COMPETITIVE AND SUSTAINABLE GROWTH (GROWTH) PROGRAMME

UNITE

UNIfication of accounts and

marginal costs for Transport Efficiency

Marginal Costs Case Study 9E: Inter-Urban Road and Rail Case Studies Germany

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Table of Contents

D.0 Executive Summary

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 related to the emission of air pollutants, greenhouse gases and noise proved to be relevant and quantifiable cost categories. Cost estimates were performed using the impact pathway approach for air pollutants and noise. Greenhouse gas emissions were valued based on a shadow value for reaching the Kyoto reduction targets in the European Union. Costs of air pollution and global warming were assessed not only for vehicle operation but as well for fuel and electricity production.

The composition of marginal costs due to the emission of air pollutants, greenhouse gases and noise for the case studies analysed differs, reflecting the different characters of the routes. For road transport the sum of all three cost categories per vehicle kilometre is similar for comparable vehicles when related to the whole route considered. But whereas variations on different sections of the motorway are comparably small, because it passes settled areas with a certain distance, the drive from Strasburg to Neubrandenburg leads directly through a number of towns and villages. Therefore the results for this route were split into road sections within built-up areas and outside built-up areas. The single cost categories vary to different degrees.

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.

D.1 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 the vehicle, train, ship or plane 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, plane), 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. In the following the application of the IPA to two inter-urban case studies in Germany is presented.

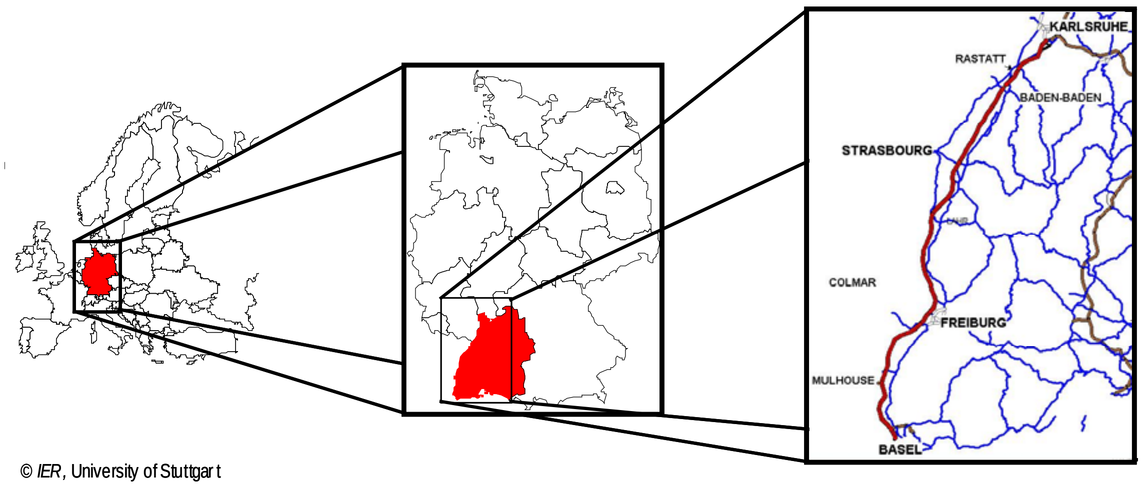
D.2 Case Study Description

Air pollution and noise costs are closely related to population density in the vicinity of a route. In the inter-urban case studies the size of the population density effect was explored for two areas with considerable differences in population densities: the federal states of Baden-Württemberg and Mecklenburg-Vorpommern. Both areas can be characterised as “rural”, however the population densities differ considerably. In Baden-Württemberg on average 293 persons share one km^2 of land, whereas in Mecklenburg-Vorpommern 77 persons live on one km^2. Road vehicle types covered comprise car, motorcycle, bus, and lorry. Different emission standards are 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.

D.2.1 Location

The route from Basel to Karlsruhe (see Figure E-1) 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.

Figure E-1 Geographical map of the route considered from Basel to Karlsruhe



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 (see Figure E-2).

Figure E-2 Route analysed for road transport from Strasburg to Neubrandenburg

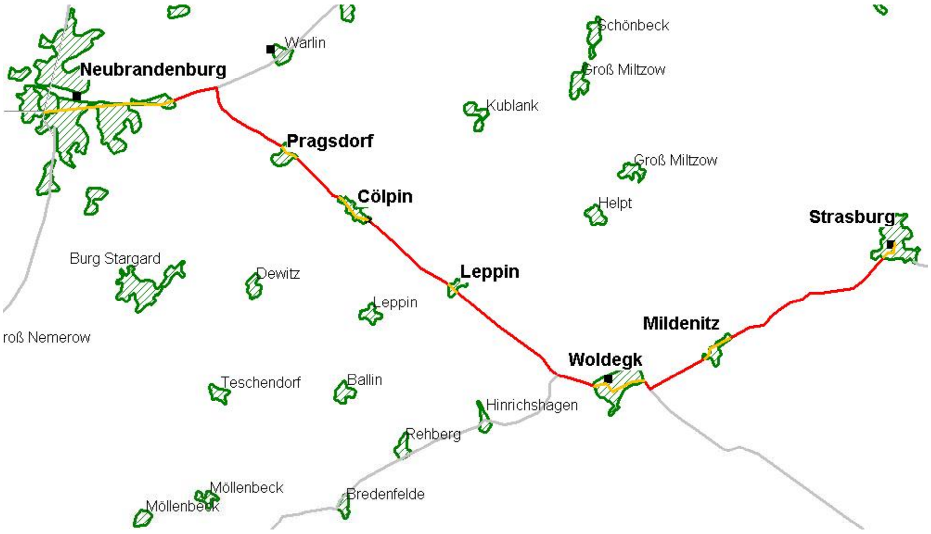


Figure E-3 shows that the railway line does not cross built-up areas, apart from start and end. This has major implications on noise costs, as only few people are affected by railway noise, compared to the route from Basel to Karlsruhe, where the line crosses many built-up areas.

