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Authors:
I. Mayeres, S. Proost, D. Vandercruyssen (CES – K.U.Leuven)
L. De Nocker, L. Int Panis, G. Wouters (VITO)
B. De Borger (UFSIA)

CES, K.U.Leuven – co-ordinator – Promotor: Prof. Dr. S. Proost – contract MD/DD/008
VITO – Promotor: Ir. G. Wouters – contract MD/DD/009
UFSIA – Promotor: Prof. Dr. B. De Borger – contract MD/DD/010

The research teams

Center for Economic Studies, K.U.Leuven (co-ordinator)

Promotor: Prof. Dr. Stef Proost
Researchers: - Inge Mayeres [1/1/1997 – 31/12/1997, 1/4/1998 – 30/4/1998, 1/7/1998 – 31/3/1999, 1/8/1999 – 30/9/1999]
- Dimitri Vandercruyssen [1/10/1999 – 31/3/2000 (50%), 1/4/2000 – 30/11/2000 (100%)]

Contact:
Dr. Inge Mayeres
CES – K.U.Leuven
Naamsestraat 69, 3000 Leuven, Belgium
Tel: +32 (016) 32 66 35, Fax: +32 (016) 32 69 10
E-mail: Inge.Mayeres@econ.kuleuven.ac.be

VITO

Promotor: Ir. Guido Wouters
Researchers (the persons with an * have got a major involvement in the project):
- Leo De Nocker, 1997 – 2000, project leader *
- Dr. Luc Int Panis, 1998 – 2000, implementation of assessment models*
- Ir. Ina De Vlieger, 1997 – 2000, emission and traffic data *
- Ir. S. Vergote, 1997 – 1998, implementation of assessment models, LCA data.* (replaced by Dr. L. Int Panis in 1998)
- Ir. Luc Vinckx, 1997 – 1998, emission data
- C. Spirinckx, 1998 – 1999, inputs on LCA analysis
- R. Torfs, 1998 – 2000, inputs on LCA analysis and impact assessment

Contact:
Leo De Nocker
VITO
Boeretang 200, 2400 Mol, Belgium
Tel: +32 (014) 33 55 11, Fax: +32 (014) 33 55 99
E-mail: Denockel@vito.be

UFSIA

Promotor: Prof. Dr. Bruno De Borger
Researchers: Christophe Courcelle (1/10/1997 – 15/4/1998)
Tom Schatteman (1/5/1999 – 31/3/2000)

Contact:
Prof. Dr. B. De Borger
UFSIA
Prinsstraat 13, 2000 Antwerpen, Belgium
Tel: +32 (03) 220 40 57, Fax: +32 (03) 220 47 99
E-mail: Bruno.DeBorger@ufsia.ac.be

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1. INTRODUCTION

When consumers and producers decide whether to make a trip, by which mode and at what time, they evaluate the available alternatives on the basis of the costs and benefits of an extra trip for themselves. These are the so-called marginal private costs and benefits. The term "marginal" refers to the change in total costs and benefits due to an extra trip. The marginal private costs include the resource costs (for example, the fuel costs, the vehicle costs and the insurance premium), the taxes, the own time costs and the costs associated with the exposure to the accident risk. However, each trip also causes costs for the other transport users and for society in general. The additional transport users only partly take into account these costs in their decision process, via the taxes and the insurance premia they pay. The costs that are not taken into account are called the marginal external costs. Because of these, the traffic flow resulting from the decisions of the households and firms is larger than what is socially optimal. Moreover, the spread of trips over time is not optimal: too many trips take place in the peak period. The share of the various transport modes and the type of vehicles used is also suboptimal.

The policy maker can make use of various instruments to remedy this situation. Three categories of instruments can be distinguished: economic instruments, command-and-control measures and changes in the infrastructure. Information about the level and structure of the marginal external costs is a crucial input in the design of the policies for tackling the transport externalities.

The project calculates the marginal external costs of transport use in Belgium. This report discusses the findings for three main categories of external costs: environmental costs, accidents and congestion costs. Road damage externalities, which are caused mainly by trucks, are not considered here – with the exception of the air pollution costs related to road maintenance. The environmental costs were analysed by the VITO team, the CES – K.U.Leuven team was responsible for the accident costs and the UFSIA team studied the congestion costs.

The environmental costs

The marginal environmental costs probably are the best understood category of external costs. The project has led to a detailed and transparent set of estimates for Belgium. The study covers all major current and future (up to 2005) transportation modes, fuels and technologies for passenger and freight transport. It is based on a detailed inventory of emissions following the state of the art in life cycle analysis and emission models of transport activities. The analysis includes emissions related to the use phase of the transportation means, those related to the supply of transportation fuels and the construction of vehicles and, finally, those related to the maintenance of infrastructure.

The environmental damages from these emissions are assessed in a detailed bottom up assessment following the damage function approach of the European ExternE method. These damages mainly cover air pollution impacts on human health, crops, materials and global warming effects. Ecological impacts and human health impacts from noise are not yet included in the analysis given the large uncertainties that are still surrounding these impacts and their monetary valuation.

The accident costs

The analysis of the marginal external accident costs still raises many conceptual difficulties. The research consists of two parts. The first part aims to give a thorough theoretical background for determining these costs. In a first step this is done with the help of a simple theoretical model which makes abstraction of the impact of liability rules and insurance on the road users' behaviour. In a second step the role of liability rules and insurance is considered explicitly.

The second part aims to determine the monetary value of one of the most important components of the accident costs, namely the health impacts. Surveys were conducted in Flanders in order to derive the value of a statistical life/injury. Three survey methods are used: contingent valuation, a combination of contingent valuation and standard gamble, and a choice experiment. The data will be used to compare these three methods in terms of their performance in producing a reliable monetary valuation of a statistical life. The results of this exercise should contribute to the current discussion in the literature about the best survey technique to use in this field.

The congestion costs

The congestion costs are another category of costs for which conceptual difficulties still exist. For this category there is also a large gap between the scientific basis and the acceptance by the policy makers. The project has extended the existing methodology to take into account three aspects: the dynamic adjustment of departure times, the treatment of uncertainty and the provision of information to the transport users.

The structure of this report is as follows. Section 2 discusses the methodology and the main results for the environmental costs. The analysis of the accident and congestion costs is presented in respectively Section 0 and Section 4. Section 5 briefly presents a number of recent publications in which marginal cost information is used to assess various transport policies. Section 6 concludes and discusses some directions for future research.

2. THE MARGINAL EXTERNAL ENVIRONMENTAL COSTS

L. De Nocker, L. Int Panis (VITO)

2.1. Introduction

If we drive our car to a nearby city, we have probably thought about how much time it will take us and what it would cost us. However, most people don't take into account the impacts of their journey on public health, the historical buildings in the city centre or on forests 1000 km downwind. These damages to man and the environment are called external costs, as they are not reflected in market prices.

The evaluation of air pollution impacts is based on the accounting framework of the European ExternE project. (see further). Earlier estimates for Belgium using ExternE data were based on extrapolation of case studies for neighbouring countries (Mayeres et al., 1996). Other estimates of transport externalities for Belgium were based on less detailed or on less up-to date methodologies (e.g., Pearce, 1996). The results of this exercise can provide the basic data for analysing a myriad of questions related to transport and environmental policies.

This study covers all major current and future (up to 2005) transportation modes (road, railway and waterway), fuels and technologies for both passenger and goods transport. Although the original aim of the study was to cover all environmental impacts we were not successful in fully quantifying and monetizing all impacts. This issue will be discussed further. As a summary we can state that the report especially covers air pollution impacts on public health, crops, materials and global warming impacts. The major missing categories are ecological impacts and public health impacts from noise.
The study covers the use phase of these technologies in detail, and also looks in a more general way to the impacts of the full life cycle (LCA analysis) of the provision of fuels, vehicles and infrastructure. The main focus of the report is on the analysis and comparison of the use phase of current and future technologies for road transport, and on the full life cycle analysis and comparison for different transportation modes.

In the next paragraphs, we first describe the ExternE methodology and its implementation for this study. Special attention is given to the calculation of emission factors and population exposure. In the results section we first discuss the impacts of all major pollutants and the importance of location effects. We focus briefly on the importance of some car and traffic related factors before discussing marginal and aggregated external costs for selected vehicle categories. In the second part of the results section we discuss the Life Cycle Analysis of fuel production, vehicle production and provision of infrastructure. This is especially detailed for inland shipping for which very few data were available before. To conclude the report we compare current technologies with some alternative future technologies and with some alternative (existing) transport modes.

2.2. Methodology

In this section we discuss the general methodology. Specific details of the methodology related to aggregation or life cycle analysis are dealt with in these chapters.

2.2.1. General introduction to the ExternE-methodology

The evaluation of environmental impacts is based on the accounting framework of the European ExternE project. This framework was developed to account the externalities of electricity generation (1991-1997; EC, 1995; EC, 1998a; EC, 2000, in prep.). Since 1996, it has been extended to account for energy related impacts of transport.
The ExternE accounting framework is based on the 'impact pathway' methodology which represents the long way from a 'burden' to an 'impact' and an external cost (Figure 1).

Impacts on human health and the environment are quantified in 5 consecutive steps: determination of emission factors, dispersion simulation, assessment of exposure, impact assessment with dose-response functions, and monetary valuation. This bottom-up approach integrates the state of the art knowledge in different scientific disciplines in a common and coherent framework. We will only discuss some methodological issues in more detail. Detailed information concerning the assessment of the impacts is found in the ExternE reports. (EC, 1998 and EC, 2000). In the next paragraphs we will discuss these 5 steps in more detail.

