery from rail terminals, manufacture of transport equipment and construction of roads and railway lines. A rail line requires only one-third of the construction and maintenance costs of road, and one-third of a road lane space. With regard to accidents and safety, it is found that road transport accounts for over 90% of the cost of Australian transport accidents, while the cost of rail accidents is just 1% of the total cost. Rail freight transport is 7 times safer than road freight transport, with only 0.55 fatality per billion tonne-kilometres of freight hauled compared with the road freight record of 3.8 fatalities per billion tonne-kilometres.

It is clear that for equivalent volumes of traffic, rail enjoys superiority over other modes, including road, with respect to both energy use and environmental well-being. In view of the increasing concern expressed over the state of the environment and human health, it becomes necessary to examine ways in which the transport sector might be so structured as to make fewer demands on scarce energy reserves and to contribute less to environmental degradation and ensure greater safety.

The External Costs of Transport

The literature on externalities of transport has generally concentrated more on the external effects of infrastructure use than on those of infrastructure provision. The external costs of infrastructure use include the following: (i) traffic congestion, in the
sense of additional time and operation costs caused by user interactions; (ii) uncovered costs of the infrastructure provision (net of the share of the state); (iii) environmental impacts related to transport activities, such as noise, air, pollution, climate change, separation of communication between neighbourhoods, water and soil pollution as well as disamenities and detrimental effects of operations on the infrastructure; and (iv) traffic accidents that deplete the stock of human resources, the costs being relevant in as much as they are not covered by insurance (Rothengatter, 2000).

The external costs of congestion, accidents, noise, exhaust gas emissions and greenhouse gas emissions have generally been quantified in the literature. However, there are a number of further external cost components that are sometimes mentioned and quantified. Some of these external effects, which are usually treated as intangibles, are: (i) risk of cancer from particulate matter and benzenes; (ii) health risks from ozone; (iii) risks for flora and fauna; (iv) noise damage outside homes; and (v) risks for biotopes and biodiversity. Furthermore, risks from oil spills and chemical treatment of rail tracks are also mentioned. If a complete balance of externalities is to be given, then the upstream and downstream effects (production of vehicles, disposal of vehicles) have to be considered (Rothengatter, 2000). In what follows, we shall discuss the external costs of transport under three broad heads: congestion, accidents, and environmental impacts such as air pollution, climate change, noise and health damage.

**Congestion**

Although there are references to a number of studies in which the costs of traffic congestion are estimated, we have been able to obtain only one such study where they are considered exclusively. The other studies to be reviewed in this chapter mention congestion as an external cost but focus on other external costs such as air pollution and health damage. Mayeres, Ochelen and Proost (1996) consider the marginal external costs of congestion, accidents, air pollution and noise in the city of Brussels. Marginal congestion costs are present whenever an additional vehicle on the road reduces the speed of other road users. The authors deal with short run marginal congestion costs due to time losses of other road users. The parameters of an exponential congestion function are estimated for the urban region of Brussels. In this function, average speed \(s\) is related to traffic flow \(q\), which is measured in millions of passenger car units (PCUs) per hour. The congestion
function expresses the minutes needed to drive 1 kilometre in a certain period as a function of the million PCU per hour at that moment in the city.

\[
\frac{1}{s} = 1.194428 + 0.005571 \times (\exp(7.890545 \times q))
\]

Once the time loss suffered by other road users is worked out on the basis of this function, use is made of a willingness-to-pay study in order to work out the value of passenger time (VOPT) for road users. The VOPT in 2005 for car users moving in peak hours is 7.7 ECU/hour, for users of public transport 5.4 ECU/hour, and for truck users 34.2 ECU/hour. A linear relationship is estimated between VOPT and income; the elasticity of VOPT with respect to income is found to be 0.368. The marginal external costs of congestion in peak hours for Brussels in 2005 are as follows: car, 1.387 ECU/hours and bus, tram and truck, 2.774 ECU/hour.

**Environmental impacts**

(i) *Air pollution and health damage*

Greene and Duleep (1993) carry out an exercise in the estimation of the costs and benefits of technology-based improvements in fuel economy for automobiles and light trucks. The benefits quantified include the changes in the emissions of CO\textsubscript{2} and other pollutants. Vehicles that are more fuel-efficient certainly have lower hydrocarbon (HC) emissions because of reduction in evaporative rather than tailpipe emissions. On the other hand, more efficient vehicles tend to be driven more because of the lower fuel cost per mile, leading to increases in emission of pollutants. The authors consider three major criteria pollutants: hydrocarbons (HC), carbon monoxide (CO) and oxides of nitrogen (NO\textsubscript{X}). Emissions of carbon dioxide, the chief greenhouse gas produced by motor vehicles, are also estimated. A vehicle stock model provides estimates of miles driven that are sensitive to the age and efficiency of each vintage, as well as the price of gasoline in the forecast year. Emissions are obtained by multiplying predicted miles with the appropriate emission rate for each vintage and summing across vintages in each forecast year.

