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UNITE

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marginal costs for Transport Efficiency

Marginal Costs Case Study 9D: Urban Road and Rail Case Studies Germany

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D.0 Executive Summary

Marginal environmental costs due to road and rail transport were assessed for specific routes in Berlin and Stuttgart. 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 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.

Following conclusions can be drawn from the results:

- It is not possible to derive one single value for the marginal costs of a certain vehicle type in urban areas. The cost categories have to be distinguished if we want to generalise values.
- Two components have to be considered for generalisation of air pollution costs: the local situation (covering at least population density and average wind speed) and the geographical location within Germany.
- Noise costs vary considerably between different times of the day, reflecting the higher disturbance effect of noise during night time and variations in background noise levels.

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 urban case studies in Germany is presented.

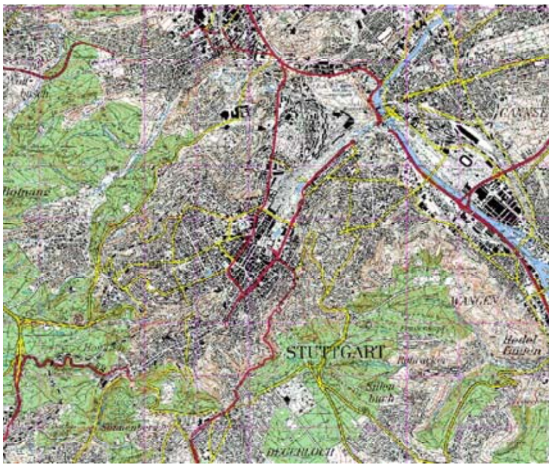
D.2 Case Study Description

Due to the close relationship of air pollution and noise costs with population density the highest environmental costs arise in urban areas. But as previous work (see Friedrich and Bickel 2001) has shown, differences within urban areas may be considerable, in particular when looking at the very local effects of noise. In this case study the variations of environmental costs and the driving parameters are analysed for route sections in Stuttgart and Berlin. For road transport main roads within densely built areas were studied. 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 tram, light rail and underground trains with electric traction.

D.2.1 Location

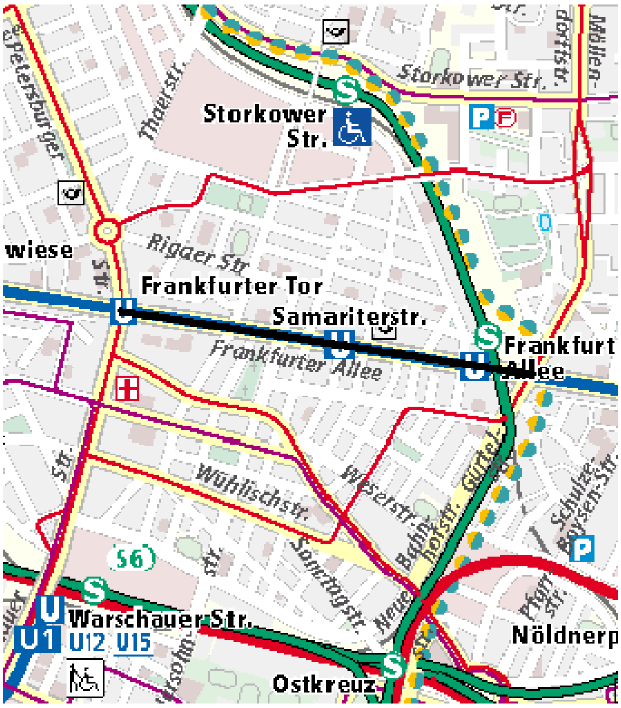
The city of Stuttgart, located in the south west of Germany, is characterised by a very densely populated city centre due to it's location in a steep basin surrounded by a large and heavily populated commuter belt. The city centre is surrounded on three sides by hills and towards the East limited by the river Neckar. Therefore it can only be reached by a limited number of access roads. The "Hohenheimer Straße" is the only major access road from the south and is labelled as federal road. The section analysed is built with 5-7 storey buildings at the kerbside. Uniformly built, the houses form a deep street canyon. The road carrying the traffic today was expanded to 4 lanes. The section assessed has a length of 440m.