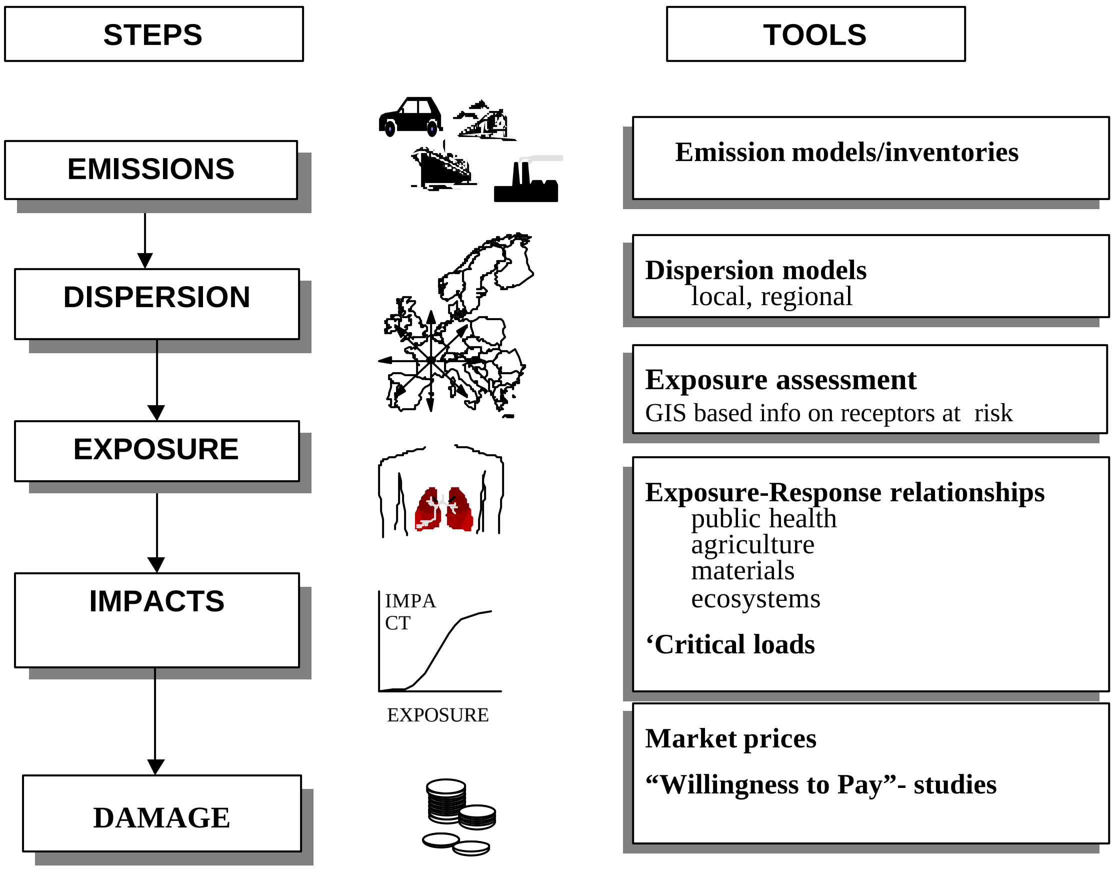


Figure 1: The impact pathway methodology

2.2.2. Models used

In the project, several models have been developed, used or integrated to make these implement these impact pathways. A major distinction is made between the use phase and the non-use phases (LCA impacts). For the use phase, an integrated model - EcoSense – has been implemented to calculate the environmental impacts and external costs of a single car on a specific trajectory. It includes emissions models, dispersion models, receptor at risk data dose-response functions, critical loads and data for monetary valuation. It has been implemented in detail in a GIS environment for Belgium. For the non-use phase, more general models representing average conditions have been used. All these results have been integrated in the ExTC software (External costs of transport). This model allows for aggregation, uncertainty analysis and scenario analysis up to 2010. An overview of all the models used is given in Figure 2.

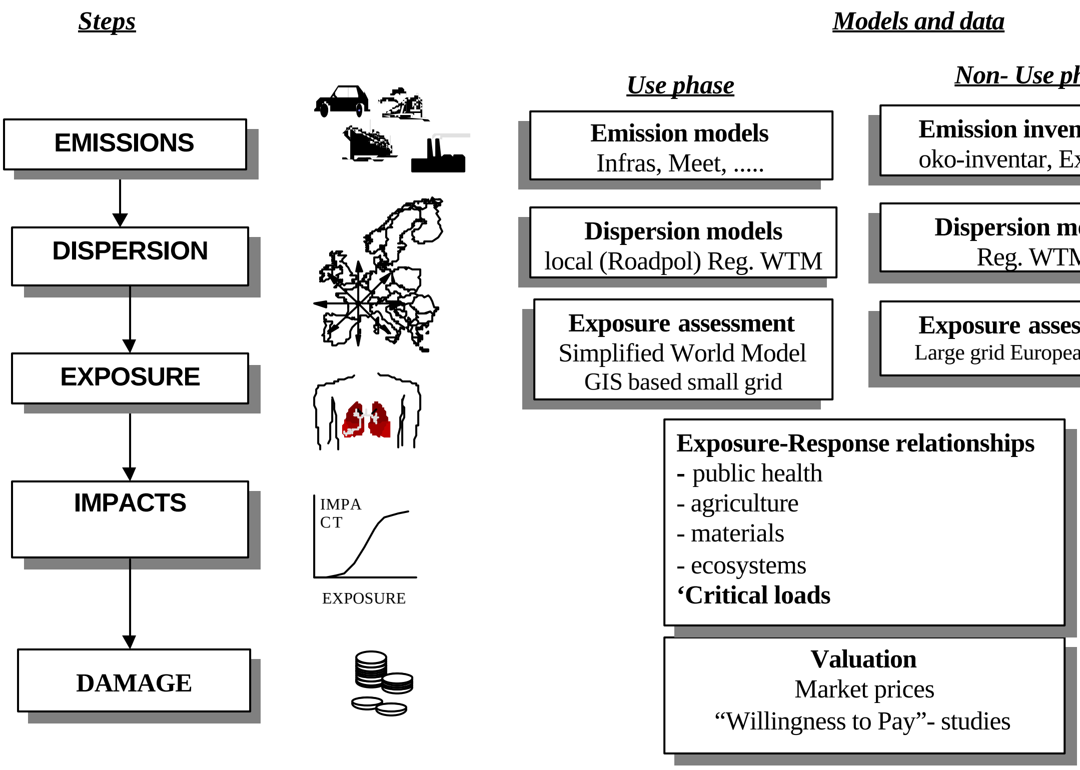


Figure 2: Models used in the different sections of the methodology

2.2.3. Emissions

In this paragraph we concentrate on the emissions from the use phase of transport. Emissions in other stages of fuel, vehicle and infrastructure production and maintenance are dealt with in the relevant chapter.

2.2.3.1. Reference Technologies and category split

For passenger cars, petrol and diesel cars make up more than 99% of the Belgian car fleet whereas LPG cars only take up a marginal portion of the Belgian car fleet (less than 1%).

According to European legislation, we distinguish between EURO 0, EURO 1, EURO 2 and EURO 3 types of vehicles. For the extrapolations up to 2010, we have also defined EURO 4 and EURO 5 vehicles which comply with future European emission standards

2.2.3.2. Variety and variability of emissions

Exhaust emissions of road transport are complex in nature due to both the **variety** and the **variability** of their composition. Exhaust emissions are a mixture of hundreds of different chemical substances. The pollutants, for which emission factors are gathered, are prescribed by the priority impacts causing high externalities, which were determined in previous phases of the ExternE project. This limits the number of pollutants to 15 'prime' pollutants, which can be subdivided in 3 main categories, namely:

- greenhouse gasses : CO_2, CH_4 and N_2O
- 'classical' pollutants : CO, NO_x, SO_2, VOCS, PM and NH_3
- and 'new' pollutants : Pb, benzene, ethene, formaldehyde, benzo(a)pyreen 1,3 butadiene

Some of these 'pollutants' (e.g. VOC, PM) are actually a mixture of different chemical substances. Emission factors for all these chemicals were calculated, but NH_3, Pb, ethene and formaldehyde were subsequently omitted for lack of reliable dose-effect relationships.

Accurate quantitative data about actual motor vehicle emissions for the different pollutants is made a complex issue due to the multitude of parameters involved. For example, some emission factors of a specific vehicle can vary by more than an order of magnitude due to differences in operational circumstances, e.g. road type, traffic density and speed. Furthermore, a range of technical specifications is related to emissions characteristics (e.g. weight, engine efficiency, gearbox, engine/vehicle/gearbox matching, motor condition and ambient temperature, manufacturing year and mileage).

2.2.3.3. Assessment of emission factor models in Europe

In Europe, a number of national or European research projects are devoted to the determination of emission factors of road transport vehicles. A common feature is that they are based on (extensive) laboratory measurement campaigns and that all attempt to a certain extent to incorporate some of the parameters described above to account for the variability of emission factors. Generally, the models can be divided in 2 main categories:

1. models based on **average speed** emission factors. e.g. The Computer Programme to Calculate Emissions from Road Traffic (COPERT) MEET, 1999

2. more complex models based on emission matrices, i.e. emissions as a function of **instantaneous speed** and **acceleration,** e.g. The Handbook of Emission Factors, called here the INFRAS model

Our initial assessment focused on 3 or more models. Finally two sets of emissions factors were derived based on the MEET and INFRAS models respectively. Both were complemented with data from different sources and publications where the original models were incomplete. Initially, some externality calculations for passenger cars were based on "INFRAS" emission factors. Later, we have systematically used emission factors from the

MEET project. This has the significant advantage that most results from this project are directly comparable to those of the European ExternE Transport project and all publications and reports that were derived from it. Numbers in this report are mainly based on MEET. A comparison between MEET and INFRAS can be found in Int Panis (2000).

As LPG is not included in the INFRAS model and only partly in MEET, emission factors were based on other information sources (mainly TNO). Future technologies (CNG, biofuels, fuel cells, electric, hybrid vehicles) are still in a developmental stage. The assumptions and choices made for these vehicles are discussed in paragraph 2.3.4.1.

2.2.4. Atmospheric dispersion

2.2.4.1. Introduction

Once emitted from the tailpipe of a car or truck, pollutants are dispersed in the atmosphere. For this study we have used three types of atmospheric dispersion models. For the classical pollutants separate models were used for the local (up to 50 km from the source) and regional (Europe wide) scale. In addition a third model was used to construct a matrix that describes the relation between emissions of NOx and VOC in one gridcell and the (annual average) ozone concentration in the rest of the (EMEP) grid.