Greene and Duleep find that increased fuel economy tends to reduce HC and CO\textsubscript{2} emissions. It has been estimated that every one mile per gallon (MPG) improvement beyond the current average level of 28 MPG would result in appropriately 0.02 gram per mile reduction in total the HC emissions. Since the environmental and health costs of vehicle emissions are difficult to determine
because of uncertainties about impacts, the authors rely instead on the avoided costs of emission control technologies. The assumption underlying this approach is that society implicitly values emission reductions to be at least the cost of the most expensive measure it is using to control them. For the criteria pollutants, typical avoided costs per tonne are: (a) $300 for CO; (b) $2000 for NOX; and (c) $2900 to $3800 for HC. So far as CO₂ emissions are considered, the authors make use of several studies to arrive at a figure of $10 per tonne of carbon as a reasonable low cost estimate for a significant but not drastic reduction in CO₂ emissions. A higher but still plausible estimate of $100 per tonne of carbon is also used while $50 per tonne of carbon is a middle estimate.

We next consider a World Bank study that values the economy-wide impact of environmental degradation in India resulting from urban air pollution, surface and around water pollution, individual pollution and hazardous wastes, soil and range land degradation, deforestation, and tourism (Brandon, 1995). We shall concentrate on the economic valuation of the adverse impacts of urban air pollution, since these include emissions from the transport sector. The World Bank study cites another study for Mumbai, which estimated the relative contribution to PM₁₀ (i.e. particulate matter of aerodynamic diameter 10 microns) pollution as follows: refuse burning, 28%; vehicle exhaust, 25%; resuspended road dust, 17%; fuel burned in residences, 15%; fuel burned by industry and power, 12%; and others (including marine), 3%. It is seen that the transport sector (represented by vehicle exhaust and road dust above) is the major source of PM₁₀ pollution in the city. Ambient air pollution levels exceed WHO health standards in many of the Indian metropolitan cities. The situation is worsening because of upward trends in power consumption, industrialization, vehicle use, and refuse burning.

Six of the ten largest cities in India were found to have severe air pollution problems, with annual average levels of total suspended particulates (TSP) at least 3 times as high as the WHO standard. The study estimates the air-pollution related health impacts of particulates, SO₂, NOX and lead. These health impacts are estimated through the use of dose-response functions drawn from epidemiological studies done around the world. These functions relate the changes in the concentration of a particular pollutant to acute morbidity and mortality occurrences. It is recognised that by using dose-response functions estimated in cities in more developed countries, the estimates derived for India are likely to be conservative. On account of the lower standard of living, sanitation and health in India, there is a
higher percentage of the population in marginal health, more susceptible to negative health impacts from air pollution.

The results obtained for Indian air pollution health impacts are summarised in table 2.1. It shows the reductions in morbidity and mortality estimated to occur in 36 Indian cities if pollution levels were reduced to the prevailing WHO annual average standard. In addition, the social value of these impacts is worked out, using a lower and upper statistical value of a life saved as well as a cost-of-treatment approach that includes medical costs and the value of productive time lost (but does not value suffering).

The number of premature deaths that would be avoided is over 40,000, with the main metropolitan cities contributing the major share. The economic valuation of these premature deaths suggests a monetary estimate of loss of between $400 million and $1600 million. Besides, the reduction of particulate levels would be expected to reduce new cases of chronic lung disease and respiratory tract infections, especially in children. Asthma attacks would also be reduced. The analysis suggest that the lowering of pollutant levels to the WHO standard would lead, first, to almost 20 million fewer respiratory hospital admissions, emergency room visits, and sicknesses requiring medical treatment, and, second, to more than 1200 million fewer restricted activity days, respiratory symptom days, cases of lower respiratory illness, and other minor sicknesses. The estimate of the social value of both these impacts is $350-490 million per year. Of the total impact costs, premature mortality represents 77% of the total value, and morbidity 23%.

Table 2.1
Annual Health Incidence and Health Costs due to Ambient Air Pollution Levels Exceeding WHO Guidelines in 36 India Cities (using data from 1991-92)

<table>
<thead>
<tr>
<th></th>
<th>Physical Impacts (Number)</th>
<th>Cost Valuation (US $ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Premature Deaths</td>
<td>40,351</td>
<td>$388-1615</td>
</tr>
<tr>
<td>Hospital Admissions and Sickness</td>
<td>19,800,000</td>
<td>$25-50</td>
</tr>
<tr>
<td>Requiring Medical Treatment</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minor Sicknesses (including</td>
<td>1,201,300,000</td>
<td>$322-437</td>
</tr>
<tr>
<td>Restricted Activity Days and</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Respiratory Symptom Days)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td>-</td>
<td>$735-2102</td>
</tr>
</tbody>
</table>
Particulate matter and sulphur dioxide are responsible for over 95% of the health impact damage. The remaining 5% is contributed by the impact of high lead levels in a few Indian cities such as Mumbai, Kolkata and Delhi, leading to heart attacks, hypertension and children’s IQ loss. Carbon monoxide (CO) and ozone, two pollutants with significant health impacts, are not routinely monitored at the stations where other pollutants are measured. Some readings taken at traffic intersections show that CO emissions often exceed 5000 micrograms per cubic metre. Ozone contributes to hospital admissions on account of respiratory diseases, restricted activity, asthma, eye irritation and heart disease. There has not been sufficient ozone monitoring in India to allow estimates of such health impacts.