Figure D-1 Geographical map of the Stuttgart city area



Berlin, Germany's capital and with 3.4 Mio inhabitants the biggest city, is located in the east of Germany. The route section analysed was the "Frankfurter Allee" between "Frankfurter Tor" and "Möllendorffstraße" (see Figure D-2). On this section, which has a length of 1.5 km, the road has three lanes in each direction. Furthermore underground transport and tram transport was considered. For reasons of simplification a tram line was assumed to run on the Frankfurter Allee. This assumption is mainly relevant for noise costs, as the same number of people exposed could be used as for road transport.

Figure D-2 Route section analysed for the Berlin case



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 due to diesel trains can be quantified with the same approach as road transport vehicles with internal combustion engine by making adjustments as necessary, e.g. emission height.

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 D-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 D-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 D-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 D-3:

$y = 99.7 - \alpha \cdot C_{O3}$

with y = relative yield change

α = sensitivity factors

C_{O3} = AOT 40 in ppmh

Table D-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 D-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 D-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 analysed in detail using digital images of the sites of the urban 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 D-5.

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

Table D-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 D-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 D-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 D-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

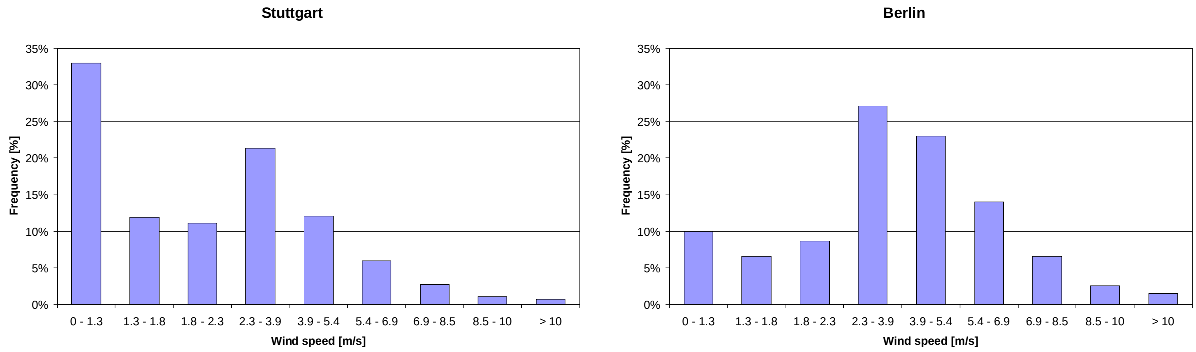
As the formation of secondary pollutants such as ozone or secondary particles depends heavily on the availability of precursors in the atmosphere, the EcoSense database provides a European wide emission inventory for SO_2, NO_x, NH_3, NMVOC, and particles as an input to air quality modelling. The emission 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 were used. It turned out, that differences in yearly average wind speed had great influence on the exposure estimates. Figure D-3 shows the high share of low wind speeds for the Stuttgart area compared to the higher wind speeds in Berlin.

Figure D-3 Frequency distribution of wind speed in the case-study areas (1981-1990 average from measurements (Deutscher Wetterdienst 1999)



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 in the city centre. Table D-9 shows the emission factors per vehicle kilometre. The same emission factors were used for Stuttgart and Berlin to facilitate comparison of the results.