2.2.4.2. Local modelling

For the local modelling Roadpol (a bigaussian model) has been used as it is the ExternE standard. In the most recent software tools, Roadpol is used in combination with the MapInfo GIS software. This has allowed us to use real trajectories (and not simple lines or points) as sources of the emissions. In addition we were able to use increasingly more detailed receptor grids. Because location is so much more important for transport than it is for energy conversion, ExternE members now use 250x250m grids close to the trajectory (see 2.2.5 exposure assessment)
The local dispersion modelling also takes into account the differences between rural and urban areas related to dispersion parameters and meteorological data. In particular lower ground level wind speeds and higher dispersion coefficients in urban areas. Different meteofiles were attributed to rural and urban areas according to the following practical definition: urban areas have a built up area of at least 25% or a population density larger than 1000 inhabitants/sqr km (over the whole community). In addition, the community must be part of an agglomeration of at least 75000 inhabitants. Smaller or less dense agglomerations are though to have a limited effect on the meteorological parameters.
No correction was made for canyon effects. Since this necessitates detailed information about the structure (height width and orientation) in each section of the trajectory which is clearly beyond the scope of this project.

2.2.4.3. Regional modelling

The local modelling looks in detail at locations up to 20 km from the line source but uses a simple bigaussian algorithm. For atmospheric transport over longer ranges it can be assumed that the pollutants have been vertically mixed in the mixing layer of the atmosphere. In addition we can no longer neglect chemical reactions in the atmosphere. Therefore we have to use a model that also takes into account the formation of secondary pollutants.

The regional model used is called Windrose Trajectory Model (WTM) which uses the Harwel trajectory model approach for the modelling of regional atmospherical dispersion. Simple chemistry modules account for the transformation of NOx and SOx into acid species. Comparison with a validated model (EMEP) indicates that the results of the regional model agree to within 20%.

2.2.4.4. Ozone modelling

The modelling of ozone concentrations that result from emissions of VOC and NOx proved to be a very challenging task. The main problem arises because of strong non-linearities between emissions (esp. of NOx) and ozone formation. In addition it is possible that the local effect of NOx emissions is an ozone reduction, while the same NOx emissions cause ozone formation further downwind of the source. All this depends on the NOx and VOC concentrations and their ratio in different locations.
To come up with a workable model that provides annual average concentrations over the whole of Europe, ExternE has taken the following approach. The EMEP Lagrangian model has been used to calculate the effect (on ozone) of reducing NOx and VOC emissions in each of the European countries in all the gridcells. In this way a country-to-grid matrix was constructed which is incorporated in the EcoSense model.[1] This simplified model is then used to estimate the effect of changes in emissions at one location on the ozone statistics (AOT60 and AOT40) in all gridcells.

2.2.5. Exposure assessment

2.2.5.1. Trajectories

Environmental impacts of transport are site specific for two reasons : location affects traffic conditions and thus the emissions, and secondly, the number of receptors at risk (people, crops, vulnerable habitats,….) is also site specific. No two locations in Belgium are identical. If we want to summarise the impacts of traffic on the immediate surroundings we need to select typical trajectories. Only the results for these typical trajectories are discussed in this report and have been included in Appendix 3.

[1] EcoSense is the general name of the model used to calculate the environmental impacts and external costs. It includes emissions models, dispersion models, receptor at risk data dose-respones functions, critical loads and data for monetary valuation.

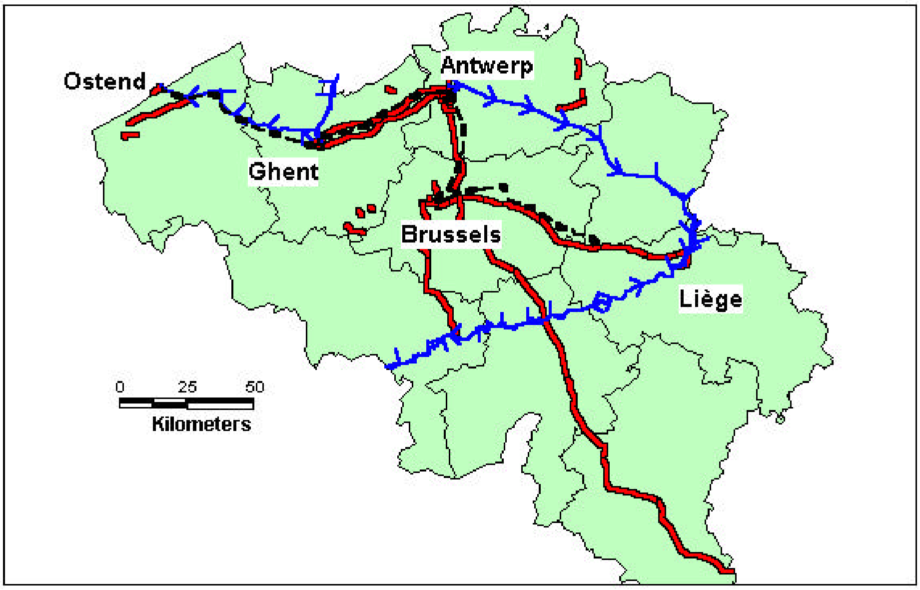


Figure 3: Trajectories of roads (===), railways (- - -) and waterways (->-->-)

Three "typical" trajectories: one urban, one highway and one rural were selected from a screening of about 30 initial trajectories (indicated in red in Figure 3), located throughout the country. In each of the three categories, we have selected the trajectory that yielded a central value for external costs per vehicle km. This does not necessarily mean that these trajectories are especially important in terms of the total annual mileage. Although the results for the selected trajectories are typical, there remains some geographical variation in each category. Urban, rural and highway trajectories can be found for which the externalities are between half and double the value that was calculated for the chosen trajectory.
Since the new EcoSense software has GIS capabilities, we aimed at a detailed modelling exercise, with high precision for roads and population density. The spatial resolution at which trajectories were digitised roughly equals the resolution of the 1/10.000 NGI topographical maps which is much higher than the resolution of the population datasets.

2.2.5.1.a. Urban trajectory

This trajectory was chosen to represent a typical commuting trip between the centre of a large city and its outskirts. To ensure that the result could be compared with urban trajectories in other European countries, only trajectories in Antwerp and Brussels were screened. Finally a trajectory was chosen in Brussels between the Berlaymont building of the European Community and the city ring to the north of the city. This trajectory is 10 kilometres long and crosses some of the most densely populated areas as well as business and industrial districts with lower population densities.
Population densities are highest between 2 and 5 kilometres from the trajectory, highlighting the lower population in Brussels inner centre. Densities remain higher than 3000 inh/km2 (a typical value for built-up areas in Belgium) over several kilometres (**Figure 4**).

2.2.5.1.b. Rural trajectory

Rural areas in Belgium can hardly be compared with rural locations elsewhere. Because of the high average population density and the poor land-use planning, typical rural sites still have relatively high population densities. To look for places with an extremely low population density (e.g. in the Ardennes) would not yield a ‘typical’ trajectory. In addition we felt that it was necessary to include only trajectories within or between villages because these are thought to be more relevant for aggregation. Of course this increases even more the average population density around the chosen roads. Typical population densities within (rural) villages or (urban) cities in Belgium are nearly always around 3000 inh/km2.
Finally a trajectory was chosen in Sint-Gillis-Waas a small village in the north of Flanders between the cities of Ghent and Antwerp. The trajectory is 2705 meters long and connects the centre of the village to the nearby national road.
Population densities drop of to low numbers even at a few 100 meters from the trajectory because houses are concentrated along the main roads (ribbon building). Densities rise again to the Flemish average at a distance of only 3 or 4 kilometres, the typical distance between villages in this region (**Figure 4**).

2.2.5.1.c. Highway trajectory

The selection of a typical highway trajectory proved to be just as difficult as the selection of a rural trajectory. Clearly most highway traffic can be found on the highways towards Antwerp and Brussels. The highways between Antwerp and Brussels cross some of the most densely populated regions of the country. The highways from the south towards Brussels cross some regions with much lower population densities. Therefore it was decided to take the E17 highway between Ghent and Antwerp, in the north of Flanders, as the reference trajectory in this category. Because of its proximity to the rural reference trajectory, this choice makes comparisons easier. Compared to the rural trajectory, densities are lowest close to the road, but these rise quickly because of the proximity of several small cities.

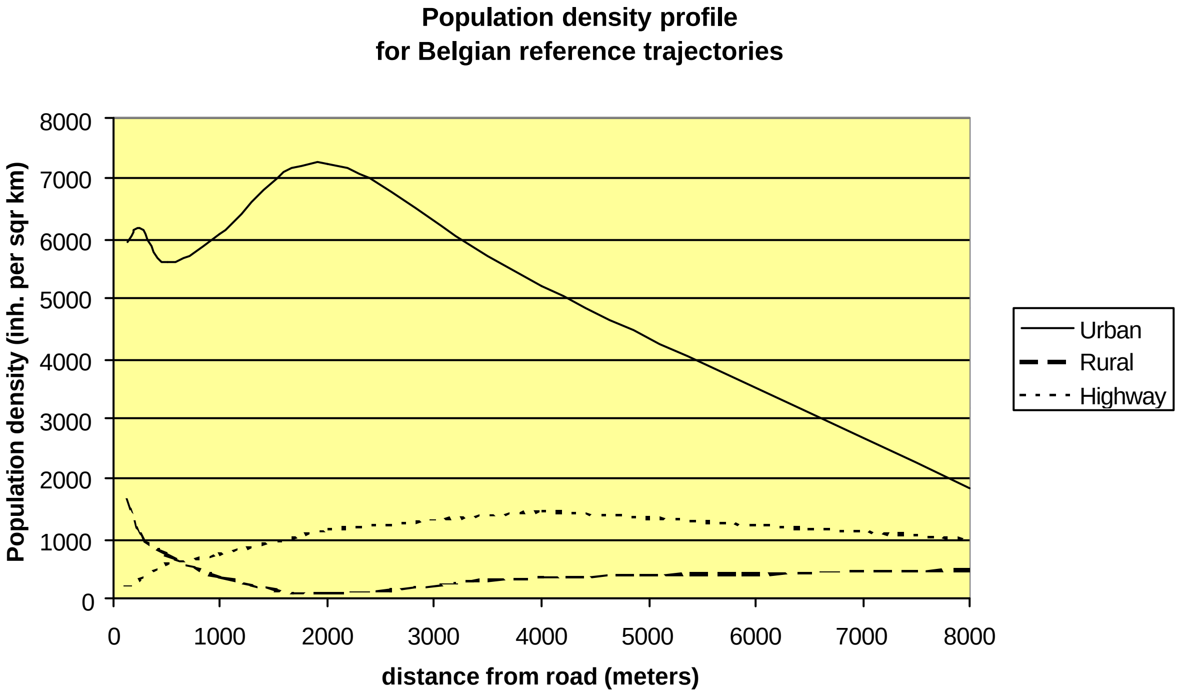


Figure 4: Population density around selected trajectories

2.2.5.2. Population exposure

Earlier studies of externalities have clearly shown that impacts on human health dominate environmental external costs. Vito has therefore spent considerable time constructing detailed population maps that allocate population to the housing districts within each community. To that purpose, a map with the administrative borders of all Belgian communities was updated with censusdata from different (official) sources for each of the three regions. Initially data for the year 1995 (Flanders and Brussels) or 1994 (Wallonia) were used. Recently we have completed a 2000 update which will be the standard for future calculations.
To match the resolution of the EcoSense grid and the receptor data, the population of each community was assigned to the housing districts within that community. For this calculation, a number of applicable categories were selected from the official land-use planning maps. In this way we neglect the houses that lay outside of the designated dwelling areas, but this error is obviously much smaller than assigning a uniform population distribution to the entire surface area of each municipality. Finally, as the fringes of the local grid often cross the Belgian border, some data for The Netherlands, Luxembourg, Rheinland-Pfalz and Nordrhein-Westfalen were added.