Figure E-3 Route analysed for rail transport from Strasburg to Neubrandenburg

D.2.2 Methodology

Marginal costs in this case study means the environmental costs caused by an additional vehicle driving on a specific route. For noise costs the time of day is relevant as well, due to the sensitivity of the receptors (which is different at night than during the day) and the high importance of the background noise level, which varies with traffic density, for the results.

This 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) or Friedrich and Bickel (2001).

D.2.2.1 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.

Emissions/burdens

In the first step the emissions from an additional vehicle on a specific route are calculated. For comparisons between modes, the system boundaries considered are very important. For instance, when comparing externalities of goods transport by electric trains and heavy duty road vehicles, the complete chain of fuel provision has to be considered for both modes. 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.

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.

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

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 E-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 E-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 E-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: $$1/t = 0.12 \cdot \left(1 - e^{\frac{-0.121 \cdot Rh}{100 - Rh}}\right) \cdot [SO_2] + 0.0174 \cdot [H^+] \Big/ R$$

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 E-3:

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

Table E-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

Monetary Valuation

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

Impact	European average	Germany	
Year of life lost (chronic effects)	74 700	80,600	€ per YOLL
Year of life lost (acute effects)	128 500	138,700	€ per YOLL
Chronic bronchitis	137 600	148,500	€ per new case
Cerebrovascular hospital admission	13 900	15,000	€ per case
Respiratory hospital admission	3 610	3,900	€ per case
Congestive heart failure	2 730	2,950	€ per case
Chronic cough in children	200	210	€ per episode
Restricted activity day	100	100	€ per day
Asthma attack	69	74	€ per day
Cough	34	37	€ per day
Minor restricted activity day	34	37	€ per day
Symptom day	34	37	€ per day
Bronchodilator usage	32	35	€ per day
Lower respiratory symptoms	7	8	€ per day

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

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.

D.2.2.2 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.

D.2.2.3 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 road. 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 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.

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 calculated with a GIS approach for the interurban case studies.

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 E-5.

Table E-5
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 E-6. They were applied as well for road as for rail traffic noise.

Table E-6
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.

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 16 € per dB(A) and is based on HP-findings by Soguel (1994). Soguel reports an NSDI of 0.9 on monthly housing rents, net of charges. This value is similar to the average derived from European studies and was taken for our calculations. It is applied to German average net rent of 1791 per person per year. The monetary values applied are presented in Table E-7.

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 E-7
Monetary values (factor costs, rounded) for impacts due to noise in Germany(€$_{1998}$)

Impact	
Myocardial infarction (fatal, 7 YOLL)	
Total per case	**564 000**
Myocardial infarction (non-fatal, 8 days in hospital, 24 days at home)	
Medical costs	4 700
Absentee costs	3 500
WTP	16 300
Total per case	**24 500**
Angina pectoris (severe, non-fatal, 5 days in hospital, 15 days at home)	
Medical costs	2 900
Absentee costs	2 200
WTP	10 200
Total per case	**15 300**
Hypertension (hospital treatment, 6 days in hospital, 12 days at home)	
Medical costs	1 800
Absentee costs	2 000
WTP	600
Total per case	**4 400**
Medical costs due to sleep disturbance (per year)	210
WTP (per year)	425
WTP for avoiding amenity losses (€/dB/person/year)	16
Source: Own calculations based on Metroeconomica (2001); country-specific valuation based on Nellthorp et al. (2001), WTP for avoiding amenity losses see text	

D.2.2.4 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 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)

D.2.3 Data

D.2.3.1 Data for the calculation of costs due to airborne emissions

Besides the emissions from the vehicles considered in the case study, 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 E-8
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.

For dispersion modelling on the local scale data sets based on 10 year's averages of 3-hourly measured data by the German meteorological service (Deutscher Wetterdienst, 1999) were used.

Emissions road vehicles

A broad range of vehicles was analysed, covering the most relevant vehicle types and emission standards. Vehicle emissions were modelled, taking into account driving pattern and traffic situation on the respective route. Table E-9 shows the emission and fuel consumption factors per vehicle kilometre for the motorway drive from Basel to Karlsruhe. Table E-10 and Table E-11 present the respective information for the drive from Strasburg to Neubrandenburg for the sections within and outside built-up areas respectively. The emission factors per vehicle kilometre vary depending on the driving pattern on the road. A motorway drive in general can be characterised by a more or less constant but higher speed than on other roads. Depending on the pollutant and engine technology this leads to different effects in terms of emissions. For example the NO_x emission factors are the highest on motorways, with the exception of heavy diesel vehicles. Those emit most NO_x per vehicle kilometre on roads within built-up areas.

Table E-9
Emission and fuel consumption factors in g/vkm for motorway drive from Basel to Karlsruhe

Vehicle type	**Motor-cycle**	**Passenger Car**						**LGV**	**HGV**	**Coach**
Fuel	petrol	petrol			diesel			diesel	diesel	diesel
Standard	EURO0	EURO1	EURO2	EURO4	EURO1	EURO2	EURO4	EURO2	EURO2	EURO2
CH_4	0.030	0.010	0.010	0.009	0.001	0.001	0.001	0.001	0.010	0.010
CO	23.68	1.98	1.69	1.43	0.30	0.20	0.15	0.38	0.61	0.35
CO_2	138.7	180.5	177.8	167.5	162.1	159.5	149.2	305.6	1080.8	769.8
Benzene	0.0500	0.0061	0.0042	0.0023	0.0008	0.0006	0.0004	0.0007	0.0099	0.0088
Fuel use	43.7	56.9	56.0	52.8	51.1	50.2	47.0	96.3	340.4	242.4
NMVOC	1.04	0.06	0.05	0.03	0.04	0.02	0.02	0.03	0.50	0.45
NO_x	0.55	0.75	0.50	0.17	0.59	0.47	0.23	0.76	8.87	6.65
N_2O	0.005	0.040	0.030	0.010	0.008	0.008	0.008	0.008	0.030	0.030
$PM_{2.5}$	n.a.	0.005	0.005	0.005	0.100	0.070	0.020	0.110	0.160	0.100
SO_2	0.010	0.017	0.006	0.005	0.036	0.035	0.005	0.060	0.240	0.170
Source: UBA/BUWAL 1999; n.a. = not available										

