COMPETITIVE AND SUSTAINABLE GROWTH (GROWTH) PROGRAMME

UNITE

UNIfication of accounts and

marginal costs for Transport Efficiency

Deliverable 11
Environmental Marginal Cost Case Studies

Version 2.0
24 January 2003

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1 Summary

Marginal environmental costs were assessed for a number of specific routes in urban areas and important inter-urban relations, covering both passenger and goods transport. All modes were covered, and a broad range of vehicle types was considered for which costs related to the emission of air pollutants, greenhouse gases and noise proved to be relevant and quantifiable cost categories.

For quantifying the costs due to airborne pollutants and noise exposure the Impact Pathway Approach was applied. It comprises the steps
– emission calculation,
– dispersion and chemical conversion modelling of air pollutants / noise propagation,
– calculation of physical impacts, and
– monetary valuation of these impacts.

One of the case studies replaced the dispersion modelling step with the statistical correlation of vehicle mileage with measurement data for carbon monoxide and noise levels. The other case studies were performed with a common set of computer tools for impact assessment. These studies quantified marginal environmental costs as the costs caused by an additional vehicle or train or vessel driving on a specific route. For noise costs, besides route and vehicle characteristics the time of day is relevant, due to the sensitivity of the receptors (which at night is different from during the day) and the high importance of the background noise level, which varies with traffic density. The following table gives an overview of the locations and modes covered as well as the character of the case studies performed.

Overview of marginal cost case studies performed

No.	Country	Location	Modes considered	Character
9A	Finland	Helsinki	Road transport	Urban drive with a passenger car.
9B	Finland	Helsinki – Turku	Road transport	Inter-urban goods transport with a heavy goods vehicle in the south of Finland.
9C	Finland / Estonia	Helsinki – Tallinn	Maritime shipping	Passenger ferry in the baltic sea.
9D	Germany	Stuttgart, Berlin	Road transport, Rail transport	Urban passenger and goods transport in two German cities, one located in the south west, one in the east of Germany.
9E	Germany	Basel – Karlsruhe, Strasburg – Neubrandenburg	Road transport Rail transport	Inter-urban passenger and goods transport on two routes, one located in the south west, one in the north east of Germany.
9F	Germany / UK	Berlin-Tegel – London-Heathrow	Aircraft transport	Flight from Berlin to London, incl. departure, cruising and arrival.
9G	Italy	Florence	Road transport	Urban transport. Methodological approach differs from the other case studies.
9H	Italy	Milano – Chiasso Bologna – Brennero	Road transport Rail transport	Inter-urban passenger and goods transport on two routes in the northern part of Italy
0A	Netherlands / Germany	Rotterdam – Mannheim	Inland waterway transport	Container transport by barge on the Rhine.

The results show significant variations between the locations studied, reflecting the different characters and conditions of the relations. Besides the magnitude of total costs, the relative shares of air pollution, noise and global warming vary to different degrees.

- Costs due to the emission of greenhouse gases are not location specific, as they are relevant on a global scale. As a consequence all the variation is caused by the emission factor of a vehicle or the underlying electricity production process.
- Besides variations in emission factors the costs due to airborne pollutants are determined by the population density close to the road, the local meteorology (mainly by the average wind speed) and by the geographical location within Europe, which is important for the number of the population affected by long-range pollutant transport and the formation of secondary pollutants.
- Noise costs are mainly determined by the population affected, the time of day (with higher disturbance effects of noise during the night) as well as the number of vehicles and their speeds, and the resulting background noise. The higher the existing background noise level is, the lower the costs of an additional vehicle. Marginal noise costs of maritime shipping and inland waterway transport were found to be negligible.

It can be concluded that it is not possible to derive one single value for the marginal costs of a certain vehicle type. The explanations above indicate that the cost categories have to be distinguished when aiming at the generalisation of values.

For global warming generalisation of costs is no problem, because the cost factor per tonne of CO_2 emitted is applicable to all countries of the European Union.

The situation is more difficult for air pollution and noise. The comparison of the case study results clearly suggests that costs due to air pollution cannot be transferred based on the population density of the local environment only. Some general rules could be derived, but an operational formula for transfer requires a broader statistical basis of case studies. A generalisation methodology for air pollution costs should account for

- the local scale conditions (population density and local meteorology), and
- the regional scale costs per tonne of pollutant emitted in a certain area (e.g. NUTS1 level)

For marginal noise costs a generalisation is even more difficult, because of the large non-linearities involved and the variability of the relevant parameters in very short time spans.

The possible effect of marginal cost pricing in the case of noise is a very good illustration of issues, that have to be taken into account in the case of strong non-linearities of impacts. A pricing scheme based on marginal noise costs would lead to a bundling effect of traffic. Marginal costs are strongly decreasing with increasing background noise levels. For this reason driving on a route where noise levels are high already would be much cheaper than driving in quiet areas. Of course this is a perfect solution for allocation of noise emitters from the perspective of economic theory. However, it has to be ensured that no absolute limits, such as thresholds for health risks or amenity losses are exceeded. In practice, other price components (e.g. air pollution costs) may attenuate the bundling effect.

The impact pathway approach (IPA) for air pollution, including the respective models, exposure-response functions and monetary values, is well established and has been applied in a large number of research projects. The application of the IPA in the context of noise is relatively new and may be subject to revision and extension in the future, in particular the exposure-response functions. The results reflect best current knowledge; however this knowledge has gaps and therefore the results are subject to uncertainty

2 Introduction

Environmental external effects of transport cover a wide range of different impacts, including the various impacts of emissions of noise and a large number of pollutants on human health, materials, ecosystems, flora and fauna. Most early studies on transport externalities followed a top-down approach, giving average costs rather than marginal costs. The basis for the calculation is a whole geographical unit, a country for example. For such a unit the total cost due to a burden is calculated. This cost is then allocated based on the shares of total pollutant emissions, by vehicle mileage, etc. But marginal environmental costs of transportation vary considerably with the technology of a vehicle, train, ship or aircraft and site (or route) characteristics. Only a detailed bottom-up calculation allows a close appreciation of such site and technology dependence.

In the ExternE project series (see e.g. European Commission (1999a,b), Friedrich and Bickel (2001)) funded by the European Commission the Impact Pathway Approach (IPA) has been developed, which meets these requirements. In ExternE the impact pathway approach was applied for assessing impacts due to airborne emissions. Starting with the emission of a burden, through its diffusion and chemical conversion in the environment, impacts on the various receptors (humans, crops etc.) are quantified and, finally, valued in monetary terms. In other words, information is generated on three levels: i) the increase in burden (e.g. additional emissions and ambient concentration of SO_2 in $\mu g/m^3$) due to an additional activity (e. g. one additional trip on a specific route with a specific vehicle, train, ship, aircraft), ii) the associated impact (e.g. additional hospital admissions in cases) and iii) the monetary valuation of this impact (e.g. WTP to avoid the additional hospital admissions in Euro). Within the UNITE project the IPA has been extended to the quantification of noise impacts and applied to a large number of case studies in Finland, Germany, Italy, and the Netherlands.

The IPA for air pollutants and its results have been applied in a number of research projects and policy application related studies, e.g. INFRAS/IWW (2000), European Commission (1998), AEA (1997). Other studies (e.g. WHO (1999), McCubbin and Delucchi (1996)) as well looked at the chain of ambient pollutant concentrations due to the transport sector, human health impacts and monetary valuation. But in contrast to ExternE, the pathway analysed did not include detailed modelling of vehicle emissions at specific locations.

Damages due to climate change is one of the most important categories of fossil fuel emission related damages, but also amongst the most uncertain and controversial. First estimates were presented by Cline (1992), Fankhauser (1995), Nordhaus (1991), and Titus (1992). Tol (2001) estimated climate change impacts with a dynamic approach consistent with the ExternE methodology. Due to the high uncertainties involved in estimating damage costs due to climate change, many studies (e.g. INFRAS/IWW, 2000) have used abatement cost estimates.

As modelling of the noise nuisance is a challenging task, most of existing studies do bypass noise modelling and allocate damages to different vehicle categories, based on rough assumptions. Such estimates lead to average instead of marginal costs. ECMT (1998) gives a broad overview of studies carried out in different European countries.

Chapter 3 describes the methodological approaches applied in the case studies for the different cost categories considered. In Chapter 4 an overview of the case studies is given as well as a summary of the main results for each case study. Chapter 5 analyses the range of case study results in the context of generalisation of methodology and results, before in the final chapter the results are discussed and conclusions are drawn.

3 Description of Methodology Used

The valuation conventions used in the case studies are the same as for the UNITE country accounts. For air pollution the models applied are the same as for the country accounts, therefore the results are consistent. Differences may occur for non-linear effects, where the absolute amount of pollutant emission plays a role. Furthermore, the number of pollutants assessed may be different, depending on data availability in the accounts. Global warming costs are fully consistent with the country accounts, as the same valuation factor of €20 per tonne of CO_2-equivalent emitted was applied. In the case of noise, population exposure estimates used in the country accounts stems from existing studies, which are not directly comparable with the exposure estimates calculated in the case studies. Exposure-response functions for noise related health effects and monetary values applied were the same for country accounts and case studies. So in general, case study results and results presented in the country accounts are consistent. However, the case study results are based on more detailed data and calculations than the country accounts; the country accounts show a national average, whereas the case studies cover specific vehicles at specific locations. Therefore, case study results may vary considerably and are not directly comparable to average costs, which can be derived from the country accounts.

With the exception of case study 9G (“Urban road and rail studies: The case of Florence”), all case studies are based on the same methodology, which is described in the following sections.

In the Florence case study instead of using dispersion models for air pollutants, a statistical analysis of traffic flows and measurement data of CO concentrations was performed. Noise exposure estimates were based on noise measurement data from the Florence municipality and the Regional Agency for Environmental Protection. A detailed description of the approaches applied in the Florence case study are given in Annex G.

In the context of the case studies marginal environmental costs are quantified as the costs caused by an additional vehicle or train or vessel driving on a specific route. For noise costs the time of day is relevant as well, due to the sensitivity of the receptors (which at night is different from during the day) and the high importance of the background noise level, which varies with traffic density. The following, description of the methodology is focussed on road transport, but it is applicable analogously for the other modes. Specific treatment of other modes is described where applicable.

The approach of looking at the impacts of one additional vehicle requires a detailed bottom-up approach. The methodology follows as far as possible the Impact Pathway Approach, which is described in the following sections. For more detailed information see European Commission (1999a,b) and Friedrich and Bickel (2001).

3.1 Air Pollution

The starting point for the bottom-up approach for quantification of marginal costs is the micro level, i.e. the traffic flow on a particular route segment. Then, the marginal external costs of one additional vehicle are calculated for a single trip on this route segment. This is done by modelling the path from emission to impact and costs. Results of recent bottom-up calculations (see e.g. Friedrich and Bickel, 2001) have shown that the value of externalities may differ substantially from one transport route to another.

For quantifying the costs due to airborne pollutants the Impact Pathway Approach was applied. It comprises the steps
– emission calculation,
– dispersion and chemical conversion modelling,
– calculation of physical impacts, and
– monetary valuation of these impacts.
These steps are described in more detail in the following sections.

3.1.1 Emissions/burdens

In the first step the emissions from an additional vehicle on a specific route are calculated. For consistent treatment of different modes, the system boundaries considered are very important. Obviously, it makes no sense to treat electric trains as having no airborne emissions from operation. Instead, the complete chain from coal, crude oil, etc. extraction up to the fuel or electricity consumption has to be taken into account. For this reason, emissions due to fuel production processes were considered for all modes.

3.1.2 Concentrations

To obtain marginal external costs, the changes in the concentration and deposition of primary and secondary pollutants due to the additional emissions caused by the additional vehicle have to be calculated. The relation between emission and concentration of pollutants are highly non-linear for some species (e.g. secondary particles). So, air quality models that simulate the transport as well as the chemical transformation of pollutants in the atmosphere are used.

Depending on the range and type of pollutant considered different models are applied: The Gaussian dispersion model ROADPOL for calculation of pollutant concentrations from line sources on the local scale up to 25 km from the road (Vossiniotis et al., 1996); the Wind rose Trajectory Model (WTM) is used to quantify the concentration and deposition of non-reactive pollutants and acid species on a European scale (Trukenmüller and Friedrich, 1995); the Source-Receptor Ozone Model (SROM), which is based on source-receptor (S-R) relationships from the EMEP MSC-W oxidant model for five years of meteorology (Simpson et al., 1997), is used to estimate changes in ozone concentrations on a European scale.

These models are applicable for emissions up to the mixing layer height, which is typically around 800 m. Pollutants emitted in higher altitudes, i.e. cruising emissions from aircraft, have to be treated with different, more complex and thus expensive models, which were not available for the case study.

The consistent use of the same impact model to calculate airborne emissions from all transport modes ensures the comparability of the results across modes. This is especially important when comparing road transport with electrified rail transport, where the latter only produces air emissions from power plants as a point source. Thus the country specific fuel mix used to generate the electricity for the railway system or the railway company specific fuel mix has to be considered. The modelling approach for rail traffic emissions is consequently similar to the energy sector.

Impacts due to diesel trains, maritime and inland waterway vessels as well as aircraft can be quantified with the same approach as road transport vehicles with internal combustion engine by making adjustments as necessary, e.g. emission height.

3.1.3 Impacts

Concentrations then translate into impacts through the application of exposure-response functions, which relate changes in human health, material corrosion, crop yields etc. to unit changes in ambient concentrations of pollutants.

Exposure-response functions come in a variety of functional forms. They may be linear or non-linear and contain thresholds (e.g. critical loads) or not. Those describing effects of various air pollutants on agriculture have proved to be particularly complex, incorporating both positive and negative effects, because of the potential for certain pollutants, e. g. those containing sulphur and nitrogen, to act as fertilisers.

The dose-response functions used within UNITE are the final recommendations of the expert groups in the final phase of the ExternE Core/Transport project (Friedrich and Bickel 2001). The following table gives a summary of the dose-response functions as they are implemented in the EcoSense version used for this study.