For each calculation a grid surrounding the trajectory was constructed. The gridsize is variable and depends on the distance to the trajectory. Gridcells that are located closer to the road become increasingly smaller. Bordering the road, we have used 250x250m gridcells. This is demonstrated for one trajectory in Figure 5. In this way we increase accuracy at the places were pollutant concentrations are highest while keeping the calculation manageable.

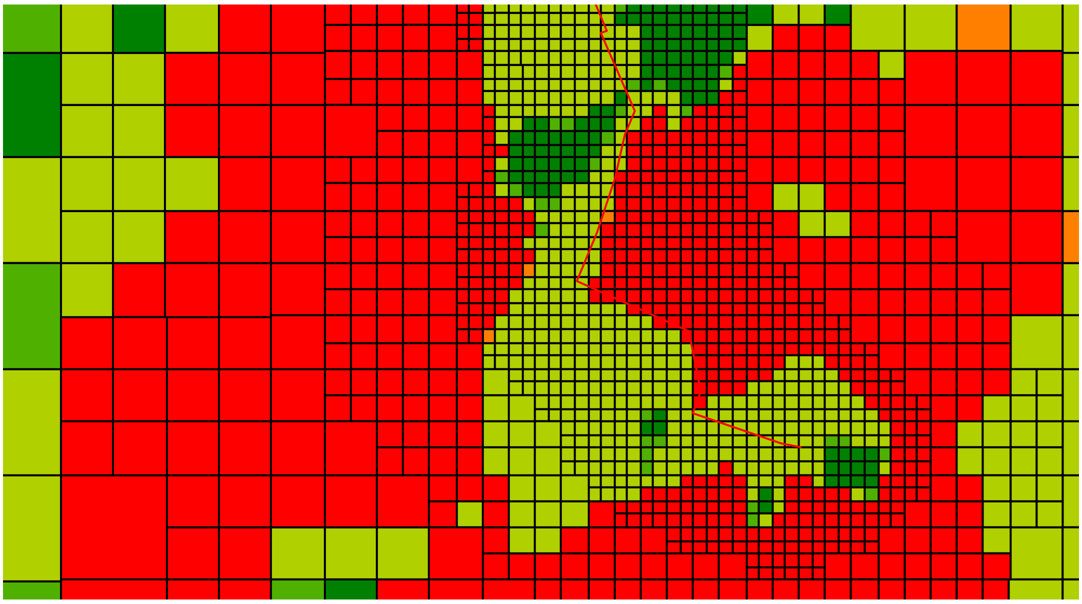


Figure 5: Example showing the conversion of population density in Brussels into variable gridcell along a trajectory in the centre of Brussels.

2.2.6. Exposure response functions

2.2.6.1. Public health

2.2.6.1.a. Introduction

In this and in other studies of externalities, we are concerned with the health effects of complex mixtures of air pollution. These mixtures are derived both from electricity production, transport as from other sources. They vary by location, technology, time and many other factors. It was and is impossible to evaluate the health effects of all these mixtures directly in human (toxicological) studies. The approach in ExternE has been to construct a representation or a model of the health effects of these complex mixtures.
Based on epidemiological evidence, experts have selected the key pollutants of the mixture which were believed to be adversely related to health. Based on current understanding (primary and secondary) particles and ozone were considered to be the main drivers of the pollution mixture. A set of E-R functions which was as comprehensive as possible for particles and for ozone was constructed. Effects of these two pollutants were considered to be additive. In recent ExternE work some SO2 functions are also included. Like any model the ExternE approach is a simplification of reality, but epidemiological data does provide a way to estimate some health impacts.
For some pathways, notably particles and acute mortality, or acute hospital admissions, the epidemiological data are very reliable. Some experts have described the strength of evidence as 'strong evidence of a weak effect'. For acute mortality and particles there are upwards of 50, possibly 100, well-conducted studies of different locations and/or years. These include two major multi-city studies, APHEA in Europe and NMMAPS in the USA, both designed with a view to meta-analysis. Astonishingly, both meta-analyses give very similar results: an increase of about 0.5% in daily mortality per 10 µg/m³ increase in PM10. There are however differences between cities which remain largely unexplained.
There is also very strong evidence, from numerous studies, linking daily ozone with acute mortality and respiratory hospital admissions. The evidence linking other pollutants with

mortality and hospital admissions, and linking any pollutant with other endpoints, is less strong, and indeed differs according to pathway.
Very recently some high profile publications have shown that general scientific acceptance of the validity of these functions has grown since they were first used by ExternE (Künzli et al., 2000; Samet et al., 2000).

The set of E-R function used for this study is based on ExternE and thus quite extensive including a number of minor health end-point next to mortality. Nevertheless we must stress that the impact pathways used here are broadly similar to those of other well-established impact assessments, both in the US and in Europe although the detailed functions may differ. The most striking differences (often cited) are with the recommendations of the UK expert group COMEAP (1998) . The main reason for the difference is that the purpose and criteria (terms of reference) are very different.
For a quantification project such as ExternE, the default contribution of any pathway is zero, i.e. if no E-R function is proposed. The question then arises: does the epidemiological and other evidence allow us to make a better estimate than zero; and if so, what is the most appropriate E-R function for this, and how reliable is it? Thus, though evidence on restricted activity days (RADs) comes from a single series of papers, and on its own might not be considered convincing, it would be a mistake to quantify this endpoint as zero. (It is strongly established that ambient PM affects more severe endpoints such as mortality and hospital admissions, and less severe ones such as symptoms in asthmatics and lung function in the general population; so it is very unlikely that there is no impact on RADs.)
COMEAP however had a more limited objective. It sought to quantify only where impacts could be estimated 'with reasonable confidence'; and this criterion was applied quite strictly, e.g. for each pathway in isolation from others. Hence, it proposed fewer E-R functions than ExternE and other impact projects.

Summarising, our goal is to quantify the total health impact from a mix of emissions. The purpose of a complete set of functions is to give the best average guess in total. For this reason we have opted for the ExternE approach which had the same goal. In the next two paragraphs we briefly discuss the most important assumptions and problems associated with particles and ozone. For the complete set of E-R functions the reader is referred to the methodology chapters of ExternE.

2.2.6.1.b. E-R functions related to PM

With respect to mortality, ExternE has always used E-R functions based on Pope et al. (1995) although a range of available studies exist. However, different functions have been used at different times, in order to adjust for the possibly higher PM concentrations in past years. In the last ExternE Transport Project (2000) it was decided to revert to the original PM2.5 function, because of its greater inherent attractions and to deal explicitly with the problem of possible over-estimation. The mid-estimate (used in this report) scales down the PM2.5 function (converted to PM10) by a factor of three, to take account jointly of possibly higher values historically, and of what at the time seemed to be more extreme acute effects of particles on mortality in US studies compared with those in Europe.
For several recent years, it was understood that estimated PM risks in the US studies were higher, per µg/m3 PM10, in the US than in Europe. This was noted and discussed by the APHEA authors but without explanation. Consequently, in ExternE in recent years, several E-R functions for particles and health based on US studies have been scaled down with the aim of improving transferability to Europe. It happens however that the most recent US results,

based on the 20 largest US cities, show meta-analyses results very similar to those of APHEA!
With respect to chronic bronchitis, we have always used the study of Abbey et al. (1995). For use in Europe the most recent ExternE approach was to scale down this function by a factor of two to account for the difference between recent and historical estimates of exposure and the (presumed) higher particle effect in the USA compared to Europe.
Because of this changes the values for PM impacts presented here are lower than in previous reports and should be considered to be conservative estimates. This is discussed further in paragraph 2.2.9.3.
In addition ExternE distinguishes between different sorts of particles, emitted during energy production, from the tail-pipe when driving or formed later (when gases interact). ExternE tries to link up each kind of particle with E-R functions which reflect the particles' relative toxicity. At present epidemiological functions for particles are indexed by PM10 or PM2.5. Which function is used depends on the nature of the particles being evaluated. Based on what is known or widely believed about the relative toxicity of different sorts of particles (and there is much that is not known about this), ExternE has treated sulphates, and primary particles from transport, as if they had the same effect (per $\mu g/m^3$) as PM2.5, whereas we have treated nitrates, and primary particles from electricity generation, as if they have the same effect (per $\mu g/m^3$) as PM10. In addition all particles from transport (esp. from diesels) are though to be carcinogenic.

2.2.6.1.c. E-R functions related to ozone

There is strong evidence linking (daily) ozone and (acute) mortality (See, e.g., ExternE (1995), the relevant APHEA meta-analysis, the ozone chapter in COMEAP (1998) and Thurston and Ito (1999)).
The most delicate point of discussion with respect to ozone has always been the presumed existence of a threshold for acute effects. The idea of 'no threshold' for the acute effects of ozone is problematic for many people, because it seems contrary to intuition although the question could also be reversed :” If there is a threshold, what might it be?”. For many years the same issue with particles was under discussion, where no threshold is now widely accepted as the best working basis for quantification, simply because various proposed thresholds have been shown not to be sustained.
In reality, the situation is unclear and will always remain so. The absence of a threshold can never be proven, because it is not possible to carry out a definitive study. Some degree of extrapolation is necessarily involved, because any ozone effects at very low concentrations are likely to be too weak to be identified reliably.
The issue was considered in some detail in ExternE (1995), where it was concluded that, on balance, the evidence favoured a no-threshold position. COMEAP (1998) highlights hospital admission results from London which are consistent with and indeed suggest a threshold between 40 and 60 ppb. Considering the evidence as a whole, however, from North America as well as from Europe, COMEAP (1998) 'recommend(s) that no threshold is assumed'.
It seems that the assumption of no threshold continues to be the approach best supported by current epidemiology. For example, the absence of a clear or statistically significant ozone effect in winter in some studies (see e.g. Hoek, 1997) does not prove that there is no effect; rather, that any effect was too weak to be detectable .