Table E-10
Emission and fuel consumption factors in g/vkm for drive from Strasburg to Neubrandenburg – road sections within built-up areas

Vehicle type	**Motor-cycle**	**Passenger Car**						**LGV**	**HGV**	**Public Tr. Bus**	**Coach**
Fuel	petrol	petrol			diesel			diesel	diesel	diesel	diesel
Standard	EURO0	EURO1	EURO2	EURO4	EURO1	EURO2	EURO4	EURO2	EURO2	EURO2	EURO2
CH_4	0.060	0.020	0.010	0.008	0.002	0.001	0.001	0.002	0.030	0.020	0.030
CO	17.85	1.85	1.80	0.76	0.59	0.40	0.29	0.37	1.04	1.64	1.08
CO_2	143.6	226.9	223.8	211.2	157.0	154.5	144.6	276.1	1627.7	1261.6	1129.5
Benzene	0.090	0.007	0.005	0.002	0.002	0.001	0.001	0.0016	0.0200	0.0100	0.0200
Fuel use	45.2	71.5	70.5	66.5	49.4	48.7	45.5	87.0	512.7	397.3	355.8
NMVOC	1.88	0.06	0.05	0.02	0.08	0.05	0.04	0.08	1.28	0.97	1.38
NO_x	0.17	0.36	0.24	0.05	0.58	0.46	0.23	0.74	16.07	13.66	11.07
N_2O	0.005	0.040	0.030	0.010	0.008	0.008	0.008	0.008	0.040	0.030	0.030
$PM_{2.5}$	n.a.	0.005	0.005	0.005	0.060	0.050	0.010	0.070	0.360	0.250	0.260
SO_2	0.010	0.021	0.007	0.007	0.035	0.034	0.005	0.060	0.360	0.280	0.250
Source: UBA/BUWAL 1999; n.a. = not available											

Table E-11
Emission and fuel consumption factors in g/vkm for drive from Strasburg to Neubrandenburg – road sections outside built-up areas

Vehicle type	**Motor-cycle**	**Passenger Car**						**LGV**	**HGV**	**Public Tr. Bus**	**Coach**
Fuel	petrol	petrol			diesel			diesel	diesel	diesel	diesel
Standard	EURO0	EURO1	EURO2	EURO4	EURO1	EURO2	EURO4	EURO2	EURO2	EURO2	EURO2
CH_4	0.030	0.010	0.010	0.005	0.001	0.001	0.0005	0.001	0.010	0.010	0.010
CO	17.97	1.20	1.17	0.49	0.25	0.17	0.12	0.24	0.59	1.19	0.36
CO_2	112.3	162.0	159.7	150.7	125.1	123.2	115.2	231.4	1004.5	899.7	693.0
Benzene	0.050	0.005	0.003	0.001	0.001	0.001	0.000	0.0008	0.0097	0.0087	0.0092
Fuel use	35.4	51.0	50.3	47.5	39.4	38.8	36.3	72.9	316.4	283.4	218.3
NMVOC	1.01	0.04	0.03	0.01	0.03	0.02	0.01	0.04	0.49	0.44	0.47
NO_x	0.28	0.38	0.26	0.06	0.44	0.35	0.17	0.62	8.64	9.66	5.98
N_2O	0.005	0.040	0.030	0.010	0.008	0.008	0.008	0.008	0.030	0.030	0.030
$PM_{2.5}$	n.a.	0.005	0.005	0.005	0.040	0.030	0.010	0.050	0.170	0.160	0.100
SO_2	0.010	0.015	0.005	0.005	0.028	0.027	0.004	0.050	0.220	0.200	0.150
Source: UBA/BUWAL 1999; n.a. = not available											

Beside these emissions from vehicle operation the emissions due to fuel provision were considered. The emission factors for crude oil extraction, refining and transport of petrol and diesel are given in Table E-12.

Table E-12
Emissions caused by fuel production processes in g/kg fuel

Type of fuel	CO_2	PM_{10}	NO_x	SO_2	NMVOC
Petrol	560	0.105	1.10	1.90	1.80
Diesel	400	0.047	0.96	1.40	0.62
Source: PM_{10}: Friedrich and Bickel (2001); other pollutants: IFEU (1999)					

It is assumed that the petrol and diesel are produced in refineries in Germany. Emissions associated with fuel production are valued with average damage factors for emissions in Germany. These damage factors were calculated based on the assumption that the emission source is not located within densely populated areas.

Table E-13
Damage factors for emissions from refineries

Pollutant	NO_x	NMVOC	SO_2	PM_{10}
€ per tonne emitted	4520	1580	4570	7070
Source: own calculations				

Costs due to electricity production

The costs due to power plant emissions (including fuel extraction, transport and where applicable refinery) in Germany were calculated with EcoSense. For costs from other effects than emissions from combustion processes, mainly due to hydro and nuclear power plants, detailed calculations performed in ExternE were used. So, the methodology is compatible with the calculations for road transport vehicles, monetary values were adjusted according to the UNITE valuation conventions. Costs per kWh of electricity were calculated, using the electricity production mix of Deutsche Bahn AG as given in Table E-14.

Table E-14
Share of fuels in the electricity production of Deutsche Bahn AG

Coal	34.4%
Nuclear	22.1%
Oil/natural gas	13.2%
Hydro	10.1%
Electricity from public grid	20.2%
Total	100.0%
Source: Deutsche Bahn (1998)	

Table E-15 shows the data of the trains considered. Modern trains recover energy from braking; for the calculations it was assumed that this compensates for losses from electricity transformation and in the grid.