Table 1
Health and environmental effects included in the analysis of air pollution costs

Impact category	Pollutant	Effects included
Public health – mortality	$PM_{2.5}$, PM_{10} [1)] SO_2, O_3	Reduction in life expectancy due to acute and chronic mortality Reduction in life expectancy due to acute mortality
Public health – morbidity	$PM_{2.5}$, PM_{10}, O_3	respiratory hospital admissions
		restricted activity days
	$PM_{2.5}$, PM_{10} only	cerebrovascular hospital admissions
		congestive heart failure
		cases of bronchodilator usage
		cases of chronic bronchitis
		cases of chronic cough in children
		cough in asthmatics
		lower respiratory symptoms
	O_3 only	asthma attacks
		symptom days
Material damage	SO_2, acid deposition	Ageing of galvanised steel, limestone, natural stone, mortar, sandstone, paint, rendering, zinc
Crops	SO_2	Yield change for wheat, barley, rye, oats, potato, sugar beet
	O_3	Yield loss for wheat, potato, rice, rye, oats, tobacco, barley, wheat
	Acid deposition	increased need for liming
	N	fertiliser effects

[1)] including secondary particles (sulphate and nitrate aerosols).

Source: IER.

Impacts on human health

Table 2 lists the exposure response functions used for the assessment of health effects. The exposure response functions are taken from the 2nd edition of the ExternE Methodology report (European Commission 1999a), with some modifications resulting from recent recommendations of the health experts in the final phase of the ExternE Core/ Transport project (Friedrich and Bickel 2001).

Table 2
Quantification of human health impacts due to air pollution[1)]

Receptor	Impact Category	Reference	Pollutant	f_{er}
ASTHMATICS (3.5% of population)				
Adults	Bronchodilator usage	Dusseldorp et al., 1995	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.163 0.163 0.272 0.272
	Cough	Dusseldorp et al., 1995	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.168 0.168 0.280 0.280
	Lower respiratory symptoms (wheeze)	Dusseldorp et al., 1995	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.061 0.061 0.101 0.101
Children	Bronchodilator usage	Roemer et al., 1993	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.078 0.078 0.129 0.129
	Cough	Pope and Dockery, 1992	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.133 0.133 0.223 0.223
	Lower respiratory symptoms (wheeze)	Roemer et al., 1993	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.103 0.103 0.172 0.172
All	Asthma attacks (AA)	Whittemore and Korn, 1980	O_3	4.29E-3
ELDERLY 65+ (14% of population)				
	Congestive heart failure	Schwartz and Morris, 1995	PM_{10} Nitrates $PM_{2.5}$ Sulphates CO	1.85E-5 1.85E-5 3.09E-5 3.09E-5 5.55E-7
CHILDREN (20% of population)				
	Chronic cough	Dockery et al., 1989	PM_{10} Nitrates $PM_{2.5}$ Sulphates	2.07E-3 2.07E-3 3.46E-3 3.46E-3
ADULTS (80% of population)				
	Restricted activity days (RAD)	Ostro, 1987	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.025 0.025 0.042 0.042
	Minor restricted activity days (MRAD)	Ostro and Rothschild, 1989	O_3	9.76E-3
	Chronic bronchitis	Abbey et al., 1995	PM_{10} Nitrates $PM_{2.5}$ Sulphates	2.45E-5 2.45E-5 3.9E-5 3.9E-5
ENTIRE POPULATION				
	Chronic Mortality (CM)	Pope et al., 1995	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.129% 0.129% 0.214% 0.214%
	Respiratory hospital admissions (RHA)	Dab et al., 1996	PM_{10} Nitrates $PM_{2.5}$ Sulphates	2.07E-6 2.07E-6 3.46E-6 3.46E-6
		Ponce de Leon, 1996	SO_2 O_3	2.04E-6 3.54E-6
	Cerebrovascular hospital admissions	Wordley et al., 1997	PM_{10} Nitrates $PM_{2.5}$ Sulphates	5.04E-6 5.04E-6 8.42E-6 8.42E-6
	Symptom days	Krupnick et al., 1990	O_3	0.033
	Cancer risk estimates	Pilkington et al., 1997; based on US EPA evaluations	Benzene Benzo-[a]-Pyrene 1,3-buta-diene Diesel particles	1.14E-7 1.43E-3 4.29E-6 4.86E-7
	Acute Mortality (AM)	Spix et al. / Verhoeff et al.,1996	PM_{10} Nitrates $PM_{2.5}$ Sulphates	0.040% 0.040% 0.068% 0.068%
		Anderson et al. / Touloumi et al., 1996	SO_2	0.072%
		Sunyer et al., 1996	O_3	0.059%

1) The exposure response slope, f_{er}, has units of [cases/(yr-person-µg/m^3)] for morbidity, and [%change in annual mortality rate/(µg/m^3)] for mortality. Concentrations of SO_2, PM_{10} , PM_{10}, sulphates and nitrates as annual mean concentration, concentration of ozone as seasonal 6-h average concentration.

Source: Friedrich and Bickel 2001.

Impacts on building materials

Impacts on building material were assessed using the most recent exposure-response functions developed in the last phase of the ExternE Core/Transport project (Friedrich and Bickel, 2001). This work includes the latest results of the UN ECE International Co-operative Programme on Effects on Materials (ICP Materials) for degradation of materials, based on the results of an extensive 8-year field exposure programme that involved 39 exposure sites in 12 European countries, the United States and Canada (Tidblad et al., 1998).

Limestone:
maintenance frequency: $1/t = [\ (2.7[SO_2]^{0.48}e^{-0.018T} + 0.019Rain[H^+])/R\ ]^{1/0.96}$

Sandstone, natural stone, mortar, rendering:
maintenance frequency: $1/t = [\ (2.0[SO_2]^{0.52}e^{f(T)} + 0.028Rain[H^+])/R\ ]^{1/0.91}$
f(T) $f(T) = 0$ if $T < 10\ ^oC$; $f(T) = -0.013(T-10)$ if $T \geq 10\ ^oC$

Zinc and galvanised steel:
maintenance frequency: $1/t = 0.14[SO_2]^{0.26}e^{0.021Rh}e^{f(T)}/R^{1.18} + 0.0041Rain[H^+]/R$
f(T) $f(T) = 0.073(T-10)$ if $T < 10\ ^oC$; $f(T) = -0.025(T-10)$ if $T \geq 10\ ^oC$

Paint on steel:
maintenance frequency: $1/t = [\ (0.033[SO_2] + 0.013Rh + f(T) + 0.0013Rain[H^+])/5\ ]^{1/0.41}$
f(T) $f(T) = 0.015(T-10)$ if $T < 10\ ^oC$; $f(T) = -0.15(T-10)$ if $T > 10\ ^oC$

Paint on galvanised steel:
maintenance frequency:
$1/t = [\ (0.0084[SO_2] + 0.015Rh + f(T) + 0.00082Rain[H^+])/5\ ]^{1/0.43}$
f(T) $f(T) = 0.04(T-10)$ if $T < 10\ ^oC$; $f(T) = -0.064(T-10)$ if $T \geq 10\ ^oC$

Carbonate paint:
maintenance frequency: $1/t = (0.12 \cdot (1 - e^{\frac{-0.121 \cdot Rh}{100 - Rh}}) \cdot [SO_2] + 0.0174 \cdot [H^+])/R$

with	1/t	maintenance frequency in 1/a
	$[SO_2]$	SO_2 concentration in µg/m^3
	T	temperature in ^oC
	Rain	precipitation in mm/a
	[H+]	hydrogen ion concentration in precipitation in mg/l
	R	surface recession in µm
	Rh	relative humidity in %

Impacts on crops

Effects from SO2
For the assessment of effects from SO_2 on crops, an adapted function from the one suggested by Baker et al. (1986) is used as recommended in ExternE (European Commission, 1999c). The function assumes that yield will increase with SO_2 from 0 to 6.8 ppb, and decline thereafter. The function is used to quantify changes in crop yield for wheat, barley, potato, sugar beet, and oats. The function is defined as

$y = 0.74 \cdot C_{SO2} - 0.055 \cdot (C_{SO2})^2$ for $0 < C_{SO2} < 13.6$ ppb
$y = -0.69 \cdot C_{SO2} + 9.35$ for $C_{SO2} > 13.6$ ppb
with y = relative yield change
C_{SO2} = SO_2-concentration in ppb

Effects from ozone
For the assessment of ozone impacts, a linear relation between yield loss and the AOT 40 value (Accumulated Ozone concentration above Threshold 40 ppb) is assumed. The relative yield loss is calculated by using the following equation, and the sensitivity factors given in Table 3:

$y = 99.7 - \alpha \cdot C_{O3}$
with y = relative yield change
α = sensitivity factors
C_{O3} = AOT 40 in ppmh

Table 3: Sensitivity factors for different crop species

Sensitivity	α	Crop species
Slightly sensitive	0.85	rye, oats, rice
Sensitive	1.7	wheat, barley, potato, sunflower
Very sensitive	3.4	tobacco

Acidification of agricultural soils
The amount of lime required to balance acid inputs on agricultural soils across Europe will be assessed. The analysis of liming needs should be restricted to non-calcareous soils. The additional lime requirement is calculated as:

$\Delta L = 50 \cdot A \cdot \Delta D_A$
with ΔL = additional lime requirement in kg/year
A = agricultural area in ha
ΔD_A = annual acid deposition in meq/m^2/year

Fertilisational effects of nitrogen deposition
Nitrogen is an essential plant nutrient, applied by farmers in large quantity to their crops. The deposition of oxidised nitrogen to agricultural soils is thus beneficial (assuming that the dosage of any fertiliser applied by the farmer is not excessive). The reduction in fertiliser requirement is calculated as:

$\Delta F = 14.0067 \cdot A \cdot \Delta D_N$
with ΔF = reduction in fertiliser requirement in kg/year
A = agricultural area in ha
ΔD_N = annual nitrogen deposition in meq/m^2/year

3.1.4 Monetary Valuation

Table 4 summarises the monetary values of health impacts used for valuation of transboundary air pollution. According to Nellthorp et al. (2001) average European values were used for transboundary air pollution costs, except for the source country, where country specific values were used. These were calculated according to the benefit transfer rules given in Nellthorp et al. (2001).

Table 4
Monetary values (factor costs, rounded) for health impacts (€$_{1998}$)

Impact	European average	
Year of life lost (chronic effects)	74 700	€ per YOLL
Year of life lost (acute effects)	128 500	€ per YOLL
Chronic bronchitis	137 600	€ per new case
Cerebrovascular hospital admission	13 900	€ per case
Respiratory hospital admission	3 610	€ per case
Congestive heart failure	2 730	€ per case
Chronic cough in children	200	€ per episode
Restricted activity day	100	€ per day
Asthma attack	69	€ per day
Cough	34	€ per day
Minor restricted activity day	34	€ per day
Symptom day	34	€ per day
Bronchodilator usage	32	€ per day
Lower respiratory symptoms	7	€ per day

Source: Own calculations based on Friedrich and Bickel (2001) and Nellthorp et al. (2001).

3.1.5 Discussion of Uncertainties

In spite of considerable progress made in recent years the quantification and valuation of environmental damage is still linked to significant uncertainty. This is the case for the Impact Pathway Methodology as well as for any other approach. While the basic assumptions underlying the work in ExternE are discussed in detail in (European Commission 1999a), below an indication of the uncertainty of the results is given as well as the sensitivity to some of the key assumptions.

Within ExternE, Rabl and Spadaro (1999) made an attempt to quantify the statistical uncertainty of the damage estimates, taking into account uncertainties resulting from all steps of the impact pathway, i.e. the quantification of emissions, air quality modelling, dose-effect modelling, and valuation. Rabl and Spadaro show that - due to the multiplicative nature of the impact pathway analysis - the distribution of results is likely to be approximately lognormal, thus it is determined by its geometric mean and the geometric standard deviation σ_g. In ExternE, uncertainties are reported by using uncertainty labels, which can be used to make a meaningful distinction between different levels of confidence, but at the same time do not give a false sense of precision, which seems to be unjustified in view of the need to use subjective judgement to compensate the lack of information about sources of uncertainty and probability distributions (Rabl and Spadaro 1999). The uncertainty labels are:

A = high confidence, corresponding to σ_g = 2.5 to 4;
B = medium confidence, corresponding to σ_g = 4 to 6;
C = low confidence, corresponding to σ_g = 6 to 12.

According to ExternE recommendations, the following uncertainty labels are used to characterise the impact categories addressed in this report:

Mortality:	B
Morbidity:	A
Crop losses:	A
Material damage:	B.

Beside the statistical uncertainty indicated by these uncertainty labels, there is however a remaining systematic uncertainty arising from a lack of knowledge, and value choices that influence the results. Some of the most important assumptions and their implications for the results are briefly discussed in the following.

- **Effects of particles on human health**
  The dose-response models used in the analysis are based on results from epidemiological studies which have established a statistical relationship between the mass concentration of particles and various health effects. However, at present it is still not known whether it is the number of particles, their mass concentration or their chemical composition which is the driving force. The uncertainty resulting from this lack of knowledge is difficult to estimate.

- **Effects of nitrate aerosols on health**
  We treat nitrate aerosols as a component of particulate matter, which we know cause damage to human health. However, in contrast to sulphate aerosol (but similar to many other particulate matter compounds) there is no direct epidemiological evidence supporting the harmfulness of nitrate aerosols, which partly are neutral and water soluble.

- **Valuation of mortality**
  While ExternE recommends to use the Value of a Life Year Lost *rather than the* Value of Statistical Life *for the valuation of increased mortality risks from air pollution (see European Commission, (1999a) for a detailed discussion), this approach is still controversially discussed in the literature. The main problem for the* Value of a Life Year Lost *approach is that up to now there is a lack of empirical studies supporting this valuation approach.*

- **Impacts from ozone**
  As the EMEP ozone model, which is the basis for the Source-Receptor Ozone Model (SROM) included in EcoSense does not cover the full EcoSense modelling domain, some of the ozone effects in Eastern Europe are omitted. As effects from ozone are small compared to those from other pollutants, the resulting error is expected to be small compared to the overall uncertainties.