2.2.6.2. Crops

The study of air pollution effects on crops (and ecosystems) is entirely limited to plants, but no direct effect on large animal species are expected from present day concentrations.
With the exception of some effects in forests, the question of causality poses much less problems. Contrary to humans, most plants (including many crops) can be studied adequately in toxicological studies. Three pollutants were evaluated for their effects: NOx, SO2 and O3.
At this moment, it is assumed that NOx alone does no damages to plants at the concentrations presently measured. It does however play a significant role in the formation of ozone and can exacerbate the effects of SO2, but this last effect is not quantified by any E-R-function.
Deposition of NOx can also contributes to acidification of the soil.
The most important effect is a fertilisation effect. In agricultural crops and (commercial) forests this increases production. In natural ecosystems where N is a limited nutrient, such deposition may alter the species composition, making them less valuable. In this study we have not looked at exceedances of critical loads (generally between 5 and 35 kg N/ha.yr for terrestrial ecosystems). Included in our calculation is the benefit of the fertilising effect (only for agricultural areas) on a European scale.
For SO2 there are three impact pathways. SO2 has a direct toxic effect, deposition of S could be beneficial (esp. as fertiliser for crops) but also contributes to acidification. For the toxic effect we have used the established set of ExternE E-R-functions for a number of crops that are common throughout Europe. For the effect of acidification (resulting from NOx and SO2 deposition) we calculate the cost of compensatory measures (i.e. the additional liming of agricultural soil).
For all pollutants studied ozone is probably the most phytotoxic component. It is a very reactive gas that readily damages cell membranes but does not acumulate or interact with cell metabolism. Disruption of the function of the stomata is also a common mechanism. A wide range of studies confirm that ozone can damage many plants at concentrations over 40 ppb.
For several crop species we used the E-R functions have derived in the ExternE project.

2.2.6.3. Materials

The life of a large number of materials is limited when they are corroded by SO2. In ExternE there are standard E-R functions for limestone, sandstone, painted and galvanised steel, paint, zinc, mortar en rendering. The impact of soiling on buildings was retained for sensitivity analysis only. These E-R functions agree to a large extend with studies conducted for the UN-ECE and have been updated in the most recent ExternE project.
A specific problem associated with materials is the limited amount of information on the amount of exposed materials. There are a number of estimates for European cities, but these are not really relevant to the Belgian situation. Lacking better information a European average was calculated from these studies. Another limitation is that the specific (and costly) damage to historical buildings is not taken into account.

2.2.7. Monetary valuation

In a last step, impacts are valued in monetary terms. There are several approaches to ‘value’ or put a weight on impacts. Our approach basically aims to value impacts, based on societal preferences, as measured by the willingness to pay of the individuals (man in the street) for environmental goods or services. This is a very different perspective then e.g. weighting impacts based on expert judgements or by means of control costs that reflect preferences of policy and decision makers.

Willingness to pay can be measured by different means. We can distinguish three major categories of information : market prices, revealed preferences methods and stated preferences. Some impacts can be valued using market prices (e.g. crop losses). For most impacts however, one has to rely on methods from environmental economics to estimate the value of environmental goods. The valuation of the non-market goods has received a lot of attention and it remains very difficult to make a good assessment of the total value of an environmental good. Figure 6 gives an example of both use and non-use values of the relevant environmental goods. Whereas it is easier to estimate the direct use value of a natural resource (e.g. timber revenues), it is more difficult to get figures for indirect uses (e.g. recreation) and even more difficult for non-use values like the bequest values (our interest to preserve resources for future generations) or existence values (e.g. preserving biodiversity). Methods have been developed to estimate these types of environmental functions by looking at related markets (e.g. housing markets) or by questionnaire approaches, but there are methodological difficulties and for a number of areas data are relatively scarce.

	health	materials	agriculture	forests	water	biodiversity
Use Values						
Additional expenses	medicine	repair buildings	liming	drive further for visiting or fishing		
Production losses	labour days lost		yield loss	timber loss	lower fish cash	genetic diversity and drugs research
Other	suffering	amenity cleaner buildings	landscape	recreation, amenity	fishing, amenity	bird watching
Non use Values						
option value		may want to visit		may want to go for recreation, fishing, watching		
existence value		cultural heritage		common heritage,		
ethical	suffering of family			common heritage,		
future generations	healthy world			ecological heritage		

Figure 6: Examples of type of values to be measured to capture the full value of ‘environmental and public health goods’

For public health we have data that cover both use and non-use values, but for the other impacts categories, we mainly have only data that cover use values like building repair costs or yield losses. Two issues are of a specific importance : i.e. valuation of mortality and discounting.

The traditional way to evaluate increased mortality risks it to try to estimate the statistical value of life, either by looking at related markets (e.g. willingness to pay for safety equipment) or by contingent valuation studies. ExternE reviewed the major studies in the EC and US, which resulted in an average value of life of 3.4 million Euro. For the valuation of mortality impacts from air pollution, the better approach is to value impacts in terms of life years lost. As there a hardly any studies on the value of life year lost, this value is derived from the average value of statistical life and amounts to around 100.000 Euro. Issue not

accounted for in this value – or still under debate – relate to age dependency, the quality of life years lost and the context of air pollution impacts.
Other health impacts (morbidity) are valued following the same principle, i.e. the average or best data from existing studies. Whereas earlier studies especially were based on US data, in recent years a number of European studies became available and were used in the ExternE 2000 accounting framework. An overview of these data can be found in Int Panis et al., 2001.
As there is no agreement among economist what discount rate to use, we tested several discount rates, and different rates for different problems. In this report, we give the central estimate which is based on the 3 % discount rates for impacts that occur within this generation, and which is especially important for chronic health impacts from PM and cancers. As a sensitivity analysis, we have also taken a value of 0% and 1% into account.
For longer term impacts, and in this context especially global warming, we do not take into account time preference, and a range of discount rates between 0% and 1.5 %, e.g. is used to reflect long term economic growth.

2.2.8. Global warming impacts

For global warming impacts, we have used the results from the ExternE analysis by Tol and Downing (2000). They have used specific models to do simplified impact pathway calculation of environmental and public health impacts from expected global warming scenario's, up to 2100 and – as a sensitivity – up to 2200. On that basis, marginal external costs have been calculated for the greenhouse gases carbon dioxide, methane, and nitrous oxide.
These models take into account the impacts on public health, agriculture, energy demand, water supply, sea level rise and extreme weather events. It has to be noted that both benefits (e.g. benefits from higher temperatures on agriculture, energy demand, public health) and costs are included. A number of (potentially important) impact categories like species loss, biodiversity or indirect effects cannot be assessed or valued.
For valuation, some specific issues arise, which are rather related to ethical and political choices, and are thus best dealt with via sensitivity analysis. This relates to the discount rates and choices on how to value in a consistent way the impacts in richer and poorer countries, especially on public health (mortality).
As a result, it is not surprising that these impacts are by far the most uncertain in our entire analysis. Therefore, ExternE does not report one single best estimate, but rather a range, which is a subtotal for impacts that can be quantified and valued, and only for impacts up to 2100.

2.2.9. Uncertainty

2.2.9.1. Introduction

In every step of the impact pathway analysis we use the best available estimates. Nevertheless all estimates of externalities have large uncertainties. These should not be neglected and properly communicated. Therefore it is important to devise a methodology to calculate or estimate uncertainties. The global uncertainty of an estimate of the external cost per tonne of a pollutant is due to several sources :

- Statistical uncertainty due to the use of scientific or technical data and studies. e.g. the slope of an E-R function, the valuation of a health end-point.

- The choice of one out of several possible models (e.g. for atmospheric dispersion) which implies uncertain assumptions about meteorology and chemical interactions between pollutants.
- Uncertainty because of political or ethical choices. e.g. concerning the discount rate for environmental or health impacts that will occur in the future, or the valuation of a statistical life in different countries.
- Uncertainty about future scenarios e.g. the possibility that ozone resistant crops could be developed.
- Human error. In the interpretation of large incomplete and ambiguous datasets

Statistical uncertainty can be dealt with using classical statistical techniques. The methodology to do this was developed by Rabl and Spadaro in (EC, 1995; EC 1998; EC, 2000) and clearly demonstrated in ExternE (2000). Based on the fact that the impact pathway approach is a multiplicative process, the distribution of the result should be approximately log-normal. An uncertainty interval can then be constructed around the geometric mean μ from the geometric standard deviation σ. The 68% confidence interval is ranges between μ/σ en $\mu.\sigma$; the 95% confidence interval is calculated from μ/σ^2 and $\mu.\sigma^2$.

The complete discussion of all uncertainty issues is clearly beyond the scope of this report, but the interested reader is referred to Int Panis et al. (2000) and ExternE report (2000). Table 1 summarises some key-figures that can help understand the magnitude of the uncertainties in estimates of environmental externalities.

Table 1: General key-figures for the magnitude of different sources of uncertainty.

Impact category	σ	Uncertainty score
Morbidity	2.5	A
Mortality	3.5	A
Materials	4	A
Crops	3 - 4	A
Global Warming	6	C
Ecosystems	6	C

Other issues can best be dealt with by means of sensitivity analysis. In this report we have reported the central estimate. We have included the impacts on ozone and for global warming, we have added the 0-16 Euro/ton CO2 equivalent on top (see below).

2.2.9.2. Marginal versus non-marginal impacts.

Our central estimates are related to a marginal emission reduction, starting from current levels of pollution and background concentrations. These results may be different from those of looking to it from a non-marginal perspective, e.g. when evaluating the benefits of an important emission reduction plan, when there is non-linearity in the chemical reactions and if thresholds do apply. It is yet unclear how future emissions may affect the formation of ammonium nitrates and sulphates. Threshold values in exposure response functions apply for agriculture, materials and ecosystems, but not for public health. Non marginal changes are most important for the regional impacts of NOx emissions on ozone, as will be discussed below.