Table D-9
Emission and fuel consumption factors in g/vkm used for road transport

Vehicle type	**Motor-cycle**	**Passenger Car**						**LGV**	**HGV**	**Urban 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.	0.0046	0.0046	0.0046	0.0600	0.0500	0.0100	0.07	0.36	0.25	0.26
SO_2	0.01	0.0214	0.0070	0.0067	0.0346	0.0341	0.0046	0.06	0.36	0.28	0.25
Source: UBA/BUWAL 1999											

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 D-10.

Table D-10
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 D-11
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 the respective operator in Berlin and Stuttgart as given in Table D-12.

Table D-12
Share of fuels in the electricity production of public transport operators

	Berlin	Stuttgart
Coal	74.0%	1.9%
Nuclear		89.6%
Oil	2.0%	0.3%
Natural gas	24.0%	1.6%
Hydro		2.1%
Other		4.5%
Total	100.0%	100.0%
Source:	Bewag (2002)	Schmid et al. (2001)

Table D-13 shows the data of the trains considered for public rail transport in Berlin and Stuttgart. 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 D-13
Technical data of trains for public rail transport

	Max. Capacity Persons	Energy use kWh/km	operation in
Tram	120	2.50	Berlin
Underground	328	5.35	Berlin
Light rail	190	5.12	Stuttgart
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 D-14 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 D-4 and Figure D-5.

Table D-14
Average annual daily traffic (AADT) used in the case studies

	Passenger Car	LGV	HGV	Motorcycle	Bus	Source
Stuttgart Hohenheimer Straße	14480	336	725	409	142	Wickert (2001)
Berlin Frankfurter Allee	50000	2000	2700	1200	600	Senatsverwaltung für Stadtentwicklung (2001), Wickert (2001)

Figure D-4 Stuttgart "Hohenheimer Straße": vehicles per hour for time periods day, evening, night (Source: own calculations based on Wickert (2001))

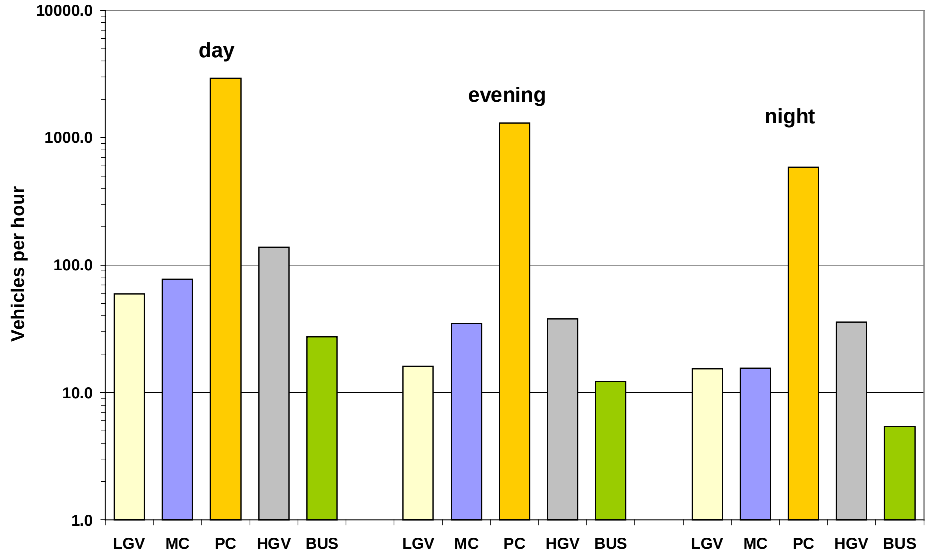


Figure D-5 Traffic density in Berlin, 'Frankfurter Allee' (vehicles per hour). (Source: own calculations based on Senatsverwaltung für Stadtentwicklung (2001) and Wickert (2001))