- **Omission of effects**
  The present report is limited to the analysis of impacts that have shown to result in major damage costs in previous studies. Impacts on e.g. change in biodiversity, potential effects of chronic exposure to ozone, cultural monuments, direct and indirect economic effects of change in forest productivity, fishery performance, and so forth, are omitted because they currently cannot be quantified. Furthermore, due to a lack of appropriate models for high altitude emissions, the impacts resulting from aircraft emissions in cruising height cannot be taken into account adequately.

3.1.6 Common Data used in the case studies

Besides the emissions from the vehicles considered in the case studies, a large number of additional information was required for the calculations. This includes data on the receptor distribution, meteorology, and on the background emissions from all sources in all European countries. Such data is available in the computer tool EcoSense's database and is briefly described in the following.

Table 5
Environmental data in the EcoSense database

	Resolution	Source
Receptor distribution		
Population	administrative units, EMEP 50 grid	EUROSTAT REGIO Database, The Global Demography Project
Production of wheat, barley, sugar beat, potato, oats, rye, rice, tobacco, sunflower	administrative units, EMEP 50 grid	EUROSTAT REGIO Database, FAO Statistical Database
Inventory of natural stone, zinc, galvanized steel, mortar, rendering, paint	administrative units, EMEP 50 grid	Extrapolation based on inventories of some European cities
Meteorological data		
Wind speed	EMEP 50 grid	European Monitoring and Evaluation Programme (EMEP)
Wind direction	EMEP 50 grid	European Monitoring and Evaluation Programme (EMEP)
Precipitation	EMEP 50 grid	European Monitoring and Evaluation Programme (EMEP)
Emissions		
SO_2, NO_x, NH_3, NMVOC, particles	administrative units, EMEP 50 grid	CORINAIR 1994/1990, EMEP 1998, TNO particulate matter inventory (Berdowski et al., 1997)
Source: IER.		

Receptor data

- *Population data*

  Population data was taken from the EUROSTAT REGIO database (base year 1996), which provides data on administrative units (NUTS categories). For impact assessment, the receptor data is required in a format compatible with the output of the air quality models. Thus, population data was transferred from the respective administrative units to the 50 x 50 km^2 EMEP grid by using the transfer routine implemented in EcoSense.

  For local scale analysis more detailed data on population density close to the route sections was used.

- *Crop production*

  The following crop species were considered for impact assessment: barley, oats, potato, rice, rye, sunflower seed, tobacco, and wheat. Data on crop production were again taken from the EUROSTAT REGIO database (base year 1996). For impact assessment, crop production data were transferred from the administrative units to the EMEP 50 x 50 km^2 grid.

- *Material inventory*
  The following types of materials are considered for impact assessment: galvanised steel; limestone; mortar; natural stone; paint; rendering; sandstone; and, zinc. As there is no database available that provides a full inventory of materials, the stock at risk was extrapolated in ExternE from detailed studies carried out in several European cities.

Emission data

As the formation of secondary pollutants such as ozone or secondary particles depends heavily on the availability of precursors in the atmosphere, the EcoSense database provides a European wide emission inventory for SO_2, NO_x, NH_3, NMVOC, and particles as an input to air quality modelling. The emission data are disaggregated both sectorally ('Selected Nomenclature for Air Pollution' - SNAP categories) and geographically ('Nomenclature of Territorial Units for Statistics' - NUTS categories). As far as available, EcoSense uses data from the EMEP 1998 emission inventory (Richardson 2000, Vestreng 2000, Vestreng and Støren 2000). Where required, data from the CORINAIR 1994 inventory. (http://www.aeat.co.uk/netcen/corinair/94/) and the CORINAIR 1990 inventory (McInnes 1996) are used. For Russia, national average emission data from the LOTOS inventory (Builtjes 1992) were included. Emission data for fine particles are taken from the European particle emission inventory established by Berdowski et al. (1997).

Meteorological data

The Windrose Trajectory Model requires annual average data on wind speed, wind direction, and precipitation as an input. The EcoSense database provides data from the European Monitoring and Evaluation Programme (EMEP) for the base year 1998. This data was used for regional scale modelling. For local scale modelling more detailed data on wind speed, wind direction and atmospheric stability classes is required, which was provided for each case study separately.

3.2 Global Warming

The method of calculating costs of CO_2 emissions basically consists of multiplying the amount of CO_2 emitted by a cost factor. Due to the global scale of the damage caused, there is no difference how and where the emissions take place.

A European average shadow value of €20 per tonne of CO_2 emitted was used for valuing CO_2 emissions. This value represents a central estimate of the range of values for meeting the Kyoto targets in 2010 in the EU based on estimates by Capros and Mantzos (2000). They report a value of €5 per tonne of CO_2 avoided for reaching the Kyoto targets for the EU, assuming a full trade flexibility scheme involving all regions of the world. For the case that no trading of CO_2 emissions with countries outside the EU is permitted, they calculate a value of €38 per tonne of CO_2 avoided. It is assumed that measures for a reduction in CO2 emissions are taken in a cost effective way. This implies that reduction targets are not set per sector, but that the cheapest measures are implemented, no matter in which sector.

Looking further into the future, more stringent reductions than the Kyoto aims are assumed to be necessary to reach sustainability. Based on a reduction target of 50% in 2030 compared to 1990, INFRAS/IWW (2000) use avoidance costs of €135 per t of CO_2; however one could argue that this reduction target has not yet been accepted.

A valuation based on the damage cost approach, as e.g. presented by ExternE (Friedrich and Bickel 2001), would result in substantially lower costs. Due to the enormous uncertainties involved in the estimation process, such values have to be used very cautiously. The same is true for damage costs due to climate changes caused by high-altitude nitrogen emissions from aircraft flying over Europe, which were estimated in the same study. The marginal damage costs reported (€337 per kg of nitrogen emitted) can therefore only be used as an illustration of the possible order of magnitude of costs that high altitude nitrogen emissions might cause.

3.3 Noise

For the quantification of marginal external noise costs, a bottom-up approach was applied to take into account the site and technology specific characteristics. Especially for noise it is very important to take into account the traffic flow which is responsible for the background noise level in order to calculate the costs of one additional vehicle. This is crucial as the perception of sound follows a logarithmic scale, which means that the higher the background noise level, the lower is the effect of additional noise. Therefore the impact assessment model for noise must be able to represent the environment (receptors, buildings), the vehicle technology (PC, HGV etc.) and the traffic situation (e.g. speed and traffic volume) adequately.

The starting point of the assessment of marginal damages is the micro level, i.e. the traffic flow on a particular route. Two scenarios are calculated: a reference scenario reflecting the present situation with traffic volume, speed distribution, vehicle technologies etc., and a marginal scenario which is based on the reference scenario, but includes one additional vehicle or train of a certain category (e.g. a passenger car). The difference in damage costs of both scenarios represents the marginal external noise costs of that vehicle.

Marginal noise costs due to maritime shipping and inland waterway transport were assumed to be negligible, because emission factors are comparably low and most of the activities occur outside densely populated areas.

3.3.1 Noise exposure modelling

Noise modelling for road noise is based on the German semi-empirical standard model RLS90 (Arbeitsausschuß Immissionschutz an Straßen 1990). The model was enhanced to allow the use of more than two vehicle categories and the respective emission functions, as well as individual vehicle speeds per category following Ullrich (1991).

Noise propagation for rail transport is modelled according to the German rail noise model Schall03 (Bundesbahn 1990). For the calculation of impacts, different noise indices are calculated: $L_{Aeq(7.00\text{-}19.00)}$, $L_{Aeq(19.00\text{-}23.00)}$, $L_{Aeq(23.00\text{-}7.00)}$ and L_{DEN} (composite indicator). Noise levels are calculated as incident sound at the façade of the buildings neglecting reflected sound. The number and type of buildings exposed were analysed in detail using digital images of the sites of the urban case-studies.

In the air transport case study detailed noise modelling results were not available. Instead, marginal noise nuisances were estimated through the share of an aircraft in the total exposure of an average day. This was used to quantify costs due to amenity losses.

3.3.2 Noise impact assessment

Consequences resulting from exposure to transport noise, which affects human life and human health are quantified by the use of exposure-response functions. A large amount of scientific literature on health and psychosocial effects considering a variety of potential effects of transport noise is available. The scientific basis used within UNITE relates to the state of the art summary by De Kluizenaar et al. (2001). In their review work, they report risks due to noise exposure in the living environment. Quantitative functions for relative and absolute risks are proposed for the effect categories presented in Table 6.

Table 6
Categorisation of effects and related impact categories.

Category	Measure given	Impacts
Stress related health effects	RR	Hypertension and ischaemic heart disease
Psychosocial effects	AR	Annoyance
Sleep disturbance	AR	Awakenings and subjective sleep quality
RR = relative risk; AR = absolute risk		

Eight endpoints for concrete health effects were identified for stress related health effects and exposure-response-functions were constructed. The endpoints are defined in a way appropriate for economic valuation. They are listed, together with the ER-functions used, in Table 7. They were applied as well for road as for rail traffic noise.

Table 7
Exposure-response functions for stress-related health effects and sleep disturbance.

Endpoint	Expectancy value [a] (per 1000 adults exposed)
Myocard infarction (MI), fatal, Years of life lost (YOLL)	0.084 L_{DEN} – 5.25
Myocard infarction (non-fatal), days in hospital	0.504 L_{DEN} – 31.5
Myocard infarction (non-fatal), days absent from work	8.960 L_{DEN} - 56
Myocard infarction, expected cases of morbidity	0.028 L_{DEN} – 1.75
Angina pectoris, days in hospital	0.168 L_{DEN} – 10.5
Angina pectoris, days absent from work	0.684 L_{DEN} – 42.75
Angina pectoris, expected no. of morbidity days	0.240 L_{DEN} - 15
Hypertension, days in hospital	0.063 L_{DEN} – 4.5
Sleep disturbance, road traffic	0.62 ($L_{Aeq,23\text{-}07h}$ – 43.2) [b]
Sleep disturbance, rail traffic	0.32 ($L_{Aeq,23\text{-}07h}$ – 40.0) [c]
[a] Threshold is 70 dB(A) L_{DEN} except for [b] 43.2 dB(A) and [c] 40 dB(A); Other assumptions: MI, 7 years of life lost per fatal heart attack in average; base risk of MI: 0.005; survival probability of MI: 0.7; MI, morbidity: 18 days in hospital per MI, 32 days absent from work; Angina pectoris, base risk: 0.0015; days in hosp.:14 / severe episode; 20 days of morbidity per episode; $L_{Aeq,23\text{-}07h}$ as assessed outside at the most exposed façade.	

Sleep disturbance is quantified by calculating the percentage of the exposed population expected to react highly sleep-disturbance annoyed. The functions are derived from noise effect surveys on self-reported sleep disturbance and night time equivalent sound level at the most exposed façade of the dwelling.

Although ER-functions to predict annoyance reactions on the population level were available, they could not be used in this study. For the valuation of annoyance impacts, expressed as the share of the population reacting little annoyed, annoyed and highly annoyed, no corresponding monetary value was available, where the use of the same definition of annoyance levels was assured. Therefore, another method to value amenity losses due to noise was used based on hedonic pricing.

3.3.3 Monetary valuation

Given the physical impacts, appropriate monetary values are needed to derive the costs. According to Metroeconomica (2001), the costs for noise impacts constitute from three components of welfare change:

(a) Resource costs, i.e. medical costs paid by the health service
(b) Opportunity costs, i.e. mainly the costs in terms of productivity losses
(c) Disutility, i.e. other social and economic costs of the individual or others

Components (a) and (b) can be estimated using market prices and are known as "Cost of illness" (COI). The latter must be added to a measure of the individual's loss of welfare (c). This is important because the values for disutility are usually much larger than the cost of illness. They include any restrictions on or reduced enjoyment of desired leisure activities, discomfort or inconvenience (pain, suffering), anxiety about the future, and concern and inconvenience to family members and others.

Especially in the case of environmental noise, hedonic pricing (HP) as an indirect method, used to be the preferred method for quantification of amenity losses due to noise. A large number of such studies has been conducted, giving NSDI values (Noise Sensitivity Depreciation Index – the value of the percentage change in the logarithm of house price arising from a unit increase in noise) ranging from 0.08% to 2.22% for road traffic noise.

Due to the lack of data to value annoyance in the population directly, the willingness-to-pay for avoiding amenity losses was quantified based on hedonic pricing. The value applied for amenity losses is based on HP-findings by Soguel (1994). Soguel reports an NSDI of 0.9 on monthly housing rents, net of charges. It is applied to average net rent per person per year of the respective country. The monetary values applied are presented in Table 8. The same approach was used for valuation of amenity losses due to aircraft noise. The UK average net rent of €3618 per person per year, results in a value of €32.6 per dB(A) per person and year.

As railway noise is perceived as less annoying than road noise, a bonus of 5 dB(A) was applied where no specific ER-function was available for rail transport noise. This is in line with noise regulations in a number of European countries (e.g. Switzerland, France, Denmark, Germany; see e.g. INFRAS/IWW 2000).

Table 8
Monetary values (factor costs, rounded) for impacts due to noise (€$_{1998}$)

Impact	Finland	Germany	Italy
Myocardial infarction (fatal, 7 YOLL)			
Total per case	**535 000**	**564 000**	**528 000**
Myocardial infarction (non-fatal, 8 days in hospital, 24 days at home)			
Medical costs	4 800	4 700	3 700
Absentee costs	2 900	3 500	2 700
WTP	15 400	16 300	12 900
Total per case	**23 100**	**24 500**	**19 400**
Angina pectoris (severe, non-fatal, 5 days in hospital, 15 days at home)			
Medical costs	3 000	2 900	2 300
Absentee costs	1 800	2 200	1 700
WTP	9 700	10 200	8 100
Total per case	**14 500**	**15 300**	**12 100**
Hypertension (hospital treatment, 6 days in hospital, 12 days at home)			
Medical costs	1 900	1 800	1 500
Absentee costs	1 600	2 000	1 500
WTP	600	600	600
Total per case	**4 100**	**4 400**	**3 500**
Medical costs due to sleep disturbance (per year)	200	210	200
WTP (per year)	400	430	400
WTP for avoiding amenity losses (€/dB/person/year)	20	16	8

Source: Own calculations based on Metroeconomica (2001); country-specific valuation based on Nellthorp et al. (2001); for the derivation of the WTP for avoiding amenity losses see text

3.4 Other effects

Air pollution, global warming and noise represent the most important and relevant cost categories for marginal environmental costs. Costs due to “habitat losses and biodiversity” represent the economic assessment of damages the presence of traffic infrastructure and its use is causing to the habitats of rare species, and thus to biodiversity. The costs are mostly related to the separation effects due to the existence of roads, rail tracks, airports and artificial waterways and thus are fixed in the short run. They are not marginal and therefore not relevant for the quantification of marginal costs. The same is true for visual intrusion in urban areas.