2.2.9.3. Stratified uncertainty analysis : comparison to 1998 methodology

In addition to formal statistical techniques we often use a stratified sensitivity analysis. This technique was used before in European and UK studies to integrate uncertainty in Cost Benefit Analysis (Krewitt et al., 1999; Holland, 1999; De Nocker, 2000).
The stratified approach ranks impacts according to the subjective uncertainty that a panel of experts has attributed to different impacts from air pollution (Table 2). This takes into account not only the purely statistical uncertainties but some of the other sources of uncertainties as well. Once the impacts have been ranked and calculated, the sub-total can gradually be increased by adding more (but less reliable) functions to the analysis. This methodology was adopted to quantify the benefits of reducing pollutants in the recent UN/ECE protocol on Long Range Transboundary Air Pollution. Below (section 2.3.1.2) we use this analysis to evaluate the importance of the change in impacts between 1998 and 2000 methodology.

Table 2

Ranking	Impact	Group
1	Material damage (excl. paint)	I
2	Crops – N fertilisation	I
3	Acute mortality	I
4	Acute morbidity (excl. RAD)	I
5	Crops – lime fertilisation against acidification	II
6	RAD (restricted activity days)	II
7	Material damage – paint	II
8	Crops – direct effects of ozone on harvest	II
9	Crops – direct effects of SO_2 on harvest	II
10	Chronic morbidity (excl. bronchitis)	II
11	Chronic bronchitis	III
12	Damage to forests caused by ozone	IV
13	Chronic mortality	IV

2.2.10. Updating and differences with earlier figures from this project.

Given these uncertainties, it is self-evident that the results change over time, as our understanding improves and data are being updated. The OSTC project (1997-2000) has therefore reported several different results during this period. All of them used the same basic ExternE framework, but as scientific understanding evolved over time, and/or models were further being developed, the framework has been constantly updated. As a consequence, results have changed over time, to reflect these methodological improvements and scientific development. The first results reported were based on ExternE 1997 or 1998, and refer to the ExternE accounting framework as it had first been developed for transport in 1998 (EC, 1998a). The methodology has been further elaborated and updated in 1998-2000 with applications to transport in several member states of the European Union, including Belgium. (EC, 2000) We refer to this as the ExternE 2000. The results in this report apply the ExternE 2000 methodology, and replace all previously reported intermediate results. Some of these final results have also been summarised in a report on external costs of transport for several European countries (EC, 2000).
Although this report updates and replaces the numbers on external costs of air pollution from transport, reported by Vito before the year 2000, the conclusions on comparisons between fuels, technologies, modes locations and traffic remain valid. The numbers in this report are consistent with articles published by Vito since 2000.

Differences relate both to emission calculations, dispersion and exposure models, dose-response functions and valuations. Therefore, we cannot summarise these changes over time. The differences in results per tonne pollutant are discussed below (section 2.3.1.2).

2.3. Results

2.3.1. Overview of results per tonne pollutant

2.3.1.1. Major impact categories for traditional pollutants

The external costs of a vehicle can be summarised as the product of the emissions from a specific fuel and vehicle technology and driving pattern, with the impact per ton pollutant representative for the chosen trajectory. To understand the different results (e.g. in Appendix 3), the different factors that influence emissions and costs per tonne pollutant should be understood. We will therefore discuss these first.
The overview of the environmental damage costs from air pollutants SO2, NOx, and particles gives us a good idea of the main issues for assessment of externalities for air pollution. Table 3 gives both percentages for different impact categories, and indicates the order of magnitude of the damage cost for emissions from transport Belgium.

Table 3: The relative share of impact categories in external costs estimates from air pollutants from Belgium, for the major pollutants emitted from high stacks.

	Share in total monetised externalities			
Impact category	**SO2**	**NO_x through nitrates**	**NO_x through ozone**	**primary particles**
Public health				
Mortality	72%	72 %	35 %	70 %
Morbidity	26 %	28 %	25 %	30 %
Agriculture	0.4%	-	40 %	-
Materials	3 %	-	-	na
Ecosystems	N.M.	N.M.	N.M.	N.M.
Subtotals (KEuro/ton) for emissions from				
Transport use phase [a]	4-15	3 – 4.5	minus 2 to ? [c)]	100 – 400[d)]
Non-use phase Belgium[b]	6	3.5		12
Non use phase EU average	1.5 - 7	1.1 - 7		1- 13

N.M. = not monetised, but critical load exceedance data are available
na : not available
a : the range represents different locations in Belgium, except for ozone for which the range refers to marginal versus non marginal impacts.
b : average for different stack heights and locations in Belgium
c : based on EMEP matrices
d : includes
e : average for EU 15

2.3.1.1.a. Public health impacts are dominant.

For SO_2, NO_x CO, VOC and particles, the most important externality – as far as it could be monetised - is their impact on public health (Table 3). Especially the impact from particles, sulphates and nitrates on chronic mortality proved to be the dominant impact category and accounts for more than 80 % of the quantified externalities. This reflects the major concern that has risen over the last years about the impact from small particles on human health. The large impacts we quantify are the product of small increases in concentrations that affect a large amount of people, living close (for PM) or up to 1500 km from the emissions' source (for nitrate and sulphate aerosols). Therefore, the analysis needs to include dispersion of emissions at both the local and regional level. The relative high figures for NOx and SO2 are related to the impacts from nitrates and sulphates, secondary particles formed from NO_x, SO_2 and ammonia (NH_3). The dose-response-functions used and underlying assumptions and uncertainties are discussed in detail in the methodology chapter. Uncertainties relate to the background concentrations of ammonia, the formation of particles, their impact on public health and the valuation of these impacts.
The impacts on morbidity are the sum of a large number of different indicators including both mild (coughing) and very serious (non-fatal cancers) conditions. The subtotal is much less important than the mortality impacts, except for ozone for which it is the most important category. Impacts on buildings are relative important for SO_2 and may be important for particles (soiling, but there are no dose-response relations available to take this effect into account). The net impacts on agriculture from SO_2 they are almost negligible.
Because impacts on public health are by far the most dominating impacts, population density plays an important role in the site specificity of the impacts per ton pollutant.

2.3.1.1.b. Regional Ozone impacts

As elaborated before, ozone formation is hard to model. At present, we only take into account the regional impacts from emissions from NOx and VOC on ozone, as the local and global impacts could not be modelled.
It has to be noted that the marginal regional ozone impacts from NOx emissions in Belgium are negative (NOx emissions reduce total ozone impacts) and thus appear as a negative cost (benefit) on some of our figures throughout the report. This reflects the current situation and the current ratio between NOx and VOC concentrations. It is estimated that this situation will continue until 2010, taken into account the emission reductions of NOx and VOC. Towards 2010, the changed ratio of NOx and VOC concentrations is expected to lead to a reduced ozone formation from NOx emissions. This has to be taken into account when these data are used for longer-term policy making or investments with impacts after 2010. Therefore, impacts on ozone are added as a sensitivity analysis.
The situation for VOC emissions is different and reducing VOC emissions today will lead to a reduction of ozone.

2.3.1.1.c. Ecological impacts

Finally, it is important to note that the impacts on ecosystems could not be monetised and this is a very important blank in the accounting framework. It is possible to quantify to which extent emissions contribute to the exceedance of critical loads. As for Belgium, critical loads are exceeded for a high percentage of ecosystems, this 'distance to target' indicator suggests that this is an important impact category, especially for eutrophication.

2.3.1.2. Stratified comparison to 1998 methodology

Most users of externality estimates are confronted with uncertainty for the first time when they notice that estimates have changed with respect to previous reports. Therefore we seize this opportunity to demonstrate the stratified uncertainty analysis. Figure 7 demonstrates for each of the pollutants SO2, NOx and PM10 the relative contribution of different functions (according to Table 2) to the ExternE mid-estimates of the 1997 methodology (EC,1999) and the most recent 2000 methodology (EC, 2000).
Chronic mortality, the single most important impact category, belongs to group 4 (lowest confidence). In 1997 this function dominated the total impact. As discussed before, the 2000 estimates for chronic mortality are lower. It is still the most important impact category, but it no longer dominates the total impact attributed to particles. The relative weight of the more reliable estimates in the total has significantly increased.
The increase in the valuation of morbidity impacts (based on the new European studies) has caused the impacts from category I, II and esp. from category III to rise from 1997 to 2000. The increased valuation of chronic mortality cases is the most important factor. Most functions (albeit not the most important ones) have remained unchanged.

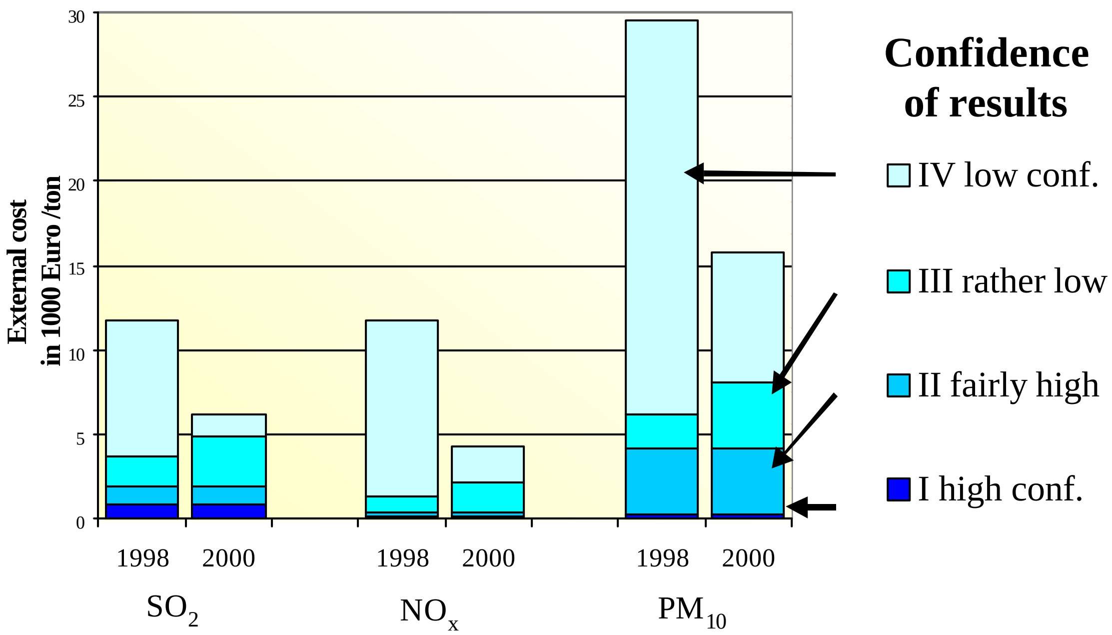


Figure 7: Stratified comparison of ExternE 1997 and “ExternE” 2000 methodology

2.3.1.3. Global Warming

The ExternE project relied on existing models – the Open Framework and the FUND model – to assess the impacts from greenhouse gasses. These models include a wide range of world-wide impacts from temperature rise, ranging from investments in flood protection to changes in the spread of malaria. Specific runs were made to make the approach as far as possible consistent with ExternE and specific attention was given to sensitivity of the results for changes in assumptions related to the socio-economic scenarios, temperature rise and valuation issues (discounting, equal valuation of impacts in poor and rich countries).