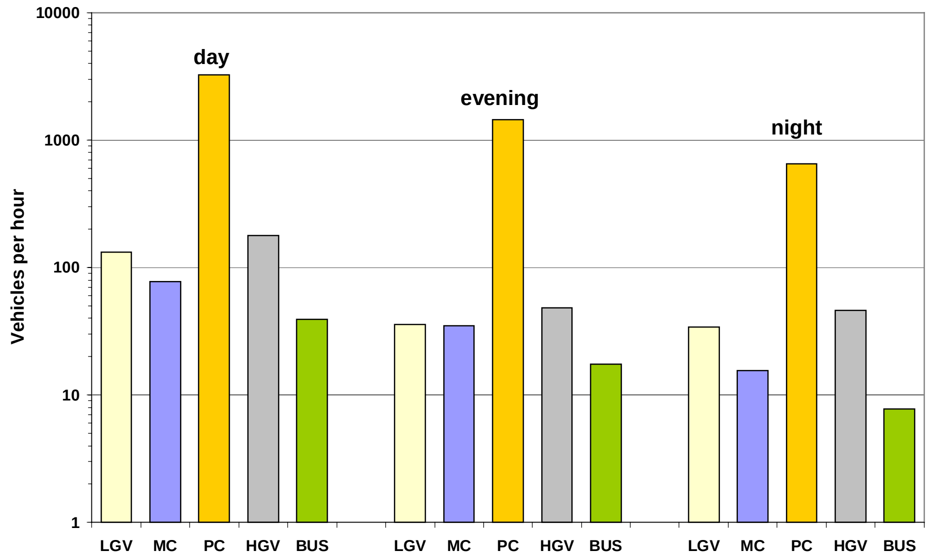


Table D-15
Number of trains per hour used for calculating marginal noise costs

	day	evening	Source
Stuttgart Hohenheimer Straße: lines U5, U6, U7	36	24	time table SSB
Berlin Frankfurter Allee: lines 20, 21	32	18	time table BVG

D.3 Results

Table D-16 presents the marginal costs due to airborne emissions per vehicle kilometre for the Stuttgart case. 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 light rail system runs on electricity, therefore costs from direct emissions (“vehicle use”) are zero. Total costs, both from air pollution and global warming are very low compared to the vehicles with internal combustion engines. This is caused by the high share of non-fossil electricity production, leading to very low emissions of air pollutants and greenhouse gases and associated costs. Less than 10% of the electricity used by the trains in Stuttgart is produced from fossil fuels (see Table D-12 above).

Table D-16
Marginal costs due to airborne emissions in EUR/100 vkm – Stuttgart

		air pollution			global warming			Total
		vehicle use	fuel/electr. prod.	total	veh. use	fuel/electr. Prod.	total	
Petrol	Motorcycle [a]	0.45	0.08	0.43	0.29	0.05	0.34	0.77
	Car Euro1	0.33	0.12	0.46	0.48	0.08	0.56	1.02
	Car Euro2	0.25	0.12	0.37	0.47	0.08	0.55	0.92
	Car Euro4	0.14	0.11	0.26	0.43	0.07	0.50	0.76
Diesel	Car Euro1	1.75	0.06	1.81	0.32	0.04	0.36	2.17
	Car Euro2	1.45	0.06	1.51	0.31	0.04	0.35	1.86
	Car Euro4	0.37	0.05	0.42	0.29	0.04	0.33	0.75
	LGV [b]	2.10	0.10	2.20	0.56	0.07	0.63	2.83
	HGV [b]	17.52	0.62	18.14	3.28	0.41	3.69	21.83
	Coach [b]	12.44	0.43	12.87	2.28	0.28	2.56	15.43
	Urban bus [b]	13.55	0.48	14.02	2.54	0.32	2.86	16.88
electric traction	Light Rail	0 [c]	2.51	2.51	0 [c]	0.72	0.72	3.23

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

For vehicles running on petrol, 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 (CO2, CH4 and N2O), above all from vehicle use. Diesel vehicles in total cause higher costs per vehicle kilometre than petrol vehicles with a comparable emission standard. This is mainly caused by the higher particle and NOx emissions of diesel engines. The shares of the different pollutants in the costs are illustrated in Figure D-6 for passenger cars complying with EURO2 standard. Furthermore it can be seen that the costs due to greenhouse gas emissions are higher for petrol vehicles.