Table E-15
Technical data of trains for public rail transport

	Max. Capacity	Energy use kWh/km	operation between
Intercity	600 persons	16.6	Basel and
Goods train	835 tonnes	20.9	Karlsruhe
Local train	340 persons	10.6	Strasburg and
Local goods train	524 tonnes	14.3	Neubrandenburg
Source:	Bialonski et al. (1990)		

D.2.3.1 Data for the calculation of noise costs

Main input to the calculation of noise costs is the average annual daily traffic (AADT). Table E-16 shows the AADT values used for the case studies. AADT is broken down to vehicles per hour by application of an average time curve. The numbers applied for the case studies for day, evening and night traffic are presented in Figure E-4 and Figure E-5.

Table E-16
Average annual daily traffic (AADT) used in the case studies

	Passenger Car	LGV	HGV	Motorcycle	Bus	Source
Basel - Karlsruhe	2858 – 9869[1)]	27 - 297[1)]	375 - 1536[1)]	34 - 185[1)]	26 - 98[1)]	Wickert (2001)
Strasburg – Neubrandenburg	6015	506	602	79	57	Wickert (2001)
[1)] total route analysed consists of many sections on which AADT values vary						

The route from Basel to Karlsruhe (total length 210 km) consists of many road sections on which AADT values vary considerably. For this reason, Table E-16 and Figure E-4 indicate the range of values observed on the different road sections. Calculations were based on the value for the specific road section.

Figure E-4 Number of vehicles per hour for day, evening, night on the route Basel – Karlsruhe (bars indicate the variation on the route sections considered; source: own calculations based on Wickert (2001))

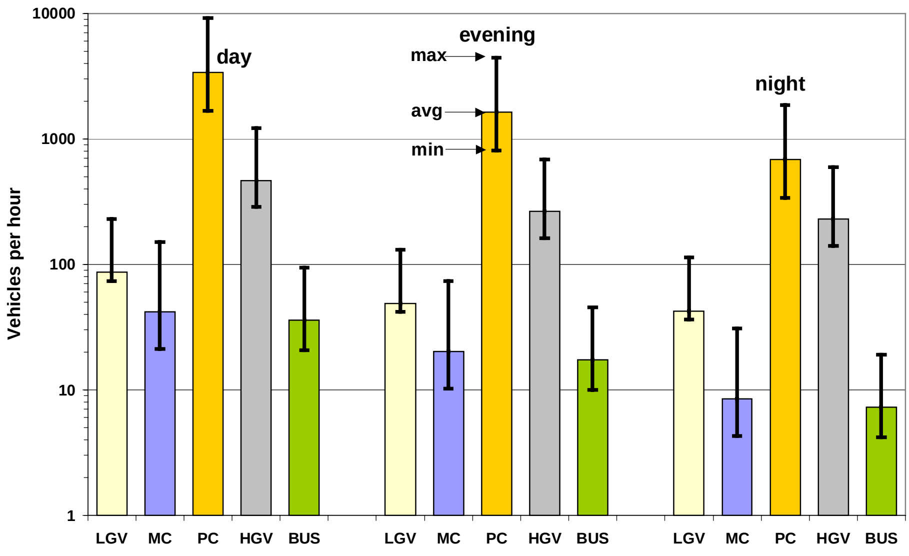


Figure E-5 Average number of vehicles per hour for day, evening, night on the route Strasburg - Neubrandenburg (Source: own calculations based on Wickert (2001))

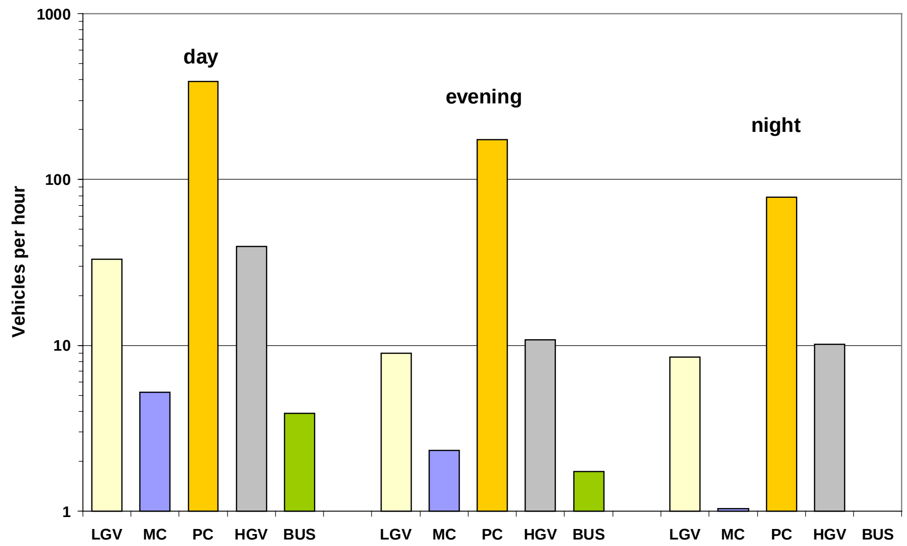


Table E-17
Number of trains per hour for time periods day, evening, night for calculation of marginal noise costs

		day	evening	night	Source
Basel – Karlsruhe	Intercity trains	2	1.5	0.75	time table
	Other passenger trains	4.7	4.5	1.25	Deutsche Bahn
	Goods trains	2.5	8	9.75	Stekeler (1996)
	total	9.2	14	11.75	
Strasburg -	Local trains	2	2	0	time table DB
Neubrandenburg	Local goods trains	0.5	0.5	0	own estimate

D.3 Results

Table E-18 presents the marginal costs due to airborne emissions per vehicle kilometre for the trip from Basel to Karlsruhe. For road transport vehicles, total costs are dominated by direct emissions from vehicle use, costs due to fuel production emissions play only a minor role. The trains considered run on electricity, therefore costs from direct emissions (“vehicle use”) are zero. Total costs for trains, both from air pollution and global warming appear considerable compared to the vehicles with internal combustion engines. These values have to be put into perspective by the much higher load that is transported by one train. Assuming average load for all vehicles the trains show the lowest costs per passenger or tonne kilometre.