The currently recommended values (Table 4) are an order of magnitude lower than those that were applied earlier[2]. Tol and Downing (2000) claim that this reflects the more optimistic tone of recent impact literature. With the inclusion of new insights (benefits, VLYL valuation) into the impacts of climate change, it can no longer be excluded that marginal costs may even be negative, particularly for methane. The sign of the costs is model and region dependent.

At this moment the mid-estimate from Table 4 is lower than the values that are used in a range of other recent studies. The results presented here should therefore not be taken as final estimates. The impacts covered by the models used are only a fraction (of unknown size) of all climate change impacts. Particularly, large-scale disruptions, such as a breakdown of North Atlantic Deep-Water formation or a collapse of the West-Antarctic Ice Sheet or impacts in the 22nd century, are excluded from the analysis. The methodologies to estimate climate change impacts in a different future remain weak. Adaptation is not included in its full complexity. Valuation of impacts is still troublesome, particularly for nature and health. Our knowledge of atmospheric chemistry and climate has substantial gaps. The estimates reflect our current best knowledge, and indicate a stimulating research agenda.

Table 4: Recommended marginal costs by ExternE.[a]

	Minimum[b]	Low[c]	Central estimate[d]	High[c]	Maximum[b]
CO_2 (€/tCO_2)[e]	0.1	1.4	2.4	4.1	16.4
N_2O (€/tN_2O)[e]	24.3	440.2	748.3	1,272.1	5,242.1
CH_4 (€/tCH_4)[e]	1.9	28.2	44.9	71.5	257.0
N (€/kgN)[f]	-5.5	198.2	337.0	527.9	1,270.2
S (€/kgS)[f]	-35.8	-16.6	-9.8	-5.8	0.0

[a] Emissions are in the period 2000-2009. Costs are discounted to 2000.
[b] Minimum and maximum are as in Tables 5, 11 and 12.
[c] High and low approximately span the 67% confidence interval.
[d] PRTP equals 1%. Values are world averages.
[e] Model is *FUND*2.0. Time horizon is 2100. Scenario is IS92a. Morbidity risks are valued based on the value of a life year lost. Note that the marginal costs of carbon dioxide are here expressed per tonne of CO_2, rather than per tonne of carbon as in earlier tables.
[f] Model is *FUND*1.6. Time horizon is 2200. Scenario is FUND. Morbidity risks are valued based on the value of a statistical life. Uncertainty is based on an assumed geometric standard deviation of 1.7, in line with CO_2, N_2O and CH_4. Nitrogen emissions are from aircraft only.

In this report, we have chosen not to apply the 2.4€/tCO2 central estimate for our calculations and graphs. Instead we have decided to display Global Warming costs in all graphs on top of all other impacts as a sensitivity. To this end we value CO2 emissions at 16.4€/t which is the maximum from the ExternE range. This way, it is easier to ‘see’ the relative weight of global warming impacts (esp. in old vehicles) compared to other impact categories. Second, given the gaps and uncertainties, we think the higher figure is more in line with the attention that global warming issues get from policy makers (in view Belgian of Kyoto commitments) as this value is of the same order of magnitude as the abatement costs for Belgium to meet Kyoto agreements. Nevertheless any interpretation should take into account that the lower limit of the ExternE range is almost zero. In this way it is very easy for the user of the graphs in this report to assess whether different assumptions about the GW cost would influence comparisons between vehicles, modes etc.. The 16.4€/t value was only used where CO2 is shown separately in relation to other external costs. In all other marginal and in all aggregated

[2] The best estimate in the ExternE 1997 methodology is a range from 18 to 48 Euro/ton of CO_2-equivalent

cost calculations we have systematically used the 2,4€/t value, mainly to ensure comparability between the analysis made by Vito and calculations in the UK (Int Panis et al., 2000).

2.3.2. Factors that influence impacts

2.3.2.1. Technical factors

A wide range of technical factors influence the emissions (and impacts) of cars. For the calculation of externalities, the two most important factors are the relevant European emission standards (or EURO types, based on year of first registration) and fuel type. The importance of these can be summarised for the Belgian context as follows.

On all trajectories, old diesel vehicles (Uncontrolled) have the highest external costs by far. Costs are often 2 or 3 times higher than those of uncontrolled petrol fuelled cars. Progress has been made in limiting the externalities of diesel vehicles with the introduction of the EURO1 and EURO2 emission standards, but only the very latest models (complying with EURO3 emission standards) seem to perform better than uncontrolled petrol cars.

Results for LPG cars come out very low, but this it partly attributable to the lack of particulate emission factors for these vehicles in the standard EcoSense database. Therefore these number are not directly comparable to those of petrol cars. Additional modelling tasks for LPG cars, including the particulate and carcinogenic emissions, have been performed for the Belgian reference trajectories. This increases externalities of LPG fuelled vehicles between 20 and 100%. This demonstrates that air pollution impacts from LPG cars with a three-way catalytic converter are indeed lower (about 50%) than for petrol cars on the same trajectory. Older LPG fuelled vehicles without a catalytic converter are no match for today's petrol cars and, in rural conditions, may even have higher externalities than the latest diesel models.

Logically in each category the later EURO-types perform better than the older cars. The largest decrease in external costs was achieved by the introduction of the EURO1 standard, which brought a decrease with a factor of 3 from EURO0 externalities. Subsequent improvements from EURO2 and EURO3 legislation are substantial but relatively lower. This is illustrated in Figure 8.

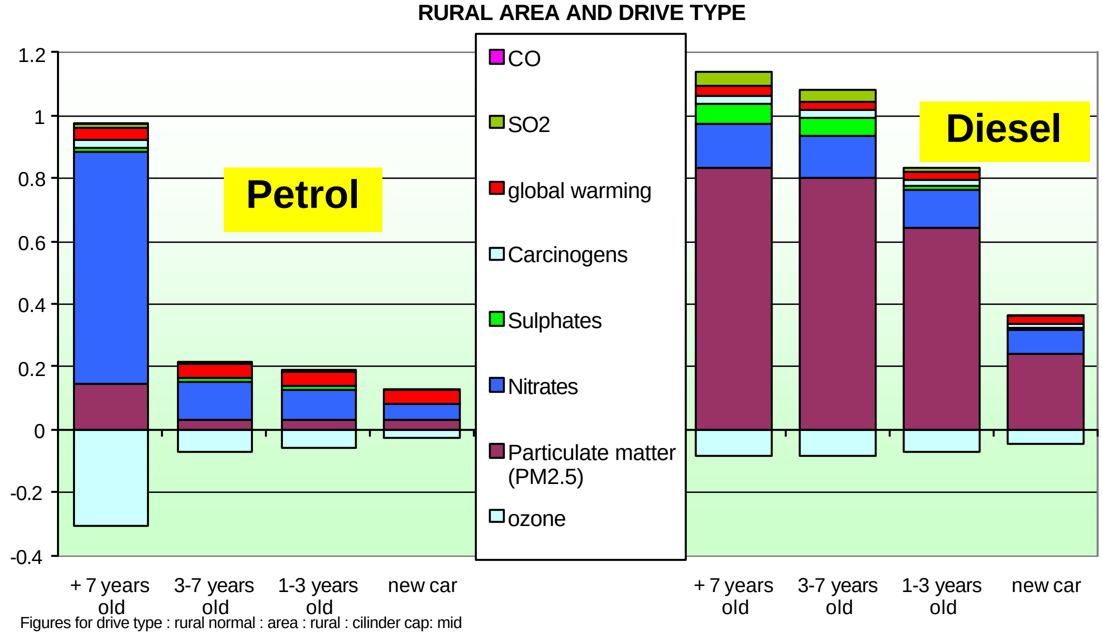


Figure 8: Estimates of marginal external costs for different vehicle technologies.

These results are of course not applicable to individual brands of cars. They are based on average values for cars belonging to a limited number of categories. Each category is taken to be representative for a large number of different models from different manufacturers with different engines. The variation within each category cannot be taken into account at this moment because accurate emission factors are not available for every model of passenger car.

2.3.2.2. Location

2.3.2.2.a. Location specific speeds and emission factors

Emissions however are strongly dependent not only on vehicle type and emission control technology, but also on the (average) speed. This speed depends on the type of trajectory. For that reason, we have used different speeds in different locations. Also we have distinguished normal from dense traffic conditions.
Table 5 gives an overview of the average speeds under normal traffic conditions for the vehicle types that were modelled in this study. It shows a pronounced difference between the average driven speed in urban traffic by different vehicles. Buses have a lower, and motorcycles a higher average speed than passenger cars because of the different number of stops (e.g. at bus stops or traffic lights). For heavy-duty vehicles, the lightest weight class has the lowest average speed; it is assumed that these vehicles drive also in the city centres (inclusive small streets), whereas the heavier ones generally take main streets.