The cost of air pollution is highly correlated with city size. However, per capita air pollution costs are significantly higher in some of India’s secondary cities on account of usually high concentrations of fine particulates, probably due to large local point sources.

Rothengatter (1996) summarizes the results of a wide-ranging study on the external effects of transport in Europe, carried out by INFRAS (Zurich) and IWW (Karlsruhe, Germany) on behalf of the UIC (Union International des Chemins de Fer, Paris). The externalities investigated are uncovered accident costs, noise, air pollution, and climate effects. At the time of this comprehensive study, a number of country-based approaches to the valuation of externalities in Europe were available. However, it was not possible either to aggregate the different methodologies applied or to develop a new methodological approach from scratch. Therefore a unified method was adopted by using a reference cost model and adjusting it for country-specific conditions. The reference unit cost was then multiplied by the statistically measured quantity of the effect. Finally, the total costs were related to traffic volume statistics to arrive at relative costs for each type of effect.

The study evaluates the costs of air pollution on the basis of $\text{SO}_2$, NOX and volatile organic compounds (VOC) emissions and cover impacts on health, natural environment and material values. These costs, based on prevention assumptions and on damage estimations, are then found to be as follows: $\text{SO}_2$ (yearly emissions), 12994 ECU/tonne; NOX (summer emissions), 13475 ECU/tonne; and VOC (summer emissions), 12175 ECU/tonne. The total external costs of air pollution from transport in Europe are found to be 5865 billion ECU per annum, or 1.17% of GDP on the average.
The study by Mayeres, Ochelen and Proost (1996) mentioned in the section on congestion also seeks to determine the costs to society of a marginal increase in the emission of air pollutants by road transport. The focus lies on the health effects of ozone (O₃) and PM₁₀. A direct damage estimation approach is followed. The first step consists of establishing a relationship between a change in emissions and the resulting concentration levels of the different primary and secondary air pollutants. This requires atmospheric dispersion models which predict the spread of pollutants from their origin, and chemical transformation models which describe how different pollutants react to form so-called secondary air pollutants. Most problems are encountered in this first step of determining the link between emissions and concentrations, and one is forced to make many assumptions. The second step consists of relating the change in the concentration level to its effects on health, vegetation, materials, visibility and ecosystems. This requires the use of so-called dose-response relationships. On account of lack of data, the researchers include only the effects on public health and some vegetation effects in their study. The final step consists of determining a monetary value for the different effects of air pollution.

Mayeres, Ochelen and Proost take monetary values for the different health effects of PM₁₀ and O₃ from existing studies. The estimate for the value of a statistical life is the average value of a number of European contingent valuation method (CVM) studies. The monetary value of statistical life for 1990 is worked out at 2,600,000 ECU. The marginal external cost or willingness-to-pay for an illness has been taken to consist of three components: (i) the value of time lost due to the illness; (ii) the individual value of lost utility due to pain and suffering; and (iii) the expenditures for averting or mitigating the effects of the illness. By using data and models from a number of studies, and authors work out the cost of an increase in VOC emissions in terms of increase in the concentration of O₃ in the higher altitudes. The mid-value of the morbidity effect equals 0.034 mECU/g for Belgium and 0.395 mECU/g transnationally. (An mECU is one-tenth of an ECU.) NOX emissions also contribute to the formation of O₃. The calculations of the effects of increase in NOX emissions are based on a study of the health effects of emissions from a new power plant in Germany. The mid-value of the morbidity effect is then estimated at 0.135 mECU/g for Belgium. In the case of air pollutants generating concentrations of particulate matter of diameter 10 microns or less (PM₁₀), data from an existing study has been modified for the Belgian situation. The pollutants that contribute to PM₁₀ formation are VOC, NOX, SO₂, besides PM₁₀ itself. The authors estimate a health
cost of 83.19 mECU/g for PM\textsubscript{10}, 1.51 mECU/g for VOC, 7.55 mECU/g for NOX, and 93.70 mECU/g for SO\textsubscript{2}.

In an important study, Eyre et al. (1997) study fuel and location effects on the damage costs of transport emissions. These costs are estimated for three transport fuels – petrol, diesel and natural gas – in typical new light vehicles on road, operating in both rural and urban conditions. Direct emissions attributable to petrol, diesel and natural gas fuelled vehicles are first considered. Data on emission factors (in g/km) for petrol and natural gas vehicles are taken from the existing literature after suitable modifications. Emissions from diesel engines are based on the results of an extensive European Commission Programme. Both urban emissions and average emissions are considered. Urban cycle emissions are assumed to vary in accordance with fuel use. In order to obtain a comprehensive assessment of emissions attributable to the use of each fuel, the authors consider it necessary also to consider the life-cycle of each fuel. This involves the extraction, transportation, refining (or processing) and distribution of the fuel, which are all termed as upstream activities. Emission factors for all upstream fuel activities are derived for both average and urban conditions. These differ only to the extent of variation in fuel efficiency. Emissions from the use of a particular fuel are then obtained by using both the direct vehicle emission factors and the upstream emission factors.