Table E-18
Marginal costs due to airborne emissions in EUR/100 vkm – Basel-Karlsruhe

		air pollution			global warming			Total
		vehicle use	fuel/electr. prod.	total	veh. use	fuel/electr. Prod.	total	
Petrol	Motorcycle[a)]	0.57	0.08	0.65	0.28	0.05	0.33	0.98
	Car Euro1	0.55	0.10	0.65	0.39	0.06	0.45	1.10
	Car Euro2	0.37	0.10	0.47	0.37	0.06	0.44	0.91
	Car Euro4	0.15	0.09	0.24	0.34	0.06	0.40	0.64
Diesel	Car Euro1	0.84	0.06	0.90	0.33	0.04	0.37	1.27
	Car Euro2	0.63	0.06	0.70	0.32	0.04	0.36	1.06
	Car Euro4	0.24	0.06	0.30	0.30	0.04	0.34	0.64
	LGV[b)]	1.02	0.12	1.13	0.62	0.08	0.69	1.83
	HGV[b)]	6.91	0.41	7.32	2.18	0.27	2.45	9.77
	Coach[b)]	5.10	0.29	5.40	1.56	0.19	1.75	7.15
Electric traction	Intercity	0[c)]	25.41	25.41	0[c)]	16.83	16.83	42.24
	Goods train	0[c)]	32.03	32.03	0[c)]	21.22	21.22	53.25

[a)] EURO0; [b)] EURO2; [c)] Relevant emissions only from electricity production.

Air pollution costs are clearly dominated by mortality and morbidity effects. Compared to the costs due to health effects, quantifiable costs due to material damages and crop losses are of minor importance.

For light duty vehicles global warming has a considerable share in the costs due to airborne emissions. The cleaner the vehicle (i.e. the higher the emission standard), the higher the importance of greenhouse gas emissions (CO_2, CH_4 and N_2O), above all from vehicle use. Diesel vehicles cause higher air pollution costs per vehicle kilometre than petrol vehicles with a comparable emission standard. This is mainly caused by the higher particle and NO_x emissions of diesel engines. The shares of the different pollutants in the costs are illustrated in Figure E-6 for passenger cars complying with EURO2 standard. For petrol vehicles the costs are dominated by nitrates formed from NO_x emissions and by greenhouse gas emissions. For diesel cars primary particle emissions are the third main source of costs.

An interesting effect is the negative cost due to ozone formation from the precursor emissions NOx and NMVOC. In Germany the situation concerning ozone formation is very special.

Caused by the existing NOx / NMVOC background concentrations an additional unit of NOx leads to a reduction in ozone and thus a decrease in ozone damages. On the other hand NMVOC emissions cause damages due to ozone formation. So the negative costs shown in Figure E-6 is the result of the effects of NOx and NMVOC, where the NOx effect prevails, leading to negative costs. But compared to the adverse effects of NOx emissions via nitrate formation this "benefit" is negligible.

Figure E-6 Composition of costs due to airborne emissions from vehicle use – Basel-Karlsruhe

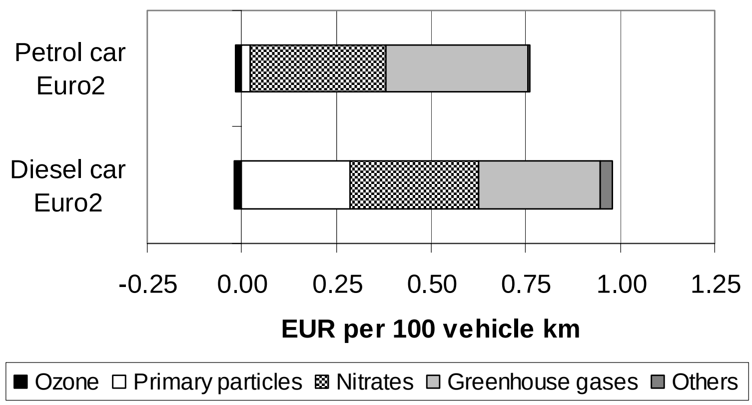


Figure E-7 shows the composition of costs due to airborne emissions from vehicle use for the trip from Strasburg to Neubrandenburg. Compared to the Basel-Karlsruhe drive the costs are 34% lower for the petrol car and 41% lower for the diesel car. Most of this is due to much lower costs caused by nitrates, which shows as well in the low relative share of nitrates in Figure E-7. This effect is caused by the geographical location in the north-east of Germany where NOx emissions lead to much lower costs due to nitrate formation, mainly because less people are affected.

Figure E-7 Composition of costs due to airborne emissions from vehicle use – Strasburg-Neubrandenburg

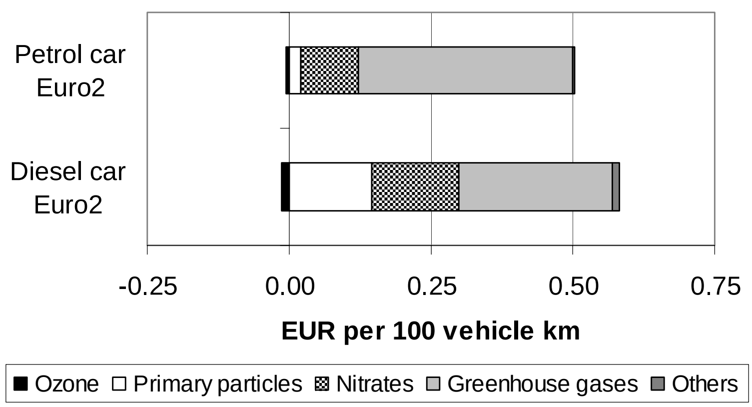


Table E-19 presents the results for the trip from Strasburg to Neubrandenburg. Air pollution costs due to fuel and electricity production and global warming costs directly reflect differences in fuel or electricity consumption and greenhouse gas emission factors, because for these cost categories the same damage factors apply.