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 D-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" can be neglected.

Figure D-6 Costs due to airborne emissions from vehicle use - Stuttgart case

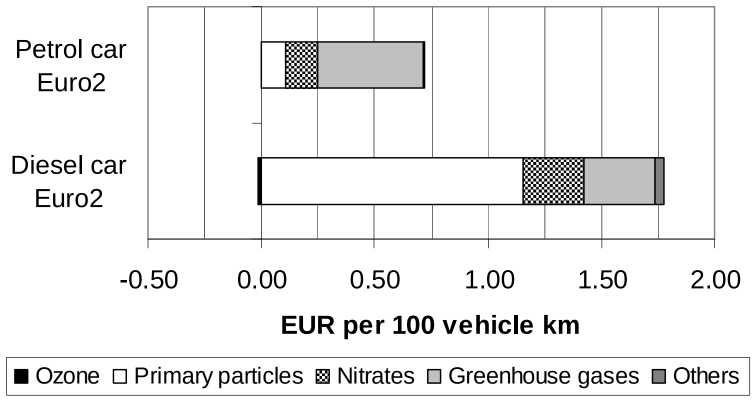


Figure D-7 shows the share of the different impacts in the air pollution costs (i.e. excluding the costs due to greenhouse gas emissions) from vehicle operation in Stuttgart. Mortality effects clearly dominate, followed by morbidity effects. Compared to the costs due to health risks, quantifiable costs due to material damages and crop losses are of minor importance. The higher relative share of these two categories, which are related to SO2 emissions, in the costs of the diesel car is caused by the higher share of SO_2 emissions in relation to the other pollutants.

Figure D-7 Split of air pollution costs (excl. greenhouse gases) - Stuttgart case

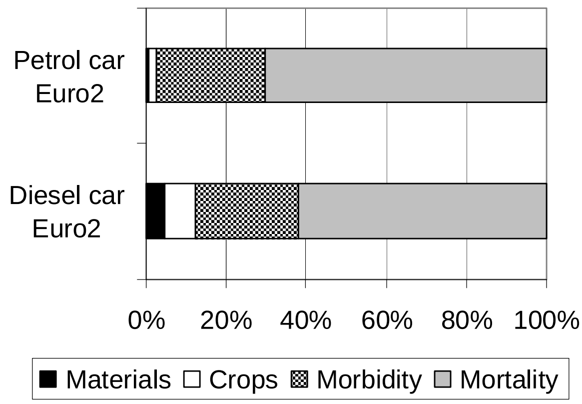


Table D-17 presents the results for the Berlin case. For road vehicles air pollution costs due to fuel production and global warming costs are the same for Berlin and Stuttgart, because for these cost categories the same damage factors apply and the emission and fuel consumption factors per vehicle kilometre are the same.

Table D-17
Marginal costs due to airborne emissions in EUR/100 vkm – Berlin

		air pollution			global warming			Total
		vehicle use	fuel/electr. prod.	total	veh. use	fuel/electr. prod.	total	
Petrol	Motorcycle [a]	0.41	0.08	0.42	0.29	0.05	0.34	0.77
	Car Euro1	0.21	0.12	0.33	0.48	0.08	0.56	0.89
	Car Euro2	0.15	0.12	0.27	0.47	0.08	0.55	0.82
	Car Euro4	0.08	0.11	0.19	0.43	0.07	0.50	0.69
Diesel	Car Euro1	0.89	0.06	0.95	0.32	0.04	0.36	1.30
	Car Euro2	0.73	0.06	0.79	0.31	0.04	0.35	1.14
	Car Euro4	0.20	0.05	0.26	0.29	0.04	0.33	0.59
	LGV [b]	1.07	0.10	1.17	0.56	0.07	0.63	1.79
	HGV [b]	10.19	0.62	10.81	3.28	0.41	3.69	14.50
	Coach [b]	7.24	0.43	7.67	2.28	0.28	2.56	10.23
	Urban bus [b]	8.03	0.48	8.51	2.54	0.32	2.86	11.37
electric traction	Underground	0 [c]	11.22	11.22	0 [c]	7.58	7.58	18.80
	Tram	0 [c]	5.24	5.24	0 [c]	3.54	3.54	8.79