Results from the electric power sector supplying energy for transport are not directly transferable to transport sector emissions primarily because power station emissions are usually from high stacks in rural areas whereas transport emission sources are invariably closer to ground level and frequently in urban areas. The paper develops simple procedures for transferring the damages of emissions from the electricity sector to the transport sector. In transferring results derived for power stations, exact calculations should ideally be undertaken of pollution dispersion and chemical transformation between each source and all potential receptors. However, this is not possible in practice and more approximate methods have to be used. For global-scale pollutants – the greenhouse gases – the impacts are generally independent of the source location. The same conclusion is not valid for short-range impacts. Urban receptors, typically people and buildings, are clustered. For sources in urban areas, there is a much higher than average receptor density around the source, and the short-range aggregate impacts may be comparable with or larger than those at longer range. The dose to human receptors in a conurbation from local emissions is worked out by using empirical relationships derived from modelling work.
in London. The results indicate that high-level sources in urban areas have only 25% of the aggregate short-range impact of equivalent low-level sources.

In the study by Eyre et al., the estimates of monetary damages for transport emissions cover both direct vehicle emissions and upstream emissions. The estimates have been transferred from other studies using the type of monetary valuation techniques developed in the environmental economics literature. For the emissions of the three greenhouse gases carbon dioxide, methane and nitrous oxide, recent estimates of monetary damage are used. The global warming impacts of carbon monoxide and non-methane volatile organic compounds (NMVOC) are included using global warming potentials relative to carbon dioxide of 1.8 and 11 respectively. Human health damage estimates are derived from dose-response functions. As far as forest damage resulting from acid deposition is concerned, the loss of timber growth across European forests is estimated at approximately 0.1 pence per gram of sulphur dioxide. The value for NOX is 0.07 p/g, based on its relative contribution to acid deposition. The costs of crop damage due to sulphur-dioxide and ozone are calculated from lost yield. The estimate of damage to building materials by sulphur dioxide and acid deposition is based on repair costs since acid-damaged buildings require more frequent repair and maintenance.

Eyre et al. make two sets of calculations in the estimation of the environmental costs of transport: (i) costs of transport emissions in rural areas, using total fuel life-cycle emissions for average driving conditions, and pollution damage prices for rural areas; and (ii) costs of transport emissions in urban areas, using total fuel life-cycle emissions for urban driving conditions, with pollution prices for urban areas applied to vehicle emissions and pollution damage prices for rural areas for upstream fuel cycle emissions. It is emphasized that uncertainties are sufficiently large and that various categories of environmental impact are not included in the assessment because of lack of sufficient information. These impacts include the contribution of NOX, particulates and aerosols to global warming, chronic health impacts, impacts of pollution on unmanaged terrestrial ecosystems, and impacts of acidification on freshwater systems.

The results derived by Eyre et al. show that in both rural and urban locations, the health damage costs of emissions from the natural gas vehicle fuel life-cycle are significantly lower than for the liquid fuels, especially diesel. The high values for diesel are caused mainly by higher emissions of particulates from diesel engines,
along with recent findings that particulates are the major cause of air-pollution related mortality and morbidity. The summary of health damage cost estimates for different types of fuel is given below:

Table 2.2
Summary of Damage Cost Estimates in Eyre et al. (1997)

<table>
<thead>
<tr>
<th>Fuel</th>
<th>Damage Cost (in pence/km)</th>
<th>Rural Emissions</th>
<th>Urban Emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gas</td>
<td></td>
<td>0.109</td>
<td>0.224</td>
</tr>
<tr>
<td>Petrol</td>
<td></td>
<td>0.341</td>
<td>0.797</td>
</tr>
<tr>
<td>Diesel</td>
<td></td>
<td>0.565</td>
<td>2.421</td>
</tr>
</tbody>
</table>

Despite the uncertainties in these numbers as well as the possibility of underestimation, it is still possible to arrive at some important conclusions: (i) damages to health and environment from vehicle emissions are significant compared to the cost of fuel; (ii) these costs are significantly higher in urban areas on account of a higher population density; and (iii) natural gas vehicles have significantly lower environmental costs than both petrol vehicles and diesel vehicles.

McCubbin and Delucchi (1999) study the health costs of motor-vehicle-related air pollution in the US. They seek to examine the cost of all the health effects of all pollutants from all emission sources related to motor-vehicle use. The researchers model the relationship between changes in emissions related to motor-vehicle use and changes in health costs in several steps: (i) estimation of emissions related to motor-vehicle use; (ii) estimation of changes in exposure to air pollution; (iii) relating changes in air-pollution exposure to changes in physical health effects; and (iv) relating changes in physical health effects to changes in economic welfare. Four kinds of sources of emissions related to motor-vehicle use are considered: (a) tailpipe and evaporative emissions from vehicles themselves; (b) emissions from upstream fuel and vehicle production processes; (c) particulate dust emissions from paved roads; and (d) particulate dust emissions from unpaved roads. The health effects of these sources of emission are estimated cumulatively since the ambient concentrations and health effects are nonlinear functions of the levels of emissions.