Table E-19
Marginal costs due to airborne emissions in EUR/100 vkm – Strasburg-Neubrandenburg

		air pollution			global warming			Total
		vehicle use	fuel/electr. prod.	total	veh. use	fuel/electr. prod.	total	
Petrol	Motorcycle[a)]	0.33	0.07	0.39	0.25	0.04	0.29	0.68
	Car Euro1	0.17	0.10	0.27	0.39	0.06	0.45	0.72
	Car Euro2	0.12	0.10	0.22	0.38	0.06	0.44	0.66
	Car Euro4	0.04	0.09	0.14	0.35	0.06	0.40	0.54
Diesel	Car Euro1	0.37	0.05	0.43	0.27	0.03	0.31	0.73
	Car Euro2	0.30	0.05	0.35	0.27	0.03	0.30	0.65
	Car Euro4	0.11	0.05	0.16	0.25	0.03	0.28	0.44
	LGV[b)]	0.49	0.09	0.58	0.50	0.06	0.56	1.14
	HGV[b)]	4.99	0.45	5.44	2.41	0.30	2.72	8.16
	Coach[b)]	3.45	0.31	3.77	1.67	0.21	1.88	5.65
	Public tr. bus[b)]	4.77	0.38	5.16	2.04	0.25	2.30	7.45
Electric traction	Local pass. train	0[c)]	16.23	16.23	0[c)]	10.75	10.75	26.98
	Local goods train	0[c)]	21.87	21.87	0[c)]	14.49	14.49	36.37

[a)] EURO0; [b)] EURO2; [c)] Relevant emissions only from electricity production.

Air pollution costs from vehicle use are considerably lower compared to the trip from Basel to Karlsruhe. These differences stem from three components: the emission factors, the situation on the local scale (i.e. the area up to 25 km around the emission source) and regional scale effects (comprising dispersion and chemical transformation of pollutants over Europe).

As illustrated in Figures E-6 and E-7 the most important pollutants for the air pollution costs are primary particles and NO_x. So differences in the emission factors of these two species between to two routes are of highest relevance. On the local scale costs are mainly driven by the size of the population affected.

Costs on the regional (i.e. here European) scale are driven by meteorology, background concentrations of precursor substances and the population affected. Due to the prevailing west winds, emissions in the northeast of Germany are transported towards the baltic sea and Scandinavia. In these areas the population density – the most important single driver of costs – is much lower than in the areas affected by the emissions from the Southwest. So in tendency regional scale costs are higher for emissions on the motorway route considered than in the other. 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. This effect overlays the population effect, which leads to deviation from the general tendency, above all for costs caused by SO2 emissions. However these do not have a significant share in the total costs caused by air pollution.

Table E-19 presents the costs per vehicle kilometre for the total route from Strasburg to Neubrandenburg. As the route for road transport leads through built-up and non built-up areas, the costs vary due to varying emission factors and the local scale effects of population density close to the road. These variations are most significant for heavy diesel vehicles (HGV and coach – see Figure E-8), whose emission factors increase considerably when driving on roads within built-up areas. The variation in costs is amplified when taking into account noise costs, which will be done below.

Figure E-8 Variation of marginal costs due to airborne emissions for the drive from Strasburg to Neubrandenburg

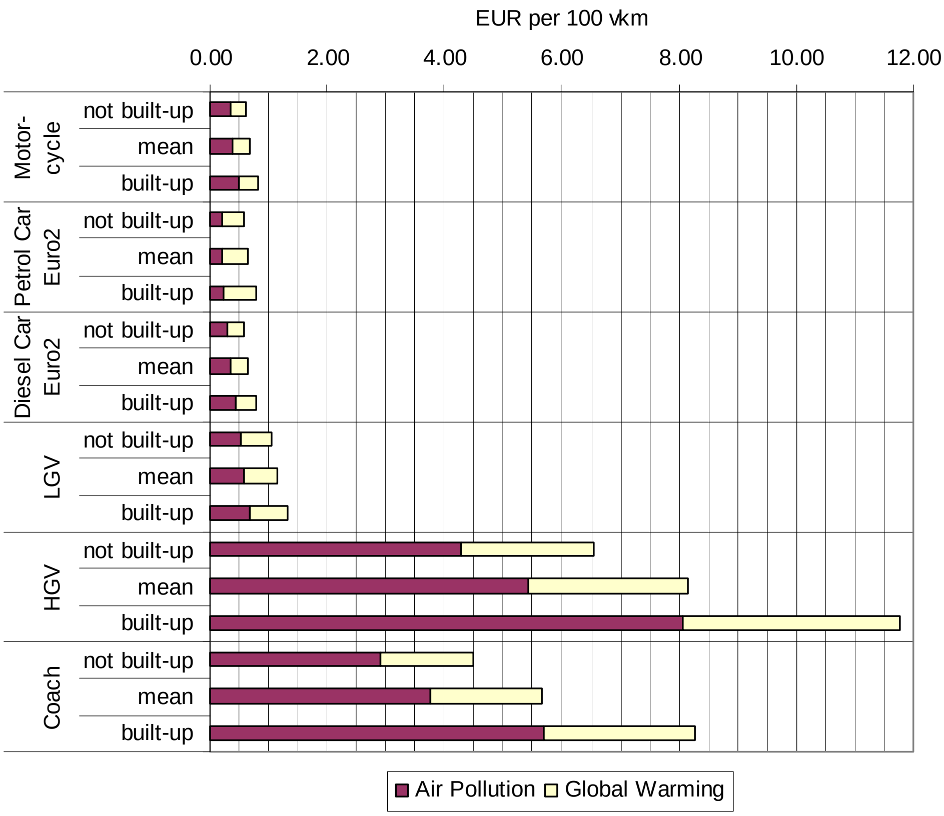


Table E-20 presents the marginal costs due to noise for the relation Basel – Karlsruhe. Costs are increasing from day and evening to night, reflecting the higher disturbance effect of noise during night time. The pattern of changes in costs for trains is different. Costs decrease from day to evening due to an increasing number of (goods) trains (see Table E-17). 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 (in general a goods train causes much more noise than a passenger train).