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

Air pollution costs from vehicle use are considerably lower in Berlin compared to Stuttgart. These differences stem from two components: 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).

On the local scale costs are driven by average wind speed and population affected. The high share of low wind speeds for the Stuttgart area (see Figure D-3) leads to a pollutant exposure of the population which is more than a factor of two higher compared to the exposure in Berlin. Due to the higher wind speeds in Berlin the pollutants are dispersed much faster which leads to lower costs on the local scale.

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 (including Berlin) 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 (including Stuttgart). So in tendency regional scale costs are higher for emissions in Stuttgart than in Berlin. 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.

The role of fuel production costs depends on the emission standard a vehicle complies with and where it is driven. For the EURO4 petrol car the costs due to fuel production make a considerable share of the total costs due to air pollution. The costs due to electricity production for trains are much higher in Berlin than in Stuttgart, because electricity is produced from fossil fuels, mainly from coal (see Table D-12).

Table D-18 and Table D-19 present the marginal costs due to noise for the Stuttgart and Berlin cases. Costs are increasing from day to evening and night, reflecting the higher disturbance effect of noise during night time. In Berlin the average number of persons per road kilometre affected by noise is slightly higher than in Stuttgart. However, the costs are more than a factor of three lower due to the much higher number of vehicles and higher speeds on Frankfurter Allee leading to a higher background noise level. This is the case for light rail and tram as well. Noise costs due to underground trains are zero.

Table D-18
Marginal costs due to noise in EUR/100 vkm – Stuttgart

	day	evening	night
Motorcycle	9.00	12.00	27.50
Car	1.50	2.00	4.50
LGV	7.50	10.00	23.00
HGV	25.75	34.25	78.25
Urban bus / Coach	6.00	8.00	18.00
Light Rail	3.70	10.63	[a]
[a] no cost estimate available due to missing data			

Table D-19
Marginal costs due to noise in EUR/100 vkm – Berlin

	day	evening	night
Motorcycle	2.67	3.50	8.17
Car	0.47	0.62	1.45
LGV	2.33	3.00	6.83
HGV	7.67	10.17	23.33
Urban bus / Coach	1.83	2.33	5.33
Underground	0	0	0
Tram	1.00	3.11	[a]
[a] no cost estimate available due to missing data			

Figure D-8 and Figure D-9 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.

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. Costs due to airborne emissions loose in importance for cleaner vehicles.

Figure D-8 Sum of marginal costs for Stuttgart "Hohenheimer Straße" (* no night time noise for light rail)