Next, as regards exposure to air pollution, McCubbin and Delucchi use the "micro environment" method to estimate exposure to toxic air pollutants, and ambient (outdoor) air-quality data to estimate exposure to criteria air pollutants – ozone (O₃), nitrogen dioxide (NO₂), carbon monoxide (CO) and particulate matter (PM). Estimation of exposure to these pollutants is done on the basis of ambient air quality
readings and population in US counties in 1990. Given the actual ambient pollutant concentration and using a simple model of emissions, dispersion and atmospheric chemistry, the researchers estimate what the concentration would have been had motor-vehicle-related emissions been reduced by 10% and 100%. The model has several simplifying assumptions: for example, given the baseline emissions or the change in emissions, the percentage change in air quality itself is directly estimated rather than the absolute air quality.

Ozone formation is assumed to be a simple nonlinear function of NOX and VOC emissions. With respect to the formation of secondary particulate matter, it is assumed that 15% to 25% (western US), and 25% to 35% (eastern US) of the sulphur oxides (SOX) emitted becomes sulphate, that 5% to 7% of the NOX emitted becomes nitrate, and that the resultant sulphates and nitrates react with ammonia (NH₃) to form secondary particulate sulphate and secondary particulate nitrate. Other assumptions relate to the neutralising of the sulphate, bisulphate and nitrates.

In respect of toxic air pollutants, the micro-environment method is followed, where, first, pollution levels in each micro-environment are estimated. Then the amount of pollution ingested (sometimes taking into account body weight and respiration rate) is estimated, and, finally, dose-response functions derived from clinical studies are used to estimate the health effects of the exposure. The results of a specific micro-environment model are used to estimate the effect of toxic air pollution on the incidence of cancer. In this model, the average exposure to pollution is the sum of the weighted daily micro-environment exposures, wherein the weights are proportional to the pollutant concentration and the length of time spent in each micro-environment.

To estimate the health effects of the criteria pollutants, McCubbin and Delucchi construct exposure-response (or dose-response) functions, typically using the results of logistic and Poisson regression analyses. To estimate the health effects of the toxic air pollutants, cancer unit risk estimates are used. The researchers review hundreds of clinical, animal and epidemiological studies of the health effects of various pollutants, and construct exposure-response functions for each criterion pollutant (O₃, CO, etc.), and each of a variety of health effects (e.g. asthma or headaches). These functions relate the change in health effects to the change in exposure. Upper- and lower-bound estimates of the effects of exposure are established for most pollutants and health effects. Particulate matter is the most
dangerous pollutant and perhaps the most complicated to model in terms of its formation and impact. When the total particulate damages to individual sources are apportioned, it is assumed that PM_{2.5} (i.e. particulate matter of diameter 2.5 microns or less) is 2-10 times as potent (in terms of damages per gram) as the coarse fraction of PM_{10}. Particulates larger than 10 microns, which generally are not deeply inhaled, are not harmful at all. Ideally, one should know the continuous size distribution of particles from each emission source, and the damages as a continuous function of size. However, there is not sufficient data to establish such functions. The mortality and chronic illness associated with particulate pollution, as well as the health effects of exposure to CO, NO\textsubscript{2} and O\textsubscript{3}, are estimated by using recent studies and making a number of assumptions.

To quantify the cancer risk from toxic air pollutants, the linear cancer model is used, which requires the estimation of a unit-risk number. This number indicates the excess risk of cancer, to one individual, from 70 years (a lifetime) of continuous exposure to one microgram per metre cubed (\(\mu g/m^3\)). To derive the number of excess cancers to a population from exposure to a toxic air pollutant, McCubbin and Delucchi determine the number of “exposure years” at an exposure level of one \(\mu g/m^3\). Multiplying the number of exposure-years by the unit-risk number then gives the number of cancer cases that would arise from exposure to motor vehicles. Epidemiological and laboratory results are used to specify the type of cancer caused by each toxic pollutant. It is assumed that, on average, cancer is discovered 25 years after exposure, and that the incidence of case follows, roughly, a normal distribution.

Finally, in the valuation of health effects, the results from the epidemiological literature are merged with the results from the economic literature. The health effects that are valued in the study are: acute morbidity, chronic morbidity, mortality and cancer. It is recognized that people may not value reduction of the Nth day of illness as much as the first, either because of income constraints or because of declining marginal utility of health benefits. Accordingly, use is made of a declining, nonlinear relationship between the average willingness-to-pay (WTP) to avoid one symptom day and the WTP to avoid many symptom days.

In the economic literature, several methods are used to estimate the cost of illnesses such as the common cold and asthma, which may be grouped under the category of acute morbidity. The “observed market” approach includes techniques
that rely on demand and cost functions, market prices and observed behaviour and choices. The "constructed market" approach includes techniques that directly ask people's WTP or to accept compensation for a postulated change, how their behaviour would change or how they would rank alternative situations involving different combinations of health and income or consumption. There are both advantages and disadvantages in the two approaches. As a result, the available valuation estimates are uncertain and often unreliable.

Since the imprecision of the available estimates of morbidity costs preclude a definitive point-estimate of the cost, McCubbin and Delucchi select a wide range of values from the literature for the valuation of acute morbidity. Chronic morbidity, on the other hand, refers to such diseases as chronic bronchitis and emphysema, which are caused by particulates. Using recent studies on the WTP to avoid a statistical case of chronic bronchitis, the authors take health values of $0.5 million and $2 million as the lower and upper bounds respectively. The health effects of three different kinds of mortality are estimated: acute harvest death, acute non-harvest deaths and chronic deaths. Acute deaths are those that occur only a relatively short time after exposure to particulate air pollution. Acute harvest deaths are those that would have occurred in a few days anyway, but are precipitated by exposure to particulate pollution. Non-harvest deaths are also precipitated by such exposure, but take a longer period to happen. Chronic deaths are those that occur many years after the initial exposure to particulate air pollution. The researchers estimate different values of life (VOL) for each type of death. For fatal cancer cases, the value of a statistical life at the point at which the cancer is discovered is assigned.