Most of the damages to soil and water are expected to be small or not relevant for marginal cost estimation. Modelling of the dispersion processes in soil and water of solid emissions by tyre, brake and wheels (emission of Cd, Zn, Cu) and infrastructure (PAH, heavy metals) abrasion as well as de-icing agents is very challenging and beyond the scope of UNITE. However due to their rather local character damages are expected to be small compared to the exposure to exhaust emissions through the air.

Some effects of airborne exhaust emissions and their impacts on soil and water (acidification of agricultural soils and fertilisation effects of nitrogen deposition) have been included in the analysis. There is evidence that marginal emissions are unlikely to cause relevant impacts to semi-natural vegetation close to roads (Friedrich and Bickel 2001). However, the impairment of ecosystems due to acidification and eutrophication, currently cannot be quantified in monetary terms consistently.

Costs due to nuclear risks are considered in the costs due to electricity production for electric traction of rail transport based on ExternE results for Germany (European Commission 1999b)

4 Overview of Marginal Cost Case Studies

Table 9 gives an overview of the locations and modes covered as well as the character of the case studies performed. Summaries of the case studies are given in the following sections. A detailed description of each case study can be found in the Annex.

Table 9
Overview of marginal cost case studies performed

No.	Country	Location	Modes considered	Character
9A	Finland	Helsinki	Road transport	Urban drive with a passenger car.
9B	Finland	Helsinki – Turku	Road transport	Inter-urban goods transport with a heavy goods vehicle in the south of Finland.
9C	Finland / Estonia	Helsinki – Tallinn	Maritime shipping	Passenger ferry in the baltic sea.
9D	Germany	Stuttgart, Berlin	Road transport, Rail transport	Urban passenger and goods transport in two German cities, one located in the south west, one in the east of Germany.
9E	Germany	Basel – Karlsruhe, Strasburg – Neubrandenburg	Road transport Rail transport	Inter-urban passenger and goods transport on two routes, one located in the south west, one in the north east of Germany.
9F	Germany / UK	Berlin-Tegel – London-Heathrow	Aircraft transport	Flight from Berlin to London, including departure, cruising and arrival.
9G	Italy	Florence	Road transport	Urban transport. Methodological approach differs from the other case studies.
9H	Italy	Milano – Chiasso Bologna – Brennero	Road transport Rail transport	Inter-urban passenger and goods transport on two routes in the northern part of Italy
0A	Netherlands / Germany	Rotterdam - Mannheim	Inland waterway transport	Container transport by barge on the Rhine.

4.1 Road and Rail

9A Urban passenger car for Finland

This case study analysed the marginal environmental costs (direct and indirect emission and noise) of a petrol fuelled passenger car travelling 9 kilometres in the centre of Helsinki. Marginal costs mean the environmental costs caused by an additional vehicle driving on a certain route. The vehicles are analysed with respect to emission characteristics, both a EUROII and a EUROIII emission norm vehicle, with average cold start characteristics considered.

The case study route is located in the most densely populated parts of Helsinki. The starting point of the route is in the southwestern part of central Helsinki (Kamppi). The route traverses the city center, arriving at the northeast suburban belt surrounding (Koskela). The length of the route is 9 km. Maximum speed limits vary between 40 to 60 km/hour.

The total marginal environmental costs (direct emission costs and noise) of a passenger car in the centre of Helsinki are approximately €cent 0.7 per vehicle kilometre with day and evening noise exposure, and approximately €cent 1.0 per vehicle kilometre with night-time noise exposure (see Table 10). Global warming and noise are the dominant marginal cost categories for both EURO norm vehicles.

At wintertime, the additional indirect emission cost of preheating the engine is up to €cent 2.4 per one start, with the assumptions of maximum heating period and electricity supplied by marginal coal fired condensing power. The additional cost of the impacts of the fuel chain allocated to vehicle-km are €cent 0.0845, which means an approximate 10 % rise in the total marginal cost.

Table 10
Marginal environmental costs for passenger car in Helsinki, €cent1998

Impact category	EUROII €cent/vkm	EUROIII €cent/vkm
Direct emissions		
Health	0.115	0.09
Crops and material	0.008	0.005
Global warming	0.354	0.349
Noise	0.22 – 0.53	
Total	**0.698 – 1.008**	**0.664 – 0.974**
Indirect emissions		
Preheating of engine	2.4 €cent/one start	
Fuel chain (average for EUROII and EUROIII)	0.755 €cent/case 0.0845 €cent/vkm	

9B Heavy goods vehicle for Finland

This case study analysed the marginal environmental costs (direct and indirect emission and noise) of a modern heavy goods vehicle traveling from sub-urban Helsinki along an inter-urban highway to sub-urban Turku. Marginal costs mean the environmental costs caused by an additional vehicle driving on a certain route.

The case study route is part of the main transport corridor (highway E18) on the west to east axis of southern Finland, connecting Turku, a port city in southwest Finland, and the capital Helsinki. From Helsinki to the east, the E18 corridor continues to the Vaalimaa border crossing connecting Finnish and Russian road networks. At the Finnish west end point, the corridor is connected to Swedish networks by a ferry link (Turku – Stockholm). Thus, the corridor serves both national and international traffic flows, and it is a part of the TEN-networks, as well as the so-called Nordic Triangle.

The length of the route is 160 km. The duration of the trip is 2 hours at the average speed of 80 km/h. The link is a motorway, with four lanes in the proximity of Helsinki and Turku (approximately 25 km at both ends), and with two lanes on other sections. The terrain is relatively flat, with minor sloping at some segments. The passage is located in an urban/semi-urban environment at both end points. Otherwise, it runs through peripheral areas with low population densities. Some smaller towns and communities are located in the proximity of the route.

The total marginal environmental costs (direct emission costs) of a HGV on route from Helsinki to Turku are approximately €cent 3.7 – 5.0 per vehicle kilometer (see Table 11).

According to the evidence from the route segments in Helsinki, with noise costs taken into consideration in urban sections, the costs are likely to be €cent 5.5 – 6.5 per vehicle kilometer with day and evening noise exposure, and approximately €cent 7.7 – 8.8 per vehicle kilometer with nighttime noise exposure.

Global warming and noise costs at urban segments of the route are the most significant marginal environmental costs. Local health impacts are also of significance. In an inter-urban environment, global warming and health impacts due to local and regional pollutants are the most significant marginal costs.

At wintertime, the additional indirect emission cost of preheating the engine is up to €cent 5 per one start, with the assumptions of maximum heating period and electricity supplied by marginal coal fired condensing power. The additional cost of the impacts of the fuel chain allocated to vehicle-km are €cent 0.438, which means an approximate 5 % rise in the total marginal cost.

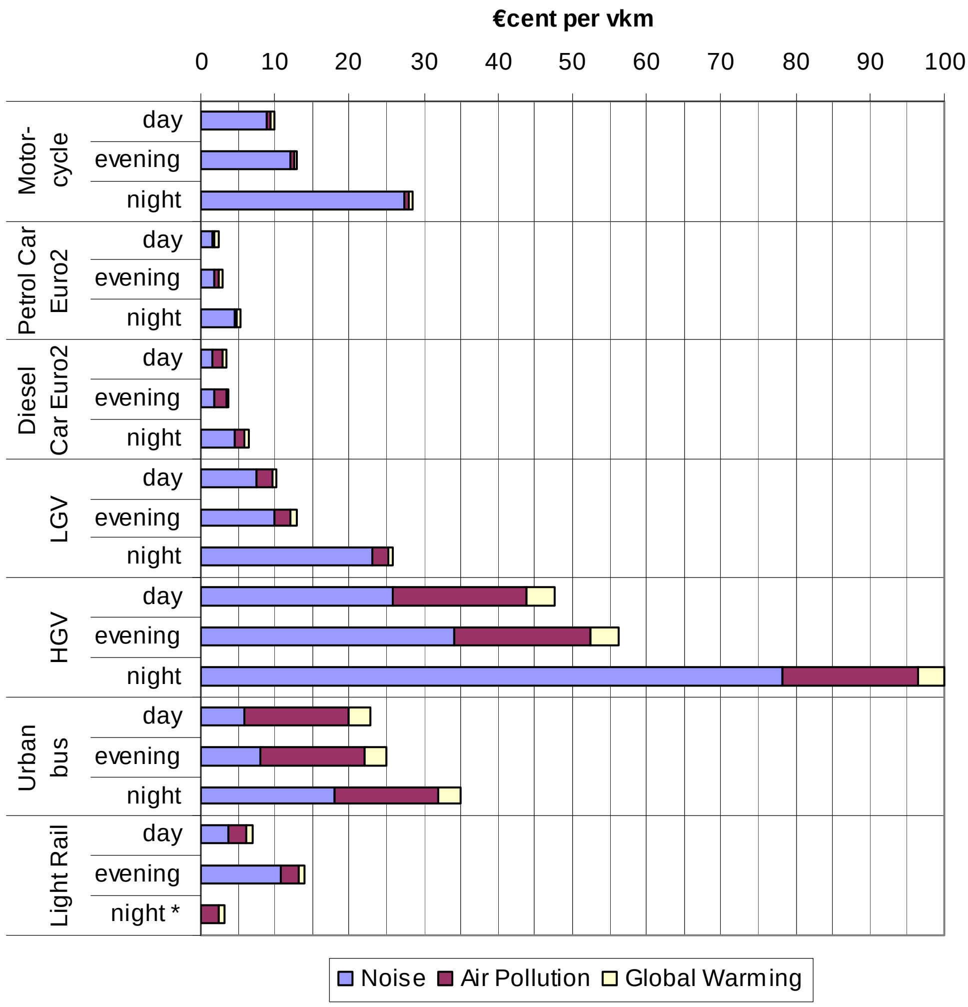
Table 11
Marginal environmental costs for HGV in southern Finland, €cent1998

Impact category	EURO II/ Total weight 42 tonnes €cent/vkm	EURO II/ Total weight 60 tonnes €cent/vkm	EURO III/ Total weight 42 tonnes €cent/vkm	EURO III/ Total weight 60 tonnes €cent/vkm
Direct emissions				
Health	1.98	2.146	1.3	1.42
Crops and material	0.11	0.177	0.108	0.118
Global warming	2.40	2.64	2.46	2.71
Noise (urban environment only)	1.58 – 3.86			
Total	**6.07 – 8.35**	**6.54 – 8.82**	**5.45 – 7.73**	**5.83 – 8.11**
Indirect emissions				
Preheating of engine	4.9 €cent per one start			
Fuel chain (average for EUROII and EUROIII)	70 €cent/case 0.438 €cent/vkm			

9D Urban road case studies Germany

Marginal environmental costs due to road and rail transport were assessed for specific routes in Berlin and Stuttgart. Berlin, Germany's capital and with 3.4 Mio inhabitants the biggest city, is located in the east of Germany.The city of Stuttgart, located in the south west of Germany, is characterised by a very densely populated city centre due to it's location in a steep basin surrounded by a large and heavily populated commuter belt. Costs related to the emission of air pollutants, greenhouse gases and noise proved to be relevant and quantifiable cost categories. Costs of air pollution and global warming were assessed not only for vehicle operation but as well for fuel and electricity production. Road vehicle types covered comprise car, motorcycle, bus, and lorry. Different emission standards were included to analyse their effect on costs. Relevant rail transport options are tram, light rail and underground trains with electric traction.

Figure 1 Sum of marginal costs for Stuttgart "Hohenheimer Straße" (* no night time noise for light rail)

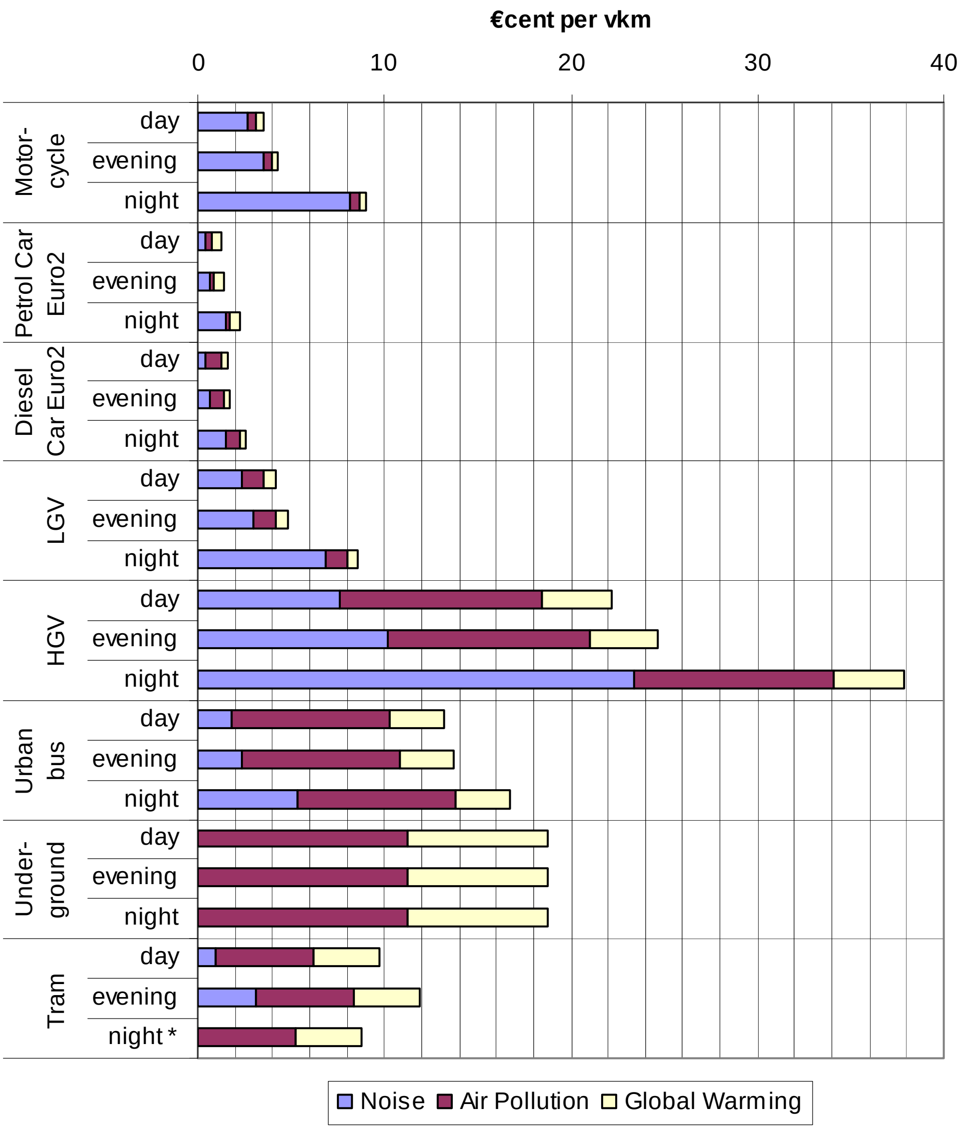


The costs quantified show significant differences. Even though both locations represent urban centres with high population densities the costs per kilometre for the same vehicle vary by up to a factor of three. The single cost categories vary to different degrees.