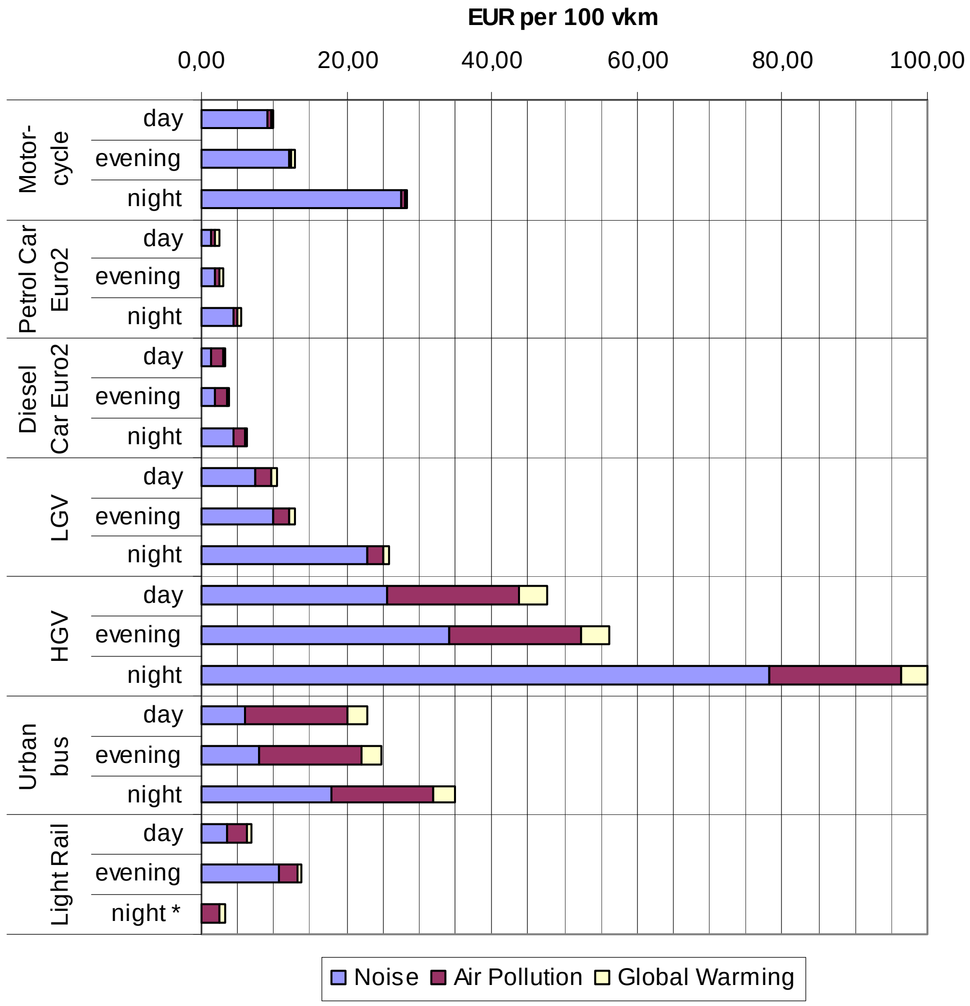
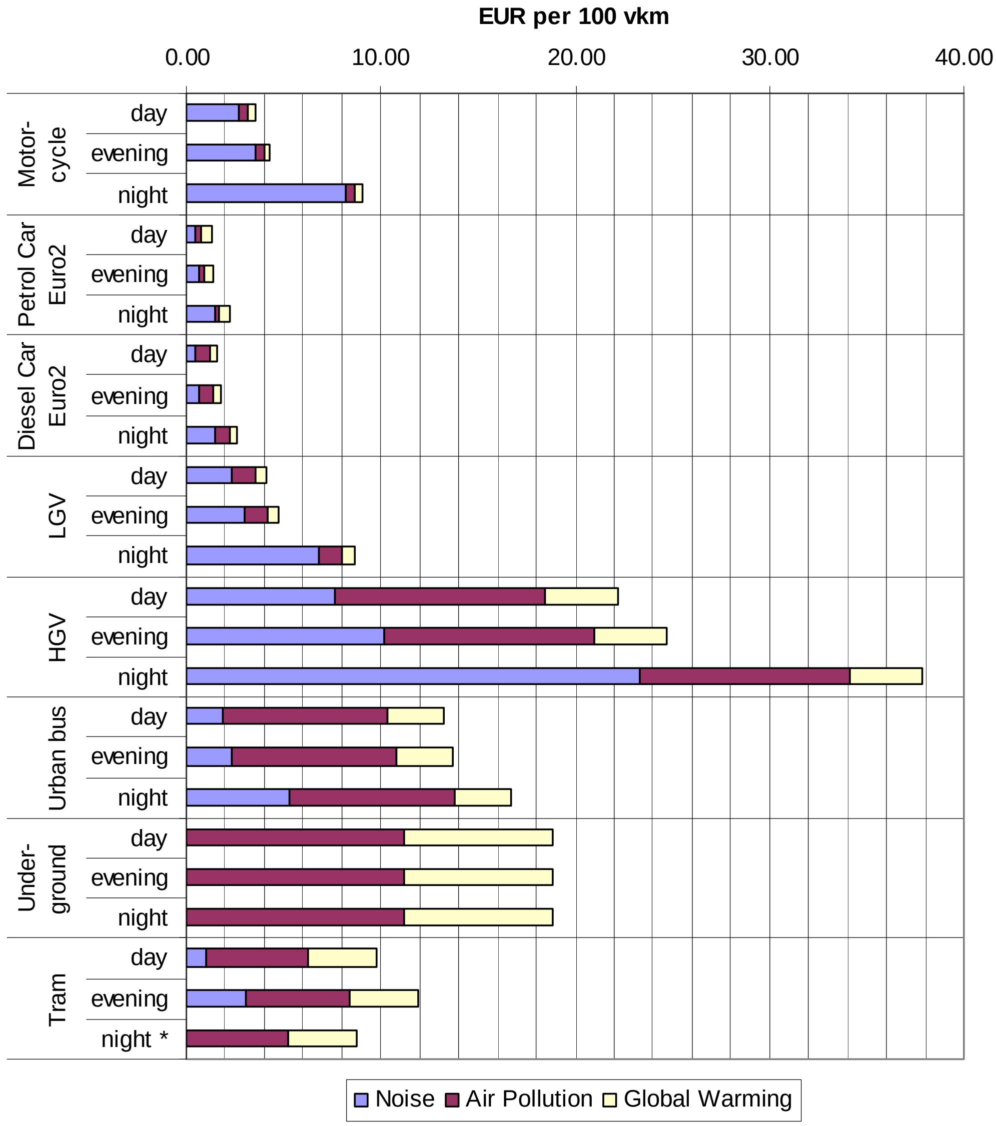