The striking features of the final results of the damages of motor-vehicle-related air pollution obtained by McCubbin and Delucchi are the large damages caused by ambient PM, and the large contribution of motor vehicles to ambient particulate levels. Ambient CO, O₃, nitrogen dioxide (NO₂), and toxics cause much smaller damages than does ambient PM. The damages from ambient O₃, which is the most heavily regulated criteria pollutant, are less than the damages from CO and NO₂, and two orders of magnitude less than the danger from ambient PM. The damages from PM are so high because PM pollution kills or chronically sickens far more people than do other pollutants. The table below shows the cost (in dollars) per kg of motor vehicle emission and upstream emissions, taken together.
Table 2.3
Cost Per Kilogram of Motor Vehicle Emissions in the USA in 1990

(1991 $)

<table>
<thead>
<tr>
<th>Emission</th>
<th>Ambient Pollutant</th>
<th>United States</th>
<th>Urban Areas</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Low</td>
<td>High</td>
</tr>
<tr>
<td>CO</td>
<td>CO</td>
<td>0.01</td>
<td>0.09</td>
</tr>
<tr>
<td>NOX</td>
<td>Nitrate-PM$_{10}$ and NO$_2$</td>
<td>1.10</td>
<td>16.21</td>
</tr>
<tr>
<td>PM$_{10}$</td>
<td>PM$<em>{2.5}$ and coarse PM$</em>{10}$</td>
<td>8.78</td>
<td>116.01</td>
</tr>
<tr>
<td>SOX</td>
<td>Sulphate-PM$_{10}$</td>
<td>2.80</td>
<td>22.60</td>
</tr>
<tr>
<td>VOC</td>
<td>Organic-PM$_{10}$</td>
<td>0.10</td>
<td>0.99</td>
</tr>
<tr>
<td>VOC+NOX</td>
<td>Ozone</td>
<td>0.01</td>
<td>0.11</td>
</tr>
</tbody>
</table>

Again, direct emissions of PM have the highest cost/kg, followed by SOX and NOX. Emissions of CO are the least costly per kg. These results apply to 1990 emission levels. The researchers admit that there is considerable uncertainty in the analysis, as reflected in the difference between lower- and upper-bound estimates. However, it is clear that, over a wide range of assumptions, damages from particulates dominate the total costs of the health effects of motor-vehicle air pollution. While suggesting a number of ways in which their research might be improved, McCubbin and Delucchi propose that regulatory and analytical efforts be broadened and redirected towards examining particulates.

(ii) Noise

The study summarized by Rothengatter and mentioned above also generates a complete data set for the population exposed to traffic noise that is classified by 5 levels (Rothengatter, 1996). A willingness-to-pay approach is applied, based on contingent valuation methods. With respect to the noise generated by railways, an increment is applied to take into account differing subjective evaluation of railway noise. The annual noise costs of transport are found to be 0.65 per cent of GDP on the average in Europe. The average noise costs for passenger transport are 4.5 ECU/1000 PKMS for cars, 4.2 ECU/1000 PKMS for buses, and 3.1 ECU/1000 PKMS for rail transport. The average noise costs for freight transport are 12.7 ECU/1000 PKMS for road and 4.7 ECU/1000 PKMS for rail transport. Again, there is a great variance in national cost values.

The study by Mayeres, Ochelen and Proost (1996) estimates the marginal external noise costs of transport in Brussels. One needs to determine the effects on the noise level of an additional km of a particular vehicle. From an existing study for Brussels, the following function is used to represent the index of noise in terms of energy mean sound level, on the assumption that the average street in Brussels has a U-shape.
\[ L_{eq}(A) = 53.9 + 10 \log(Q_{vl} + EQ_{pl}) - 10 \log 1 + k, \]

where \( L_{eq}(A) \) is the equivalent noise level in decibels at 2m from the facade, \( Q_{vl} \) stands for the flow of light vehicles (<3.5 tonnes) in vehicles/hour, \( Q_{pl} \) is the flow of heavy vehicles (>3.5 tonnes) in vehicles/hour. \( E \) is an equivalence factor between different types of vehicles. For slopes smaller than 2 per cent, one heavy vehicle is assumed to be equivalent to 10 light vehicles. The width between the facades (in metres) is given by \( l \). The term \( k \) is a correction factor for speed, which means that 1 decibel is added for each speed range of 10 km/hour above a speed of 60 km/hour.

Since the function is established to compute the noise in a particular street and not the average city-wide noise level, several assumptions are made to derive an average noise function from the information. The above function is suitably modified for this purpose. For the monetary valuation of noise nuisance, existing studies on the hedonic housing market method are used. After making certain assumptions concerning house lifetime and the number of exposed houses per km, Mayeres, Ochelen and Proost obtain a value of 0.6 ECU per decibel per street of 1 km. The total external noise cost for Brussels is the monetary value per decibel, multiplied by the noise level above a certain threshold, multiplied by the number of road kilometres where a noise externality is generated. The marginal external noise cost (in mECU per vehicle km) for Brussels in 2005 in peak hours is as follows: car, 1.141; bus, 14.1; and truck, 14.1.