- Costs due to the emission of greenhouse gases are not location specific, as they are relevant on a global scale. As a consequence all the variation is caused by the emission factor of a vehicle or the underlying electricity production process.
- Besides variations in emission factors the costs due to airborne pollutants are determined by the local meteorology (mainly by the average wind speed) and by the geographical location within Germany, which is important for the formation of secondary pollutants and the number of the population affected by long-range pollutant transport.
- Noise costs are mainly determined by the time of day (higher disturbance effects of noise during the night) as well as the number of vehicles and their speeds, and the resulting background noise. The higher the existing background noise level is, the lower the costs of an additional vehicle.

Figure 1 and Figure 2 present the sums of marginal costs due to air pollution, global warming and noise for the studied routes in Stuttgart and Berlin. The general level of costs is much higher in Stuttgart than it is in Berlin, while the composition of costs is very similar for the different vehicle categories. It has to be noted that it is due to the specific properties of the selected routes that costs due to air pollution and noise are both higher in Stuttgart than in Berlin. While in the case of comparable population densities the noise costs mainly depend on the number of vehicles and their speed, the air pollution costs are determined by local meteorology and geographical location within Germany. On other roads in Berlin with similar population density but lower background noise level the marginal costs would be the same or even higher than for the road considered in Stuttgart.

Figure 2 Sum of marginal costs for Berlin "Frankfurter Allee" (* no night time noise for tram)



The share of the different costs varies with the time of day and the vehicle type. For motorcycles and light goods vehicles marginal costs are dominated by noise costs for all times of day. For urban busses as well as for underground and tram in Berlin the costs due to airborne emissions are higher than noise costs. For the other vehicles at daytime the picture changes from daytime to night time. Of course the proportion of the cost categories changes when other emission standards than EURO2 are considered for the different road vehicles.

For air pollution from road vehicles the differences between Stuttgart and Berlin are mainly caused by the average wind speed, which is much higher in Berlin, and the geographical location within Germany. Both effects lead to higher costs for emissions in Stuttgart for most of the species included in the analysis. For trains with electric traction the differences stem from the share of different fuels in the electricity production process. Here the costs are much lower in Stuttgart due to a very low share of fossil fuels.

9E Inter-urban road case studies Germany

Marginal environmental costs due to road and rail transport were assessed for two inter-urban routes: a drive on the motorway from Basel to Karlsruhe and a drive from Strasburg to Neubrandenburg. Costs of air pollution and global warming were assessed not only for vehicle operation but as well for fuel and electricity production. Road vehicle types covered comprise car, motorcycle, bus, and lorry. Different emission standards were included to analyse their effect on costs. Relevant rail transport options are Intercity passenger train and goods train on the relation Basel – Karlsruhe and local passenger train and local goods train on the route from Strasburg to Neubrandenburg, all with electric traction.

The route from Basel to Karlsruhe is one of the key corridors for European passenger and goods transport from south to north for both road and rail. The train route in the Rhine valley was built to cross the centres of most settlements of importance and so causes considerable problems due to noise exposure. In contrast, the motorway usually passes built-up areas in some distance so that extreme noise exposure is avoided. The route considered for road transport has a length of 210 km, that for rail transport is 196 km long.

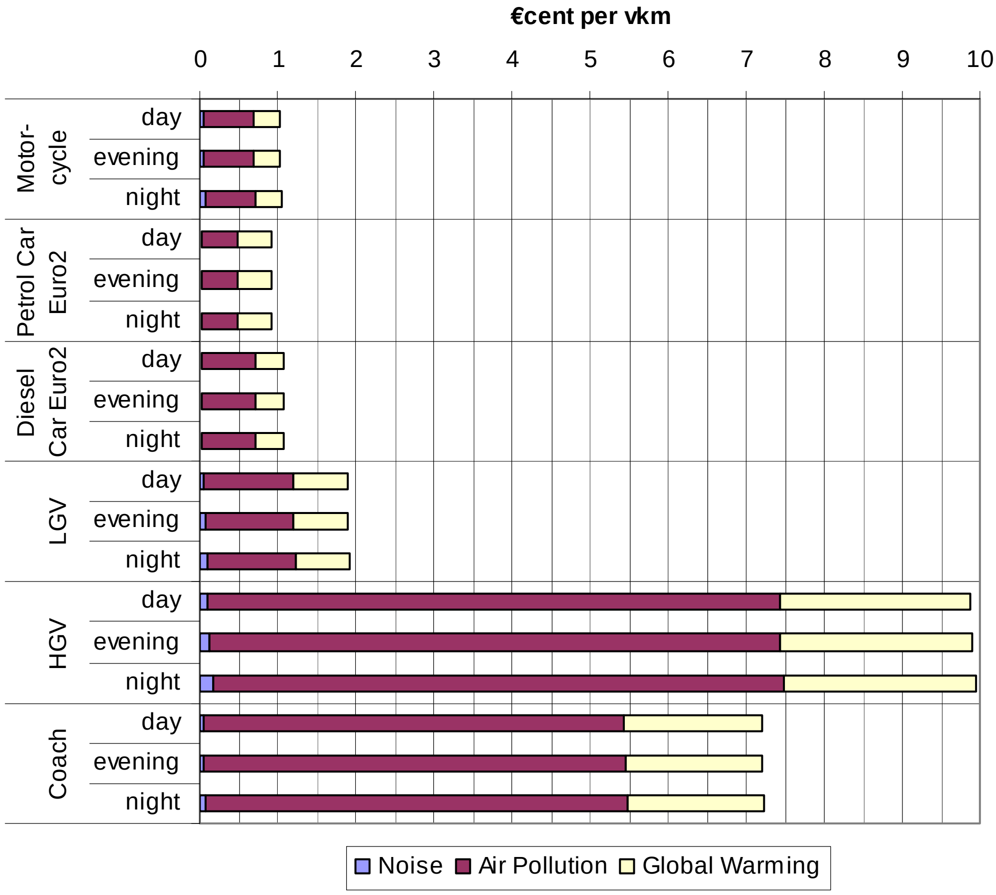
Neubrandenburg is the third largest city in the federal state of Mecklenburg-Vorpommern, located in the north east of Germany, and counts 77 000 inhabitants. Strasburg is a small town, located 39 km from Neubrandenburg and 50 km from the border to Poland. As Neubrandenburg represents the economic centre to the rural surrounding, the route is frequently used by commuters and goods transport. The road crosses a number of smaller towns and villages between Neubrandenburg and Strasburg. In contrast, he railway line does not cross built-up areas, apart from start and end. This has major implications on noise costs, as only few people are affected by railway noise, compared to the route from Basel to Karlsruhe, where the line crosses many built-up areas.

Following conclusions can be drawn from the results:

- The geographical location of roads outside urban areas plays an important role for the costs, because local effects are of minor importance. Differences in the costs due to long-range effects of air pollution may be large, depending on the formation of secondary pollutants and population affected by long-range pollutant transport.
- Marginal noise costs are relevant only on roads within built-up areas, depending on the vehicle type. In the example of the drive from Strasburg to Neubrandenburg noise costs were relevant mainly for motorcycles, LGV and HGV. Noise costs vary considerably between different times of the day, reflecting the varying disturbance effects and variations in background noise levels.
- For trains with electric traction the marginal costs quantified may vary heavily depending on the fuel mix from which the electricity is produced – the lower the share of fossil fuels, the lower the resulting costs.

Figure 3 presents the total of marginal costs due to air pollution, global warming and noise for the motorway drive from Basel to Karlsruhe. The share of noise costs is extremely low, indicating that local effects are comparably small for this relation. This is the case for air pollution costs as well, which are dominated by long range impacts.

Figure 3 Sum of marginal costs for motorway drive from Basel to Karlsruhe



Compared to that, noise costs have a higher share for roads within built-up areas on the route from Strasburg to Neubrandenburg as shown in Figure 4. But with the exception of motorcycle and LGV total costs are still dominated by airborne emissions. As a consequence of the limited share of noise costs the variation of costs with time of day are comparably small. Of course the proportion of the cost categories changes when other emission standards than EURO2 are considered for the different road vehicles.

Figure 5 shows the total costs for the trains considered. The proportion of costs due to air pollution and global warming are the same for both locations analysed, because the electricity mix of the national rail operator was used for both routes. The variations in these cost categories stem from the different electricity consumption of the train types. The costs due to airborne emissions could change considerably, if a different fuel mix for electricity production would be assumed – the lower the share of fossil fuels, the lower the resulting costs. The pattern of noise cost variation from day to night is different from the normal situation. For Basel-Karlsruhe costs decrease from day to evening due to an increasing number of (goods) trains. Then the costs increase again from evening to night due to higher disturbance effects and a decreasing number of trains. However, marginal costs are still lower than at daytime due to the higher number of trains per hour and a different train mix (almost only goods trains).

Figure 4 Sum of marginal costs for road transport from Strasburg to Neubrandenburg (roads within built-up areas)

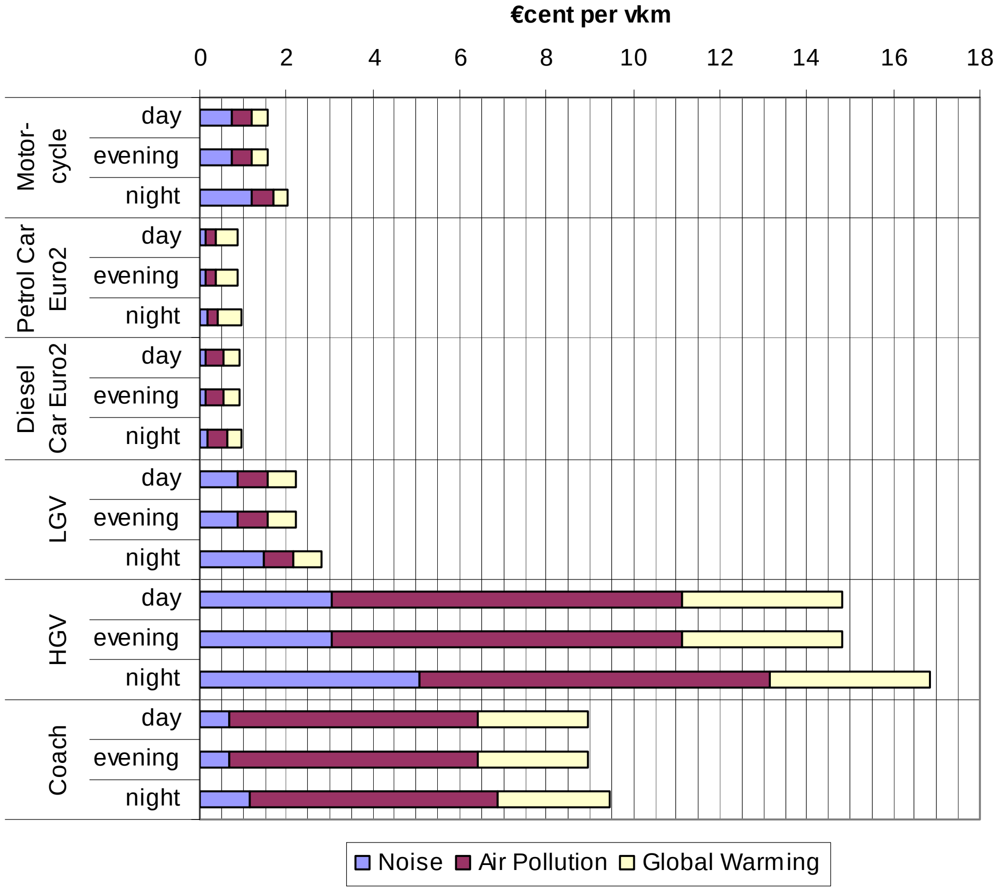
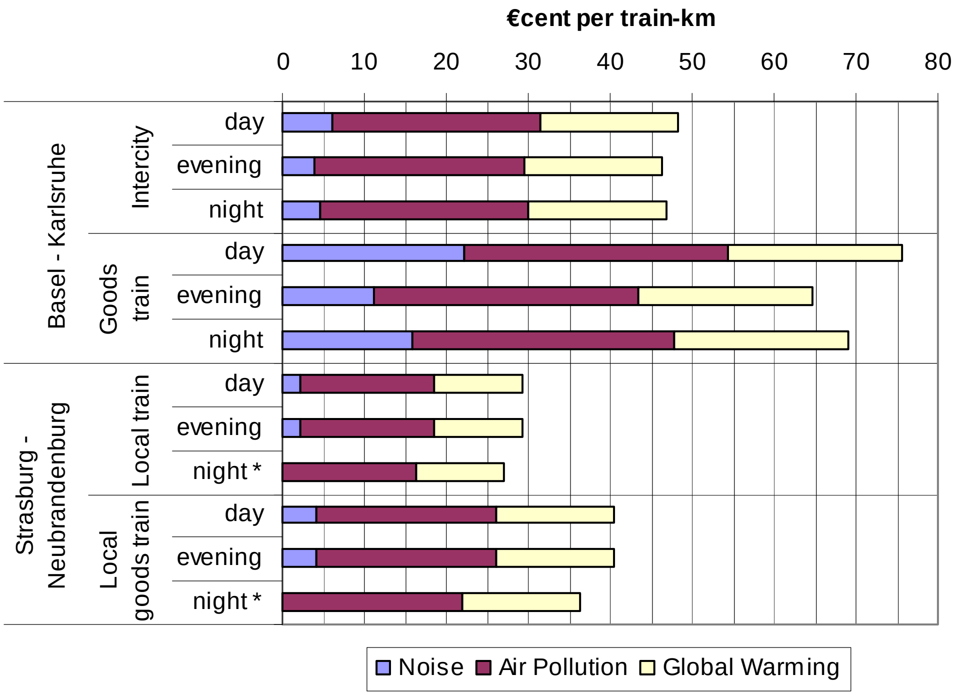


Figure 5 Sum of marginal costs for inter-urban rail transport (* no trains during night time)



Whereas the track from Basel to Karlsruhe crosses the centres of many towns, the track considered from Strasburg to Neubrandenburg only crosses built-up areas in the start and in the end. For this reason the costs per km are lower compared to Basel – Karlsruhe even though the number of trains is much lower. In addition, the local passenger train and local goods train cause less noise compared to the Intercity and long distance goods train due to lower speeds and shorter trains.