Table E-20
Marginal costs due to noise in EUR/100 vkm – Basel-Karlsruhe

	day	evening	night
Motorcycle	0.04	0.05	0.07
Car	0.02	0.02	0.03
LGV	0.06	0.07	0.10
HGV	0.11	0.12	0.18
Coach	0.05	0.06	0.08
Intercity	6.17	3.99	4.63
Goods train	22.25	11.31	15.74

For road transport noise costs the pattern presented in Table E-21 is the same as above. The costs for road vehicles are much higher than for the motorway drive, due to a higher number of persons exposed to noise, and much lower traffic density with lower vehicle speeds and resulting background noise level. Noise costs on road sections outside built-up areas are negligible. The situation for trains is opposite to that for roads. 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.

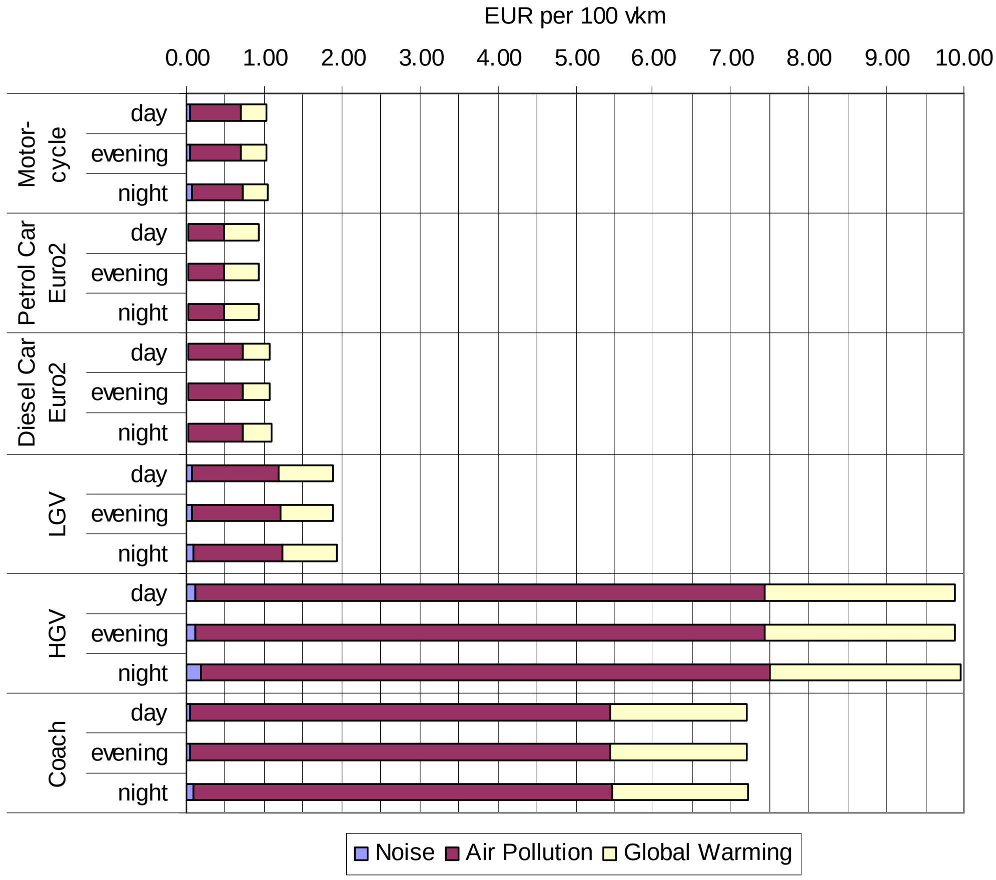
Table E-21
Marginal costs due to noise in EUR/100 vkm – Strasburg-Neubrandenburg (roads within built-up areas)

	day	evening	night
Motorcycle	0.73	0.73	1.22
Car	0.12	0.12	0.19
LGV	0.89	0.89	1.49
HGV	3.04	3.04	5.06
Public transport bus / Coach	0.70	0.70	1.17
Local passenger train	2.26	2.26	0[a)]
Local goods train	4.22	4.22	0[a)]

[a)] no trains at night time

Figure E-9 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 E-9 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 E-10. 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. Costs due to airborne emissions loose even more in importance for cleaner vehicles.

Figure E-11 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 as given in Table E-15. 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 variations in the noise costs for different daytimes were explained above.

Figure E-10 Sum of marginal costs for road transport from Strasburg to Neubrandenburg (roads within built-up areas)