(iii) **Global warming and climate change**

The wide-ranging study on transport costs in Europe estimated the cost of climate change (Rothengatter, 1996). The valuation of the cost of climate change is based on the prevention approach. Reduction targets are established by country according to a target allocation rule. The relevant pollutant is \( \text{CO}_2 \), for which an average unit cost estimate of 50 ECU per tonne of reduction is assumed, based on general fuel tax and technical measures. The growth scenario of \( \text{CO}_2 \) emissions is assumed at 2% per year between 1991 and 2010, and 1% per year between 2010 and 2040. The study finds that the overall prevention costs of climate change are 0.74% of GDP. For passenger transport, the costs are highest for aviation (slightly below 10 ECU per 1000 PKMS), followed by passenger cars (about 6 to 7 ECU per 1000 PKMS). Buses and rail show an average value slightly below 3 ECU/1000 PKMS. In respect of freight transport, aviation shows by far the highest value of about 50 ECU/1000 NTKMS, followed by road (10 ECU/1000 NTKMS), shipping
(slightly below 2 ECU/1000 NTKMS) and rail (about 1 ECU/1000 NTKMS). There are significant cost differences across countries due to different reduction targets, economic conditions and the share of fossil fuel-based power generation for rail traction.

Mayeres, Ochelen and Proost (1996) study the effect of global warming as a result of increased CO₂ emissions. Use is made of a study where the marginal cost of carbon emissions in the period 2000-2010 is estimated. This estimate is the expected value of present and future damages resulting from a unit increase in emissions in this period. The damages contain diverse items: coastal defence, dryland loss, wetland loss, ecosystem loss, agriculture, forestry and fishery losses, gains and losses in energy and water supply, life and morbidity effects, air pollution damages, migration costs, and an estimate of natural hazard damages. The marginal damage for the world is $22.8 per tonne of carbon. This estimate is converted into an estimate for the European Union by using the relative share of damages of the EU in world damages for the case of doubling of CO₂ concentrations. Estimates for Belgium are obtained by assuming that, within the European Union, damages are distributed proportionally to GDP. Translating the values for 1990 into values for 2005, Mayeres, Ochelen and Proost obtain a cost of 0.00021 mECU/g carbon at the Belgian levels, 0.00669 mECU/g at the European levels and 0.02835 mECU/g at the world level. If the health costs of emissions, effects on vegetation, and global warming are considered together, then the marginal social costs (middle estimates) of Belgian emissions of air pollutants in 1990 are as follows: PM₁₀, 83.19 mECU/g; NOX, 13.8 mECU/g; VOC, 2.95 mECU/g; and SO₂, 95.21 mECU/g. The authors caution that one should keep in mind the assumptions made to derive these estimates and the uncertainties associated with them.

**Accidents**

The study by Mayeres, Ochelen and Proost (1996) of the marginal external costs of urban transport in Brussels also works out the marginal external accident costs for the city. The costs of the direct economic effects of death or injury are relatively easy to determine since they are directly observable effects. On the other hand, the willingness-to-pay on the part of the road user as well as his friends and relatives to avoid an accident is not observable, and the only way to determine them is on the basis of revealed or stated preferences for risk reductions. Accident risk is expressed as the ratio between the number of victims and the number of vehicle-
kilometres of the transport mode concerned. The different accident cost categories include the costs of mortality, serious injury, light injury and material damage. The resulting marginal external accident costs for the transport situation in Brussels in 2005 are worked out for car, bus, train, metro and truck in peak and off-peak hours, for both occupants and non-occupants. In peak hours, the total marginal external accident costs (in mECU per vehicle-kilometre) for the various types of vehicles are as follows: car, 98.29; bus, 854.02; tram, 616.58; metro, 7.76; and truck, 268.27.

Persson and Odegaard (1995) present a model of road traffic accident pricing in which the marginal external cost of an accident is basically computed as the product of the marginal cost per accident multiplied by the marginal accident risk. The marginal accident risk is assumed to be equal to the average accident risk. When the risk of death or injury has been assessed and taken into account by the occupants of a motor vehicle, it can be argued that human cost, derived as the implicit value from the willingness-to-pay to reduce the actual risk to zero of a representative motorist, should not be counted as an external cost. On the other hand, for the non-occupants or the unprotected group of road users, the cost of lives lost or impaired should be counted as an external cost and charged to the account of the vehicle involved. In the authors' model, the risk factors for occupant and non-occupant groups are assumed to be different.