9G Urban road and rail studies: the case of Florence

The aim of this study was to provide a methodological framework for the evaluation of environmental marginal external costs in the city of Florence. The city of Florence, about 380,000 inhabitants in 1999, over the past 10 years has followed a path similar to several western cities belonging to the industrialised world: population falling, increase of urban sprawl and further modification of demographic composition, in particular through the progressive ageing of population and the percentage growth of households with one membership.

In this study, the environmental marginal costs of air pollution, and noises are analysed, with the aim to assess their impacts on human health. The marginal impacts of air pollutants on natural environment, i.e. agriculture, water availability, are considered negligible in the Florence urban context. The marginal impacts on building materials are not estimated due to the paucity of data and local case studies. Global warming impacts are estimated with limitations, due to the unavailability of specific case studies . In the context of the city of Florence, road transport is the transport mode to be dealt with, since rail transport is negligible. In fact, the provision for the city of Florence of tram network and rail services is only at the initial stage.

A methodological departure from the traditional implementation of IPA has been used in this case study. With reference to air pollution, it basically consists in avoiding to run dispersion models through the use of statistical analysis for validating the relationships between concentration data and traffic flows.

Concerning the noise emissions, a partial overcoming of the difficulties of a full-fledged implementation of the impact pathway approach, has been made possible by the Florence municipality that has conducted between 1987 and 1996, in collaboration with the Regional Agency for Environmental Protection, an extensive urban campaign for noise measurements. Such a campaign has allowed to:

1. provide a classification of urban roads through a set of factors associated with the measured noise emissions, i.e. main street with high traffic flows, local street with bus transit, etc

2. avoid to run dispersion models by means of regression analysis between traffic flows/vehicle type and measured Leq(A).

The calculation of marginal costs of air pollution for the Florence urban area leads to the following results.

Table 12
Marginal external cost due to PM10 emissions by vehicle type

	€cent / vkm
Diesel Car (conventional not cat.)	3.04
Diesel Car (Euro 1)	1.17
Diesel Car (Euro 2)	0.87

Noise costs were assessed through regression analysis based on measured noise levels at a particular road with a specific fleet composition. The values shown in the following table give the yearly willingness-to-pay per inhabitant affected to reduce the equivalent noise level by one dB(A). The costs vary with the vehicle flow – decreasing costs with increasing vehicle flow –, as the effect of additional noise decreases with increasing noise levels.

Table 13
Marginal external cost of noise by traffic flow

Vehicle flow [vehicle / h]	Marginal cost [€ / (inhabitant year)]
891	1.7
708	2.0
562	2.4
447	2.8
355	3.3
282	3.9

As these results vary heavily with traffic conditions, vehicle speed, and fleet composition a transfer to other streets is heavily restricted. Furthermore the values are not comparable to that of the other case studies due to a different approach of estimating noise exposure and the use of a different monetary value.

9H Inter-urban road and rail case studies Italy

The Italian rail-road inter-urban environmental marginal external costs case studies focused on two corridors: Milano-Chiasso and Bologna-Brennero, with significant shares in terms of overall movements of passengers and freight involved. In 1998, the routes Milano-Chiasso and Bologna-Brennero showed the circulating average daily number of vehicles higher than the average national values, both for freight and passenger cars traffic along the Milano-Chiasso, and for freight traffic along the Bologna-Brennero.

Concerning rail traffic, freight movements (train km) along the routes in 1998 showed clear reversal trends with reference to the national average: +3.2% for the Milano-Chiasso and +1.0% for the Bologna-Brennero, compared to the decreasing national average of –4.9%. Conversely, passenger movements showed two different trends: increasing for the Milano-Chiasso (+1.6%) and decreasing for the Bologna-Brennero (-1.6%), which followed the national average (–0.8%).

Road environmental marginal external costs (air pollution and global warming) in 1998 per vehicle kilometre showed on average higher values on the route crossing the most densely populated areas, i.e. along the Milano-Chiasso. The analysis by vehicle types indicates that HGV, coaches and LGV exhibited higher external costs than gasoline and diesel cars.

Table 14
Road environmental marginal external costs (air pollution and global warming) €cent/vkm, 1998

VEHICLE TYPE	MILANO-CHIASSO	BOLOGNA-BRENNERO
Car Diesel	2.271	1.089
Car Gasoline	0.606	0.551
Coach	6.207	5.241
LGV	3.022	1.582
HGV	8.878	7.230

Rail environmental marginal external costs per vehicle kilometre (air pollution and global warming), showed the high incidence of the air pollution external costs arising from power plant emissions (including fuel extraction, transport and refinery).

Table 15
Rail environmental marginal external costs (€cent/train-km) 1998

	AIR POLLUTION	GLOBAL WARMING	TOTAL IMPACT
FREIGHT TRAIN			
MILANO-CHIASSO	14.758	0.149	14.907
BOLOGNA-BRENNERO	18.334	0.185	18.520
PASSENGER TRAIN			
High Speed Train	41.756	0.731	38.691
Intercity	31.650	0.554	29.327
Local train	23.261	0.407	21.553

Concerning noise marginal external costs, striking differences on the two routes can be observed. Road marginal external costs on the Milano-Chiasso outnumbered of a factor ten the external costs on the Bologna-Brennero, due to the different density of exposed population.

Table 16
Road marginal noise costs (€cent/vkm) 1998

	day	evening	night
MILANO-CHIASSO			
Passenger car	0.01	0.02	0.04
LGV	0.03	0.05	0.12
HGV	0.09	0.13	0.35
BOLOGNA-BRENNERO			
Passenger car	0.001	0.001	0.002
LGV	0.001	0.002	0.005
HGV	0.006	0.008	0.021

Rail marginal external costs confirmed the differences between the two routes. In order to explain the differences, a possible underestimation of the exposure level on the route Bologna-Brennero should be also taken into account. In fact, due to the long distance along the route, i.e. about 362 km, it has not been possible to provide a sufficient level of detailed information along the entire routes length, in particular for specific segments of the northern section of the route.

Table 17
Rail marginal noise costs (€cent/train km) 1998

	day	**evening**	**night**
MILANO-CHIASSO			
Goods train	13.2	14.1	10.0
Inter city	1.4	1.7	4.1
Local train	1.2	1.4	1.2
BOLOGNA-BRENNERO			
Goods train	0.3	0.37	0.59
Inter city	0.04		0.03
Local train	0.19	0.20	0.25

4.2 Maritime shipping

9C Nordic maritime shipping

This case study analysed the marginal environmental costs for atmospheric emissions of a typical passenger ferry travelling from Helsinki (Finland) to Tallinn (Estonia) in the Finnish Gulf. Marginal costs mean the environmental costs caused by an additional vessel on a certain route or visiting a port. Marginal costs are assessed both for the route, and berth periods at ports.

The results are presented at the Finnish price level including impacts in Tallinn and Estonia. Therefore, for assessing the costs at Estonian price level, a purchasing power adjustment must be made.

In Table 18, the marginal emission costs at open sea are presented. The marginal emission cost of a vessel kilometre at open sea is €18, which means that the total marginal cost of emission impacts for the trip (90 km) from Helsinki to Tallinn is €1 622. Regional health impacts clearly dominate the costs, but global warming is also significant.

Table 18
Marginal emission costs for the vessel at open sea by damage category

Impact category	€cent/case	€cent/km
Local impacts		
Morbidity	1 470	16.3
Mortality	5 506	61.2
Regional impacts		
Crops & material	7 999	88.9
Morbidity	31 142	346.0
Mortality	62 514	694.6
Global warming	53 412	593.5
Fuel chain	119	1.38
Total	**162 202**	**1 803**

The emission costs of an average berth period (8.5 hours) in Helsinki and Tallinn are presented in Table 19. The marginal emission cost of the case vessel at berth is €0.3 per hour, which means that the total marginal costs of emission impacts for each berth period (8.5 hours) in Helsinki and Tallinn are €2.5 – €2.6. Again regional health impacts cause the highest environmental cost, but the costs of global warming are also of significance.

Table 19
Marginal emission costs for the vessel at berth in Helsinki and Tallinn

Impact category	€cent/visit at port Helsinki (8.5 hours)	€cent/visit at port Tallinn (8.5 hours)
Local impacts		
Morbidity	7.4	2.3
Mortality	2.1	0.7
Regional impacts		
Crops & material	11.6	11.6
Morbidity	46.6	46.6
Mortality	91.2	91.2
Global warming	81.2	81.2
Fuel chain	15	15
Total	**255**	**249**

Local impacts in Tallinn are lower compared to Helsinki due to lower population density and the fact that dominant wind directions carry pollutants away from the town in Tallinn, whereas the opposite holds for Helsinki.

For the whole trip, the marginal emission costs for the open sea part are much larger than the marginal emission costs of the berth periods altogether. At ports the vessel uses reserve engines and low sulfur fuel, whereas at sea the main engines are run on fuel with high sulfur content.

For a round trip (two route periods + one berth period), the marginal environmental costs due to emissions are approximately €3 247. Health impacts due to regional impact of emissions cause the highest environmental costs, but the costs of global warming are also of significance.

In the case of scheduled passenger ferry traffic discharges of wastes or contaminated liquids to sea are not considered a problem. Because of well-established waste management practices of shipping companies, waste and bilge waters are disposed of at ports.

4.3 Inland waterway transport

0A Container transport on the Rhine – Marginal cost case study

Besides environmental costs the case study covers as well infrastructure and accident costs. Within this document we refer only to the environmental costs quantified.

Inland waterway transport has a reputation for environmentally friendly transport as it has very little impact on landscape, pollution of water is small and air emission per tonne kilometre is low compared to road transport given the current applied technologies.

The most important type of pollution by barges is air pollution and global warming related, and is fully dependent on fuel use. Barge owners are of course very much motivated to achieve the highest possible utilisation rate, decreasing the average consumption of fuel per loaded ton. The higher the ships utilisation rate, the more effective the use of fuel and the less the environmental pollution per tonne kilometre.

Since the majority of the Rhine river stretch flows through rural areas, most of the pollution will occur in those areas. However, the wind may cause pollutants to end up in city areas, as will the rivers' water current transport waste deposits through city areas, or wash the garbage ashore in city areas.

The study area comprises the areas along the lower and middle Rhine, from the seaport of Rotterdam to the inland port of Mannheim. The justification for this stretch of river lies in the scale of inland shipping operations. On the River Rhine, the largest vessels can proceed up to Mannheim. Beyond Mannheim, navigation constraints will increase and downgrade both ship size and total transported volumes.

Noise impacts of barge shipping are minimal. The actual noise emission is low and very little habitation is located close to inland waterways. The main noise emissions result from container handling at terminals and for this it may be argued that this is not specifically related to inland waterways, but to transhipment activity. External costs of noise for this inland waterways case study are therefore considered to be negligible.

Marginal costs due to impacts on water quality and soil are existent but small. There is always a risk of accidents causing unintended water pollution (see also marginal accident costs). During 1998, in accidents within the Netherlands only one case of oil pollution was recorded within the port of Rotterdam. In five other cases the environmental damage is unknown. Data on environmental pollution within German waters is not available. It is estimated that the total number of environmental accidents per year is 10 with an average cost of € 10 000. In case an oil slick is resulting from a collision between two ships, the oil will pose a direct threat to the nearby embankments. Intended pollution of the water also happens, by throwing garbage into the river. Although this is prohibited and stiff penalties are imposed on this behaviour, quite some waste is dumped overboard on an annual basis. Law enforcing bodies are patrolling the rivers on a daily basis, in order to prevent pollution or apprehend the perpetrators. Also the garbage put overboard is washed upon the embankments, causing an array of tin cans, broken plastic buckets, mooring ropes, etc. This ecological pollution disturb the local wildlife or even cause some deaths of animals when swallowing waste.

Marginal costs due to air pollution and global warming are quantified for the *use* of a vehicle (in this case container barges), costs due to vehicle *maintenance*, *building* and *infrastructure provision* are expected to be very small.

The marginal increase in damage factor per unit of pollutant was calculated for three specific Rhine segments (see Table 20). The Rotterdam-Nijmegen segment of Rhine causes much higher damages compared to the other two segments. This is the result of the high population density near Rotterdam. The other Rhine segments have quite similar results. To calculate the marginal costs the pollutant emissions on the different route segments were multiplied by the damage factors and related to a TEU – a twenty feet equivalent unit). With an average capacity of 200 TEUs per ship and a utilisation rate of 80%, the marginal costs can be estimated at 1.8 €cent per TEU-km (or 360 €cent per vessel-km) upstream and 1.2 €cent per TEU-km (or 240 €cent per vessel-km) downstream, averaged for the three route segments.