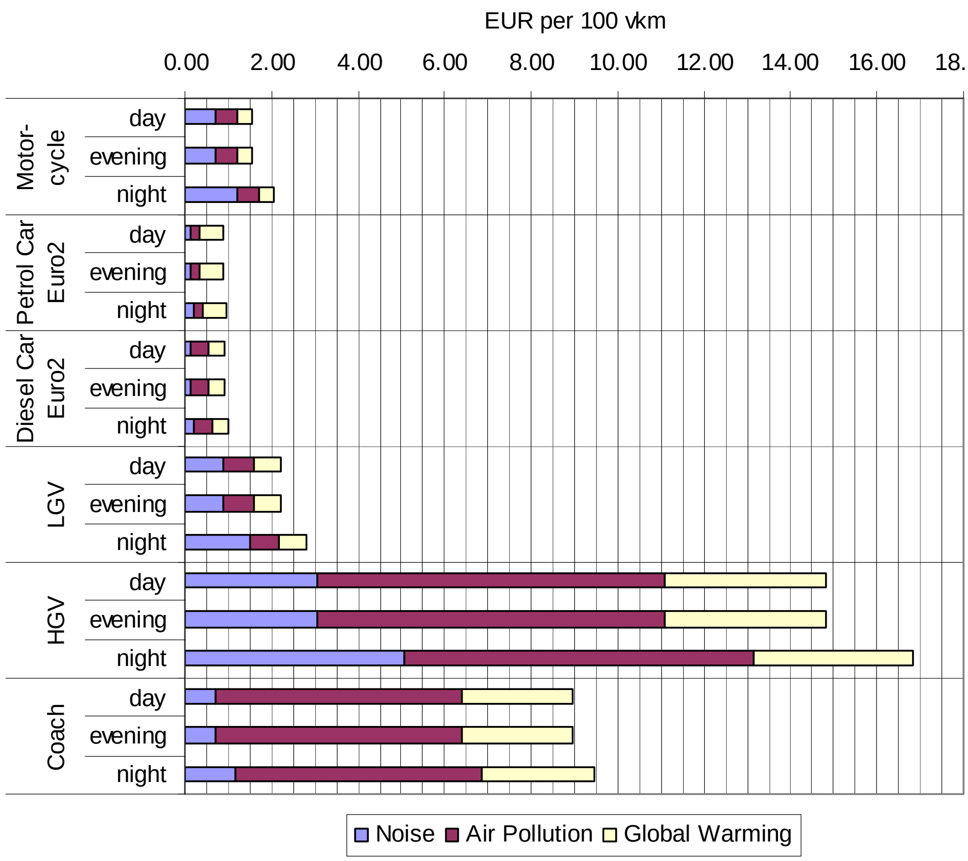
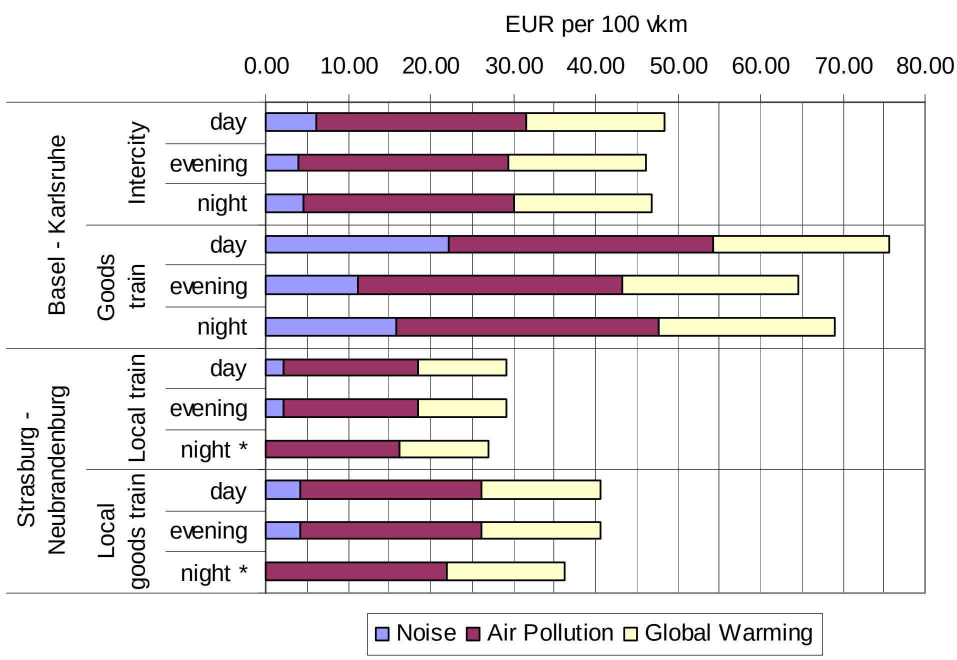


Figure E-11 Sum of marginal costs for inter-urban rail transport (* no trains during night time)



D.4 Discussion and conclusions

The composition of marginal costs due to the emission of air pollutants, greenhouse gases and noise for the case studies analysed differs, reflecting the different characters of the routes. For road transport the sum of all three cost categories per vehicle kilometre is similar for comparable vehicles when related to the whole route considered. But whereas variations on different sections of the motorway are comparably small, because it passes settled areas with a certain distance, the drive from Strasburg to Neubrandenburg leads directly through a number of towns and villages. Therefore the results for this route are split into road sections within built-up areas and outside built-up areas. 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 geographical location within Germany, which is important for the formation of secondary pollutants and population affected by long-range pollutant transport, and by the local scale population affected.
- Noise costs are determined by the time of day, the total vehicle flow and vehicle speeds (both influencing the background noise – the higher the existing background noise level the lower the costs of an additional vehicle), and the population affected.

Both case studies represent locations outside urban areas, i.e. that local effects are of minor importance due to a lower population exposure close to roads and railway tracks. Still the case studies have a different character: The motorway from Basel to Karlsruhe passes settled areas with some distance, leading to very small local effects from noise and air pollution. The respective railway line crosses the most important towns and cities, leading to considerable shares of noise costs. Whereas the railway line from Strasburg to Neubrandenburg passes most settlements, the respective route for road transport crosses the towns and villages on the way. As a consequence this route was split into road sections within and outside built-up areas. The road sections outside built-up areas show a situation comparable to the motorway drive as regards the local effects. Still the regional impacts due to air pollution differ considerably due to the different geographical location. Furthermore, the driving patterns and resulting emission factors vary depending on the vehicle technology considered. In effect the costs per vehicle kilometre are much higher on the motorway compared to the road sections outside built-up areas.

So the first conclusion is that the geographical location of roads outside urban areas plays an important role for the costs. This is caused by potentially large differences in the costs due to long-range effects of air pollution.

The second conclusion is that 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 day, caused by different 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.

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. In contrast, 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, but are subject to uncertainty.

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