The study compares the external costs of road traffic accidents in Sweden, Finland, Denmark, the UK, Germany and Switzerland and finds substantial differences in the external costs per vehicle-kilometre of road use in these countries. If risk posed to road users by different types of motor vehicles is defined as the number of persons killed/injured per vehicle-kilometre travelled in a particular vehicle, then it is found that passenger cars and motorcycles primarily pose a hazard to the occupants of those vehicles, whereas buses and lorries primarily pose a hazard to the non-vehicle users. Differences in settlement pattern and population density in the different countries lead to differences in the incidence of road traffic accidents. In densely populated countries, the hazard posed by passenger cars to the non-occupant group of road users is higher than that in less densely populated countries. The risk of being killed by a bus is some 5 to 8 times higher for the non-occupant group of road users than for the bus passengers. The risk of suffering injuries, however, is greater for bus passengers than for other road users in the more densely populated countries. This is due to the higher load factor of bus in these countries. The hazard road users face from lorries is also found to differ substantially in the
countries compared. In general, lorries pose a greater risk to the non-occupant group of road users than vans do. It appears that the hazard a lorry poses to the non-occupant group of road users increases with its weight. When accident externality prices are estimated, it is found that for cars the external cost per 1000 km of road use is lowest in the sparsely populated countries and highest in the densely populated ones. In the case of lorries, the reverse is the case. Lorries and buses generate substantially higher external costs than passenger cars on account of the greater risks faced by non-occupant road users. The study makes a recommendation for an extra charge to internalise the external accident costs, especially in the case of lorries and buses.

In the study reviewed by Rothengatter (1996), administrative costs (costs for police, justice and insurance), medical costs, costs of replacement and reintegration of victims, net production loss due to fatality or injury and loss in “human value” are included in the cost of accidents. Human value refers to the value of a human life as well as to the loss of utility caused by pain and suffering. In the case of a fatality, net production loss is assumed to be the lost working time until the statistical end of lifetime, while in the case of an injury, it is assumed to be lost working time (82 days). An adjusted European value of life of 1.1 million ECU is used in the computation of accident costs. The study finds that 99 per cent of the total accident costs are caused by the road sector, and less than 1 per cent by rail. The “human value” component covers 91 per cent of the accident costs for a fatality and 96 per cent for injury. The annual external costs of accidents are worked out at 148 billion ECU, which is 2.5% of GDP. The average European accident costs for passenger transport are 32 ECU/1000 passenger kilometres (PKMS) for cars, 9 ECU/1000 PKMS for buses and 1.9 ECU/1000 PKMS for trains. The accident costs for freight transport are 22 ECU/1000 NTKMS (net tonne kilometres) for trucks and 0.9 ECU/1000 NTKMS for trains. There are strong variations in country-specific costs resulting from very different accident rates.

Estimates of Total External Costs of Transport

In the study by Mayeres, Ochelen and Proost (1996), the final marginal external costs (in ECU per vehicle-km) of transport moving in peak hours for Brussels in 1991 are as follows: small gasoline car, 0.385; small diesel car, 0.405; bus, 1.661; tram, 1.771; and truck, 1.560. These costs include marginal external air pollution costs (including health impacts of air pollutants, effects of pollutants on crops and
vegetation, and global warming), marginal external accident costs, and marginal external noise costs. For 2005, the values are 1.511 (small gasoline car), 1.508 (small diesel car), 4.042 (bus), 4.278 (tram) and 3.213 (truck). The lower 1991 values are due to the lower level of traffic (and the corresponding higher speed), the use of cars not equipped with three-way catalytic converters, and lower levels of income. In 2005, the external congestion costs dominate the external cost estimates. The marginal external cost estimates for the same year are valid only for the particular transport situation in Brussels under unchanged policy conditions. Although the study is clearly useful, the authors concede that the findings are subject to many uncertainties which depend primarily upon the quality of the data used in the technical relationships and less upon the quality of the valuation techniques.

In Eyre et al. (1997), the total damage costs (covering health impacts, damage to crops, global warming, damage to building materials, and damage to timber) of urban emissions from the transport sector are 1.060, 0.375, and 2.717 pence/km for petrol, gas and diesel respectively. Thus the cost of emissions from the natural gas life-cycle are significantly lower than for the liquid fuels. It is found that the only category for which natural gas produces the highest damages is the effect of methane emissions on global warming. High losses of methane occur in the upstream part of the gas life-cycle. The high damage costs of diesel are caused mainly by the higher emissions of particulates from diesel engines, as noted in the section on air pollution and health damage. The difference between petrol and natural gas costs arises largely from lower emissions of SO\textsubscript{2} and NO\textsubscript{X} for the latter, and consequent lower impacts on health, timber and building materials. The results by Eyre et al. also show that damages other than global warming due to emissions of vehicles using natural gas are 2.5 to 10 times lower than similar damages from conventionally fuelled vehicles.

The study described by Rothengatter (1996) finds that the total external costs of transport in 17 European countries sum to about 270 billion ECU (European Currency Units) for the year 1991. This corresponds to 4.6% of national GDP on the average. The individual country values vary considerably. The external costs of road transport comprise 92% of the total external costs of transport; cars account for 60%, buses 3% and freight 21% of the total costs. The share of the rail mode in the total external costs is only about 2%, passenger movement and freight movement being roughly equal in their contribution to total external costs.
Having discussed some of the literature on transport demand and the external costs of transport, and observing the significantly lower contribution of the rail mode to these costs for similar volumes of traffic, we shall in the next chapter review the trend in the modal split between the rail and road modes in India, before inquiring into the factors behind the dramatic shift of traffic away from rail.