Table 20
Marginal costs expressed per unit of pollutant emitted

Emitted pollutants	NO_X	$PM_{2.5}$	SO_2	CO	Benzene	NMVOC
Damaging pollutants	Nitrate+Ozone	$PM_{2.5}$	SO_2+Sulphates	CO	Benzene	Ozone
	€/kg	€/kg	€/kg	€/kg	€/kg	€/kg
Rotterdam-Nijmegen	3,1	145,6	9,1	0,0012	0,8	1,5
Nijmegen-Duisburg	2,5	69,2	6,5	0,0006	0,4	1,5
Duisburg-Mannheim	3,9	68,7	5,4	0,0006	0,4	1,8
Source: IER						

4.3 Air transport

9F Air Transport Case Study

Marginal environmental costs due to a flight from Berlin to London were quantified. Berlin Tegel and London Heathrow are important international airports, linking the capitals of Germany and the United Kingdom. Both airports are located within densely populated agglomerations, which is important for air pollution and noise costs, which are closely related to the population density in the vicinity of the emission source. The Boeing 737-400 considered is a medium range aircraft, commonly used by many airlines on domestic and European services.

Table 21 presents the marginal costs due to airborne emissions per LTO-cycle at Berlin Tegel and London Heathrow, as well as quantifiable costs due to a flight from Berlin to London. Total costs of aircraft movements at airports are dominated by direct emissions, costs due to fuel production emissions play only a minor role. For cruising only costs from CO_2 emissions and fuel production could be considered, causing a potentially considerable underestimation of costs.

Table 21
Marginal costs due to airborne emissions of a Boeing 737-400 in EUR

		air pollution			global warming			total
		direct emissions	fuel production	total	direct emissions	fuel production	total	
Berlin Tegel	LTO-cycle	42.18	8.56	50.74	44.74	5.68	50.42	101.16
	Departure	28.29	4.64	32.93	24.26	3.08	27.35	60.28
London Heathrow	LTO-cycle	37.86	6.01	43.87	48.57	6.17	54.74	98.62
	Arrival	13.21	2.77	15.98	22.35	2.84	25.19	41.17
Flight Berlin -	Cruise	[1]	33.47	33.47	175.00 [2]	22.22	197.22	230.70
London	Total [3]	41.51	40.88	82.39	221.61	28.14	249.75	332.15

[1] Costs due to direct air pollution emissions not included; [2] Possible order of magnitude for global warming effects due to high altitude nitrogen emissions: ca. €3000; [3] Consisting of departure at Tegel, cruise, and arrival at Heathrow.

Marginal noise costs for arrival and departure of a Boeing 737-400 at Heathrow amount to almost €59. Together with the costs due to air pollution and global warming the costs for a flight from Berlin to London can be estimated to €391. This assumes that the costs of a starting aircraft at Berlin Tegel are about the same as at Heathrow. As both airports are located within a densely populated area this assumption is justifiable. With the distance between Berlin and London of about 930 km, the costs can be expressed as €cents 42 per aircraft kilometre.

The shares of the cost categories in the LTO activities of the flight are about the same: air pollution €49, global warming €52.50 and noise €59, adding up to €160.50. The costs of cruising of €230.70 are dominated by CO_2 emissions, costs due to fuel production emissions are only of minor importance.

5 Generalisation of Results

In this chapter the variability of the case study results is analysed with respect to the transferability and generalisation of marginal costs. The impact pathway approach for airborne pollutants and noise is in general transferable to other modes and locations, including exposure-response functions and to a large extent monetary values. Dispersion models may have to be modified to reflect mode-specific characteristics. This is the case in particular for noise propagation modelling for road, for rail and for aircraft transport. The approach of multiplying the amount of greenhouse gas emissions with a specific cost factor is transferable per se as no other parameters have to be taken into account.

5.1 Costs due to Air pollution

The costs quantified can be generalised for vehicles or vessels on the same relation as studied, adjusting for differences in the specific emissions. For example the costs for a petrol car complying with EURO1 can be derived from the existing results.

When it comes to transferability we first have to compare the results of the case studies performed in different locations and check whether there are parameters that hamper a transfer. This will be done in the following sections.

5.1.1 Direct exhaust emissions

Road transport

Table 22 gives an overview of costs due to air pollution from road vehicle exhaust emissions quantified in the case studies for three vehicle technologies: petrol and diesel passenger cars and heavy goods vehicles, all complying with the EURO2 emission standard. The costs per vehicle kilometre vary for different locations and vehicle/fuel types. In the following, the reasons for these variations are explored to identify relevant parameters for generalisation of results.

The main parameters determining the costs due to direct vehicle emissions (representing “line emission sources”) are:

- Emission factors, which differ by fuel (e.g. petrol – diesel), vehicle type (e.g. heavy diesel vehicles – diesel cars), emission standard (e.g. EURO2 – EURO4), and driving pattern (speed, acceleration processes).
- The local environment close to the road (receptor density, meteorology, above all average wind speed).
- The geographical location (determining the number of receptors affected by long-range pollutant dispersion and formation of secondary pollutants).

Emission factors for specific vehicle technologies (e.g. passenger car complying with EURO2 standard) can be transferred to other countries/locations. A correction for the actual emission of vehicles in different countries may be desirable, as the typical age or maintenance behaviour may vary between countries. In practise it is very difficult to obtain empirical data on such differences.

Emission factors for vehicle fleets (e.g. of a country, on a certain road) are not generally transferable, because the fleet composition usually varies.

Table 22
Overview of damage costs due to air pollution from road vehicle exhaust emissions in €cent / vkm

		Car Petrol EURO2	Car Diesel EURO2	HGV Diesel EURO2
urban case studies	Helsinki	0.12	n.a.	n.a.
	Stuttgart	0.25	1.45	17.52
	Berlin	0.15	0.73	10.19
	Florence [a)]	*0.01* [a)]	*0.26* [a)]	*4.69* [a)]
inter-urban case studies	Helsinki – Turku	n.a.	n.a.	2.09
	Basel – Karlsruhe	0.37	0.63	6.91
	Strasburg – Neubrandenburg (outside built-up areas)	0.12	0.26	3.89
	Strasburg – Neubrandenburg (within built-up areas)	0.11	0.38	7.46
	Milano – Chiasso	0.25 [b)]	1.91 [b)]	6.72 [b)]
	Bologna – Brennero	0.20 [b)]	0.73 [b)]	5.07 [b)]

[a)] restricted comparability to other results, because estimate is based on a different methodological approach; only human health impacts on the local scale due to CO, Benzene and PM_{10} considered (NO_x, SO_2, Ozone, NMVOC not included);

[b)] emission standard not specified

To eliminate the effect of different emission factors used in the case studies and to allow the analysis of differences between locations, damage costs are related to a unit of emission of $PM_{2.5}$. $PM_{2.5}$ is one of the key pollutants as regards the share in the costs quantified, in particular on the local scale (i.e. up to ca. 25 kilometres from the emission source). Table 23 shows damage costs on the local and on the regional scale (covering long-range pollutant transport all over geographical Europe). Costs on both scales add up to the total costs caused by a unit of pollutant emitted in the respective area. In urban areas local scale effects dominate the costs due to the high receptor density. For this reason, it was one of the hypotheses that costs could be generalised based on the number of population affected in the urban area.

But a closer look at the case study results shows, that there must be another relevant parameter. In Helsinki and Stuttgart the number of persons affected is almost the same, in absolute and in relative (inhabitants per km^2) terms. Even though, in Stuttgart costs on the local scale are about a factor of two higher than in Helsinki. Furthermore, in Berlin where many more persons are affected than in Helsinki and Stuttgart, the local scale costs per unit of $PM_{2.5}$ emitted are lower than in the other two cities. These differences in the local scale costs are determined by the local meteorology. In Stuttgart, the yearly average wind speed is much lower than for example in Berlin, so that pollutants stay longer in the densely populated area before being dispersed, resulting in higher damages. As a consequence, local meteorology has to be taken into account when generalising cost estimates for emissions in urban areas.

Table 23
Comparison of damage costs due to $PM_{2.5}$ emissions of urban case studies

Location	Population density (inh. / km^2)	Costs due to damages on the local scale in € / tonne of $PM_{2.5}$	Costs due to damages on the regional scale in € / tonne of $PM_{2.5}$
Helsinki	2800	95000	2800
Stuttgart	2800	200000	26800
Berlin	3800	90000	17500
Florence [a)]	*4100*	*50000* [a), b)]	n.a.

[a)] restricted comparability to other results, because estimate is based on a different methodological approach;
[b)] € / t PM_{10}; n.a. = not available

Regional scale costs depend on the geographical location within Europe, determining the number of receptors affected and when looking at other species than $PM_{2.5}$ the formation of secondary pollutants (above all ozone as well as nitrate and sulphate aerosols) via air chemistry. Such differences may be considerable as illustrated in Table 23. Due to the prevailing west winds, emissions in the south of Finland are transported towards the baltic sea and to sparsely populated areas in Russia, leading to very low costs per unit of pollutant emitted. Even within Germany there are considerable differences between emissions in the (North-) East and the Southwest, because the areas affected by emissions in the Southeast are more densely populated. Besides, the formation of sulphates and nitrates is influenced by the background concentrations of the reactive species involved and the ratio of SO2 and NOx emitted.

Whereas for urban areas the share of local scale costs is high and therefore the geographical location is of minor importance, the regional scale damages are very important for locations outside urban areas.

As a consequence the generalisation of results has to take into account at least the parameters
a) for emissions in urban areas: the local population density and the meteorology
b) for emissions in extra-urban areas: the geographical location and the character of the route (passing built-up areas or crossing built-up areas)

The values per vehicle kilometre are valid for the engine specifications given in the respective case studies.

Other modes

The relationships between pollutant emission and associated costs are in principle the same as for road transport. Emissions from diesel locomotives, maritime and inland waterway vessels can be treated like emissions from road vehicles, taking into account the character of the route. For these modes however the main part of emissions will occur in extra-urban areas. Aircraft emissions are a special case, because most of the emissions take place in high altitudes. Assessment of the resulting impacts is still to be improved, because modelling of dispersion and chemical conversion is not as advanced as for low level emissions. On the other hand impacts due to low level emissions at airports occurring during arrival, ground activities and departure can be assessed with the existing models.

5.1.2 Indirect emissions from electricity production and fuel production

Costs due to air pollution from electricity production depend on the pollutant emissions from the power plants in which the electricity is produced. Based on the emissions for producing a unit of electrical energy (e.g. 1 kWh), damage costs per kWh can be calculated. If electricity is generated from different fuels, the resulting costs per kWh can be combined according to the share of the different fuels in the electricity production. Table 24 illustrates the fuel mixes considered in the German case studies. The resulting costs per kWh of electricity produced vary considerably, mainly depending on the share of fossil fuels used: by a factor of three between Stuttgart urban transport and the German Rail and a factor of four between Stuttgart and Berlin.

Table 24
Share of fuels in the electricity production of rail transport operators

	Berlin	Stuttgart	Railway Germany
Coal	74.0% [a]	1.9% [b]	34.4% [c]
Nuclear		89.6% [b]	22.1% [c]
Oil/Natural gas	26.0% [a]	1.9% [b]	13.2% [c]
Hydro		2.1% [b]	10.1% [c]
Electricity from public grid and other		4.5% [b]	20.2% [c]
Total	100.0%	100.0%	100.0%

[a] Bewag (2002); [b] Schmid et al. (2001); [c] Deutsche Bahn (1998)

Generalisation of costs per kWh of electricity requires information on the geographical location of the power plant and the respective emission factors. If the electricity is produced from different fuels, the costs per kWh can be calculated according to the share of the different fuels in the electricity production.

Costs due to indirect emissions from fuel production gain in importance for vehicles complying with stricter emission standards. For petrol cars complying with EUROII standard or higher the costs from fuel production may reach the same order of magnitude as the costs from exhaust emissions. For diesel vehicles this is usually not the case, because the costs due to exhaust emissions are generally higher, and the costs due to diesel production are lower than for producing petrol.

It was not within the scope of the case studies to take into account where the fuel burnt specifically was produced. Therefore average costs for the emissions due to fuel production within a country were quantified, taking into account emission factors for fuel production in refineries and the specific costs due to air pollution from the refineries within a country. These values only vary with the fuel consumption of a vehicle and therefore can be generalised – based on the fuel consumption – for the country for which they were calculated.

5.1.3 Summary of generalisation aspects

A comparison of the case study results clearly suggests that a direct transfer of costs due to air pollution cannot be recommended. Some general rules could be derived, but an operational formula for transfer requires a broader statistical basis of case studies. A generalisation methodology for air pollution costs should account for

- the local scale conditions (population density and local meteorology), and

- the regional scale costs per tonne of pollutant emitted in a certain area (e.g. on NUTS1 level)

Table 25 presents a list of the main generalisation aspects.

Table 25
List of generalisation aspects for marginal costs due to air pollution

	aspect to be generalised	important mainly for	basic requirements for generalisation
Metho-dology	overall methodology: Impact Pathway Approach		can be generalised
Inputs	inputs to dispersion models		generalisation not recommended
	exposure-response functions		can be generalised
	monetary values for health impacts		country-specific adjustment/values for local scale impacts
	exhaust emission factors for specific vehicle technologies		same emission standard; same driving characteristics/speeds
	exhaust emission factors for vehicle fleets		generalisation not recommended
	emission factors for the production and transport of fuel		refinery processes and fuel distribution are comparable
Output values	regional scale unit costs per tonne of pollutant	extra-urban	pollutant is emitted in the same geographical area (e.g. administrative unit on NUTS1 level)
	local scale unit costs per tonne of pollutant for low-level emissions from vehicles with internal combustion engine	urban	comparable local environment, i.e. population density and local meteorology; country-specific adjustment of monetary values
	costs due to fuel production (per litre of fuel or per vehicle kilometre)	extra-urban	comparable emission factors for production and transport of fuel; pollutants are emitted in the same geographical area
	costs from exhaust emissions per vehicle kilometre	urban, extra-urban	comparable emission factors and local environment and geographical area/regional scale unit costs

5.2 Costs due to Noise

Noise is a very local burden, marginal costs due to noise exposure are mainly determined by
- the distribution and distance of exposed persons from the source,
- the existing noise level, which in most cases is dominated by the traffic (number of vehicles, trains or aircraft per hour, mix of vehicle, train or aircraft types, speed),
- the time of day (variation in disturbance effects of noise).