Figure D-9 Sum of marginal costs for Berlin "Frankfurter Allee" (* no night time noise for tram)



D.4 Discussion and conclusions

Marginal costs due to the emission of air pollutants, greenhouse gases and noise for route sections in the city centres of Stuttgart and Berlin 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 size of the population affected by long-range pollutant transport.

- Noise costs are mainly determined by the time of day (higher disturbance effect 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.

For air pollution from road vehicles the differences between Stuttgart and Berlin are caused by the average wind speed, which is much higher in Berlin, and the geographical location within Germany. Both effects leading 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.

It is due to the specific properties of the selected routes that marginal noise costs are higher for the road analysed in Stuttgart than for that in Berlin and that the relation is the same as for air pollution costs. The road selected in Berlin has a much higher number of vehicles with higher speeds and thus a higher background noise level than the route considered in Stuttgart. As with similar population densities along the road the background noise level is the main determinant of the cost of an additional vehicle, the same vehicle causes lower marginal costs on a busier road.

So the first conclusion is that it is not possible to derive one single value for the marginal costs of a certain vehicle type in urban areas. The explanations above indicate that we have to distinguish the cost categories if we want to generalise values.

The second conclusion we can draw from the results is that two components have to be considered for generalisation of air pollution costs: the local situation (covering at least population density and average wind speed) and the geographical location within Germany.

The third conclusion which can be derived is that noise costs vary considerably between different times of the day, caused by varying disturbance effects and variations in background noise levels. As an additional vehicle causes less costs on a noisy road than on a quiet road, the use of marginal noise costs for pricing leads to a bundling of traffic in areas, which are noisy already.

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.

D.5 References

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