Table 26 illustrates the broad variation in noise costs quantified in the different case studies. Marginal costs are generally higher at night time than at daytime, with a difference of up to a factor of three. This is due to the higher disturbance effects of noise at night and a lower background noise level. Differences between the case studies are large, reflecting the variability of marginal costs with the detailed population distribution, number and speed of vehicles, share of HGVs, etc.

Table 26
Overview of costs due to noise from road vehicles in €cent / vkm

		Passenger car		HGV	
		daytime	night time	daytime	night time
urban case studies	Helsinki	0.22	0.53	n.a.	n.a.
	Stuttgart	1.50	4.50	25.75	78.25
	Berlin	0.47	1.45	7.67	23.33
	Florence	a)			
inter-urban case studies	Helsinki – Turku	n.a.	n.a.	1.58	3.86
	Basel – Karlsruhe	0.02	0.03	0.11	0.18
	Strasburg – Neubrandenburg (outside built-up areas)	0	0	0	0
	Strasburg – Neubrandenburg (within built-up areas)	0.12	0.19	3.04	5.06
	Milano – Chiasso	0.01	0.04	0.09	0.35
	Bologna – Brennero	0.001	0.002	0.006	0.02

a) marginal costs not given per vehicle kilometre, but per 1 dB(A) reduction

The methodology can be generalised, considering mode-specific characteristics of noise propagation and exposure-response functions. A generalisation of input data and results is very difficult, due to the non-linearities involved and the variability of the local characteristics. Table 27 presents the main generalisation aspects.

Table 27
List of generalisation aspects for marginal costs due to noise

	aspect to be generalised	basic requirements for generalisation
Metho-dology	overall methodology: Impact Pathway Approach	can be generalised
Inputs	inputs to noise propagation models (different models for road, rail and aircraft transport required)	generalisation not recommended
	exposure-response functions for health impacts (partly mode-specific)	can be generalised (for same mode)
	monetary values for health impacts and amenity losses	country-specific adjustment/values
	noise emission factors for vehicle/train/aircraft types	only if same parameters (e.g. driving characteristics, speeds)
	noise emission factors for vehicle fleets	generalisation not recommended
Output values	costs per vehicle/train kilometre or aircraft movement	generalisation difficult due to high sensitivity concerning local characteristics

Marginal noise costs due to maritime shipping and inland waterway transport are negligible, because emission factors are comparably low and most of the activities occur outside densely populated areas and therefore relevant thresholds for observing effects are not exceeded.

5.3 Costs due to Global Warming

Due to the global character of global warming impacts, there is no difference where the emission of greenhouse gases takes place. For the choice of the abatement cost value a European perspective was taken, i.e. the value is applicable to all countries of the European Union. It is based on calculations for reaching the Kyoto targets of the European Union, optimising abatement measures in the Union and not country by country. It is assumed that measures for a reduction in CO2 emissions are taken in a cost effective way, implying that reduction targets are not set per sector, but that the cheapest measures are implemented, no matter in which sector or country.

Table 28
Overview of damage costs due to global warming from exhaust greenhouse gas emissions from road vehicles in €cent / vkm

		Car Petrol EURO2	Car Diesel EURO2	HGV Diesel EURO2
urban case studies	Helsinki	0.35	n.a.	n.a.
	Stuttgart	0.47	0.31	3.28
	Berlin	0.47	0.31	3.28
	Florence	0.69	0.43	2.00
inter-urban case studies	Helsinki – Turku	n.a.	n.a.	2.40
	Basel – Karlsruhe	0.37	0.32	2.18
	Strasburg – Neubrandenburg (outside built-up areas)	0.34	0.25	2.03
	Strasburg – Neubrandenburg (within built-up areas)	0.47	0.31	3.28
	Milano – Chiasso	0.36 [a)]	0.36 [a)]	2.16 [a)]
	Bologna – Brennero	0.36 [a)]	0.36 [a)]	2.16 [a)]
[a)] emission standard not specified				

Differences in the costs per km are caused only by variations in the emission factors, which vary with vehicle/train/aircraft/vessel type, fuel used, and operation parameters (e.g. traffic situation). Table 29 lists the main generalisation aspects.

Table 29
List of generalisation aspects for marginal costs due to global warming

	aspect to be generalised	basic requirements for generalisation
Metho-dology	overall methodology: application of abatement costs	can be generalised
Inputs	monetary value per unit of greenhouse gas emitted	value used for calculations is applicable for countries of the European Union
	exhaust emission factors for specific vehicle/train/vessel/ aircraft technologies	only if same parameters (e.g. driving characteristics, speeds)same driving characteristics/speeds
	exhaust emission factors for vehicle fleets	generalisation not recommended
	emission factors for the production and transport of fuel	refinery processes and fuel distribution are comparable
Output values	costs from exhaust emissions per vehicle kilometre	comparable emission factors
	costs due to fuel production (per litre of fuel or per vehicle kilometre)	comparable emission factors for production and transport of fuel

6 Conclusions

6.1 Major findings of the case studies

Marginal environmental costs were assessed for a number of specific routes in urban areas and important inter-urban relations, covering both passenger and goods transport. All modes were covered, and a broad range of vehicle types was considered for which costs related to the emission of air pollutants, greenhouse gases and noise proved to be relevant and quantifiable cost categories.

Due to the significant variations of results between the locations studied, reflecting the different characters and conditions of the relations, it is difficult to draw general conclusions concerning magnitude and composition of costs. It is not possible to derive one single value for the marginal environmental costs of a certain vehicle type in urban areas. Therefore, the cost categories have to be looked at separately, in particular when it comes to the issue of generalisation of results.

6.1.1 Air pollution

Quantifiable air pollution costs are dominated by health effects, in particular loss of life expectancy. Costs due to crop losses and materials are only of minor importance. Diesel vehicles cause considerably higher costs than petrol vehicles, resulting from much higher emissions of primary particles. The difference between both fuel types is highest in urban areas, because primary particles have very high local effects.

Besides exhaust emissions, emissions due to fuel production processes are relevant and gain in importance with stricter emission standards for road vehicles. For petrol cars complying with the EURO4 standard, costs due to fuel production are comparable to those from exhaust emissions. In the case of electric trains, the marginal costs quantified vary heavily depending on the fuel mix from which the electricity is produced – the lower the share of fossil fuels, the lower the resulting costs.

Further to variations in emission factors the costs are determined by the population density close to the emission source, the local meteorology (mainly average wind speed) and by the geographical location within Europe, which is important for the number of the population affected by long-range pollutant transport and the formation of secondary pollutants.

In the context of generalisation of results the comparison of the case studies clearly suggests that costs due to air pollution cannot be transferred based on the population density of the local environment only. Some general rules could be derived, but an operational formula for transfer requires a broader statistical basis of case studies. A generalisation methodology for air pollution costs should account for

- the local scale conditions (population density and local meteorology), and
- the regional scale costs per tonne of pollutant emitted in a certain area (e.g. NUTS1 level)

6.1.2 Global warming

For petrol cars complying with EURO2 emission standard, marginal costs associated to greenhouse gas emissions are higher than the costs due to air pollution; the same holds for the aircraft studied. For EURO2 diesel cars and heavy goods vehicles, global warming costs are generally lower than air pollution costs. The same is the case for trains with electric traction, the maritime passenger ferry, and the container ship on the Rhine.

Costs due to the emission of greenhouse gases are not location specific, as they are relevant on a global scale. Abatement costs were calculated based on the same monetary value (€20 per tonne of CO_2) for all case studies, which is applicable for all countries of the European Union. As a consequence all the variation is caused by the emission factor of a vehicle, vessel, aircraft, or the underlying electricity production process.

6.1.3 Noise

Noise costs are extremely variable; night time values up to a factor of three higher than day time values could be observed for road transport vehicles in urban areas. In extra-urban areas absolute levels and differences between day time and night time are much smaller. Marginal noise costs of maritime shipping and inland waterway transport were found to be negligible.

Noise costs are mainly determined by the population affected, the time of day (with higher disturbance effects of noise during the night) as well as the number of vehicles and their speeds, and the resulting background noise. The higher the existing background noise level, the lower the costs of an additional vehicle.

For marginal noise costs a generalisation is even more difficult than for air pollution, because of the large non-linearities involved and the variability of the relevant parameters in very short time spans.

The possible effect of marginal cost pricing in the case of noise is a very good illustration of issues, that have to be taken into account in the case of strong non-linearities of impacts. A pricing scheme based on marginal noise costs would lead to a bundling effect of traffic. Marginal costs are strongly decreasing with increasing background noise levels. For this reason driving on a route where noise levels are high already would be much cheaper than driving in quiet areas. Of course this is a perfect solution for allocation of noise emitters from the perspective of economic theory. However, it has to be ensured that no absolute limits, such as thresholds for health risks or amenity losses are exceeded. In practice, other price components (e.g. air pollution costs) may attenuate the bundling effect.

6.2 Methodology and data used

In general, the models and data used reflect best current knowledge; however this knowledge has gaps and therefore the results are subject to uncertainty.

The impact pathway approach (IPA) for air pollution, including the respective dispersion models, exposure-response functions and monetary values, is well established and has been applied in a large number of research projects. Case study specific data which had to be provided comprised emission factors for direct emissions and fuel production processes, the power plant mix for electricity production, and detailed data on local meteorology. The local meteorology was found to be a very important determinant for air pollution costs in urban areas. For this reason special attention should be paid to the quality of this kind of data in future case studies, as was in the UNITE case studies. Compared to the emission factors available for road vehicles, those available for marine and inland waterway vessels as well as for aircraft are less elaborated. In the special case of high altitude aircraft emissions information on the dispersion and chemical conversion of pollutants is urgently required to be able to assess the impacts of all flight phases adequately.

The application of the IPA in the context of noise is relatively new and may be subject to revision and extension in the future, in particular the exposure-response functions. Due to the very local nature of noise and the strong non-linearities involved, a lot of detailed input data is required. Availability of noise emission factors for different vehicles is only restricted. The integration of a propagation model for aircraft noise will allow the application of exposure-response functions as for road and railway noise.

In the context of estimating costs due to global warming the main source of uncertainty is the valuation factor used. This reflects the huge uncertainties about the share of anthropogenic emissions in climate change and the associated effects, as well as reasonable reduction targets for greenhouse gas emissions. Compared to this, the emission factors for CO_2 and other greenhouse gases appear very reliable.

In one case study (Florence, Italy) an alternative approach of replacing the dispersion modelling step by statistical correlation between vehicle mileage and pollutant or noise measurements was explored. The results are not directly comparable with those of the other case studies, because with this approach only local scale effects can be covered and the range of pollutants considered was different. Health effects due to noise could not be quantified due to the strong non-linearities involved and the unavailability of required noise indices. It remains debatable if the application of this approach is significantly simpler and cheaper than the established impact pathway approach, which was one of the motivations for its development.

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Glossary

CORINAIR	Programme to establish an inventory of emissions of air pollutants in Europe. It was initiated by the European Environment Agency Task Force and was part of CORINE (COoRdination d'Information Environmentale) work programme set up by the European Council of Ministers in 1985. End of 1994 the EEA's European Topic Centre on Air Emissions (ETC/AEM) took over the CORINAIR programme.
GDP	(= Gross Domestic Product). The GDP is the sum of all goods and services produced within a country and a year. GDP per capita can be regarded as the relative economic power of a country per inhabitant.
Impact Pathway Approach (IPA)	Methodology for externality quantification developed in the ExternE project series. It follows the chain of causal relationships from pollutant emission via dispersion (including chemical transformation processes), leading to changes in ambient air concentrations from which impacts can be quantified using exposure-response functions. Damages are then calculated using monetary values based on the WTP approach.
NUTS	Nomenclature of territorial units for statistics; level 0 = countries, level III = départements, Kreise, etc. (depending on country considered).
PPP	PPP means purchasing power parity. PPPs are the rates of currency conversions which equalise the purchasing power of different countries. This means that a given sum of money, when converted into different currencies at the PPP rates, will buy the same basket of goods and services in all countries. In particular, PPPs are applied if figures for specific products or branches shall be expressed in foreign currency (for example in ECU or in US $) because in these cases the use of official exchange rates is not appropriate.
Primary particles	Particles, that are directly emitted.
Secondary particles	Particles, such as nitrates and sulphates, that are formed in the atmosphere through atmospheric chemical reactions.
Vehicle category	Road: passenger car, motorcycle, bus, goods transport vehicles. Public transport: bus, tram, trolley bus, metro. Rail: electric passenger train, diesel passenger train, electric goods train, diesel goods train. Inland Waterways / Marine: Goods transport. Air: passenger, goods transport
WTP	Willingness to pay: The explicit or implicit willingness-to-pay for a good, reflecting the individual's preferences. For example the WTP for higher safety.
YOLL	Year of life lost

Abbreviations

CH_4	Methane (greenhouse gas)
CO_2	Carbon dioxide (greenhouse gas)
COI	Cost of illness
dB(A)	Decibel, weighted with the *A*-filter. Logarithmic unit of sound pressure level.
EMEP	European Monitoring and Evaluation Programme
GDP	Gross Domestic Product
GIS	Geographical Information System
GWP	Global warming potential
HGV	Heavy goods vehicle
kWh	Kilowatt hour
L_{Aeq}	Energy equivalent noise level
LTO	Landing and take-off cycle
mill.	Million
MWh	Megawatt hour
N_2O	Nitrous oxide (greenhouse gas)
n.a.	No data available
NMVOC	Non-methane volatile organic compounds
NOx	Nitrogen oxides (mix of NO and NO_2)
NUTS	Nomenclature of territorial units for statistics; level 0 = countries, level III = départements, Kreise, etc. (depending on country considered)
PM_{10}	Fine particles with a diameter of 10 µm and less
$PM_{2.5}$	Fine particles with a diameter of 2.5 µm and less
PPP	Purchasing power parity
PT	Public transport
SO_2	Sulphur dioxide
vkm	vehicle kilometre
VOC	Volatile organic compounds
WTP	Willingness to pay
YOLL	Years of life lost