Table 5: Average speeds (km/h) for different vehicle types and road types under normal traffic conditions [sources: MIRA2, KMS and Vito]

Vehicle type	Average speed (km/h)		
	urban	rural	highway
Passenger car	*22*	*51*	*110*
Public bus	*15*	*45*	*80*
Coach	*20*	*45*	*85*
Light duty	*22*	*51*	*110*
Heavy Duty (>32 t)	*30*	*45*	*85*

2.3.2.2.b. Location specific exposure and impacts

For all vehicle types we find that (for the Belgian reference trajectories) costs are highest in urban settings and lowest in rural areas. Externalities from highway traffic are intermediate between urban and rural traffic. This result is directly linked to the differences in exposure of the population between the chosen reference sites (**Figure 4**).
It is important to stress that this observation only holds for the reference trajectories. Externalities for some stretches of highway (e.g. between Brussels and Leuven or close to Antwerp) may be higher than for some urban trajectories because of a combination of high population density and higher emissions at high speeds. On the other hand, some highways that cross sparsely populated areas approach damage/km values that are typical of rural traffic. This is illustrated in Figure 9.
A trajectory from the centre of Brussels to Liège via the E40 motorway was split in 10 sections to study the variation of externalities along this route. It can be clearly seen that urban traffic causes higher externalities than highway traffic. In addition, there are significant differences in impacts from different stretches of highway (up to a factor of 6 for an uncontrolled diesel car). Externalities first fall to lower values when the car leaves Brussels but then climb again to a peak where the highway passes south west of the city of Leuven. The lowest values were found for the part of the trajectory along the border between Flanders and Wallonia. In the neighbourhood of Liège costs rise again especially in urban traffic. This type of geographical variability should always be kept in mind when extrapolations from the reference trajectories are made.
Some of the differences that we find between results for Belgium and other European countries can be explained by typical attributes of the population distribution.

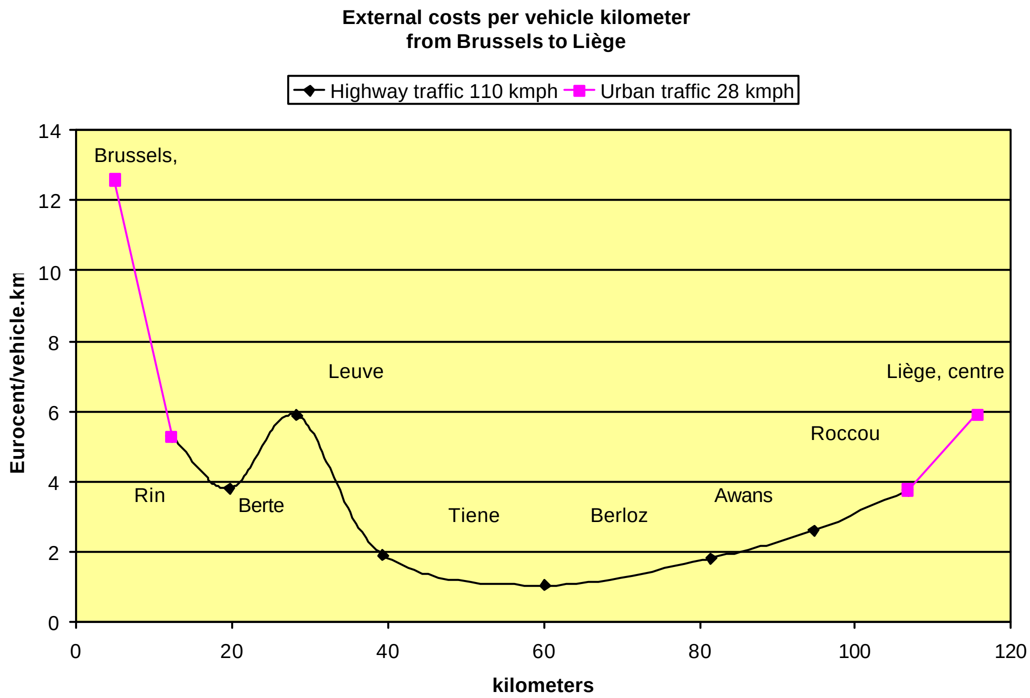


Figure 9: **External costs/km for an uncontrolled diesel car are strongly correlated with local population density along the trajectory.**

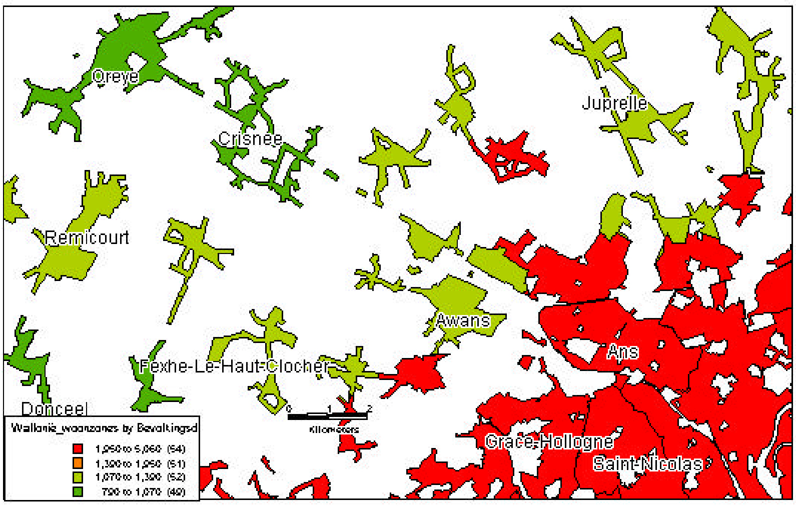


Figure 10: **Building areas NW of Liège clearly show the trajectories of major roads (Plan secteur)**

Results for Brussels are among the highest in the country, but relatively low compared to other capitals because both the area and the population of Brussels are much lower than in major foreign cities. In contrast to this are the much higher impacts that we find in rural areas in Belgium. This is caused predominantly by the high population density in general. Not the villages themselves have high population densities, but there are many more of them close by compared to rural areas elsewhere in Europe. In addition, the (lack of) land use planning has led to uncontrolled ribbon building in many areas . Therefore relatively more people live right next to important roads. See Figure 10 for an example.

2.3.2.3. Aggregation

Most reports and papers on transport externalities give only estimates of the marginal external costs (the costs caused by an additional vehicle). In this report we also present aggregated results i.e. results for all cars or all road transport in Belgium which mean that values for urban, rural and highway traffic are weighted.

In this paragraph we briefly describes the issues and methodology associated with aggregating marginal externality data to levels required by policy instruments. Later in this report we will use this methodology to analyse the effects of successive emission standards (EURO types) and other factors in the evolution of road sector externalities.

Although previous work under the ExternE project had developed and demonstrated aggregation of sector emissions for the electricity generation there are theoretical and practical issues that are different for the transport sector and which necessitate a different approach.
The first issue is with respect to speed. Transport emissions vary significantly with speed and this variation is non-linear. Moreover, for road transport, the speeds on particular road types (highway, main road, etc.) also vary with the numbers of other vehicles on the road. As a result, even emission aggregation becomes an issue for transport.
The second issue centres on the assessment of mobile emission sources. The emissions from vehicles vary over time as well as by location. As ExternE uses linear dose-response functions, the issue of time dependency is less important (all emissions from any one location for any time period can effectively be assessed as an annual emission). In contrast, the issue of location is of major concern. Previous aggregation studies have shown that the geographical transferability of global impacts and regional impacts (within one country) are acceptable but the transferability of local impacts is poor. For power stations this is not a major issue; emissions from the electricity generation sector are generally released from high stacks, away from major urban areas, and local impacts are a small part of overall damages. In contrast, local impacts are often dominant for transport, because emissions are released at ground level, often in areas of high population density. The implications of this are clear. A much greater level of geographical resolution is required for the local level assessment for transport aggregation.
Work within ExternE has shown that there are small differences between total and marginal impacts. Potentially greater differences exist between total and marginal costs, especially when considering large sectors such as transport, though to date these have not been assessed with ExternE in detail. In all cases we have therefore assumed linearity and the marginal external costs have been applied. Non-linearity in dispersion and chemical transformations is also not accounted for, except for ozone where a European average cost has been used.
The aggregated estimates were calculated by adding all marginal external costs/km for all combinations of vehicle type, drive type and location. Marginal costs runs were undertaken for each combination of emission standard (pre-EURO, EURO I, EURO II, EURO III), fuel (diesel, gasoline, LPG) for six drive types. The six drive types (urban peak, urban non-peak, highway peak, highway non-peak, rural peak, and rural non-peak) each assumes representative average speeds. Effects from cold starts on emissions and vehicle age on mileage were taken into account for passenger cars only. The analysis then uses these costs per km with the national data on fleet distribution and national vehicle km to estimate total

costs. Clearly this is a very tedious and data intensive calculation which is discussed in more detail by Watkiss and Int Panis in ExternE 2000 and Int Panis and Watkiss (2000).
In all cases, the marginal external costs have been applied. Non-linearity in dispersion and chemical transformations is not accounted for, except for ozone (see marginal analysis for discussion) where a European average cost has been used.

2.3.3. Results for road transport

2.3.3.1. Use phase impacts from passenger transport

2.3.3.1.a. Passenger cars

Introduction

Given the dominance of passenger cars in the transport sector today. This study has focussed on cars and more emission factors have been deduced and externalities calculated for passenger cars than for any other vehicle. As a consequence many more results have been obtained than can be presented or discussed in this report. Nevertheless it is our goal to provide the user/reader of this report both with objective ready-to-use numbers as with guidance on the interpretation of the results. Therefore we have decided to include detailed results for most road vehicles in tables in an annex to this report. Marginal externalities are listed for many vehicle types and the contribution of different pollutants and impact categories is given. In the next two paragraphs we first discuss some characteristics of the marginal externalities before we aggregate them into a national total. More detailed accounts can be found in a number of publications that were drawn from this work (Appendix 4).

Marginal external costs of passenger cars

To begin our discussion of marginal external costs we first take a look at cars that comply with the current EURO3 emission criteria. Emissions were calculated with MEET. The results are shown in Figure 11. For comparison the external costs of (pre-catalyst) EURO0 cars have also been included in the graph. The results are presented as stacked bars to highlight the fact that external costs are always a subtotal of the damages caused by each of the different pollutants. The importance of particulate matter (PM) and aerosols (nitrates and sulphates) becomes immediately obvious. Marginal impacts from ozone are thought to be negative in Belgium because of the depressing effect that additional emissions of NOx currently have on ozone formation. The cost for each pollutant (depicted as a single fragment in the bar chart) is also a subtotal of all damages that it has caused following its emission. Most costs of PM, NOx and SOx are derived from public health impacts including both “mild” conditions such as coughing and asthma as well as more serious respiratory and cardiovascular illness. Health impacts from CO and carcinogenic VOCS are much less important and hardly visible in Figure 11. Although these pollutants can cause cardiovascular diseases and fatal cancers such as leukaemia, they are emitted in such low concentrations that the number of affected people is very low. Damages to crops and buildings are always much lower than the damages to public health and don’t have a significant effect on the conclusions presented here.

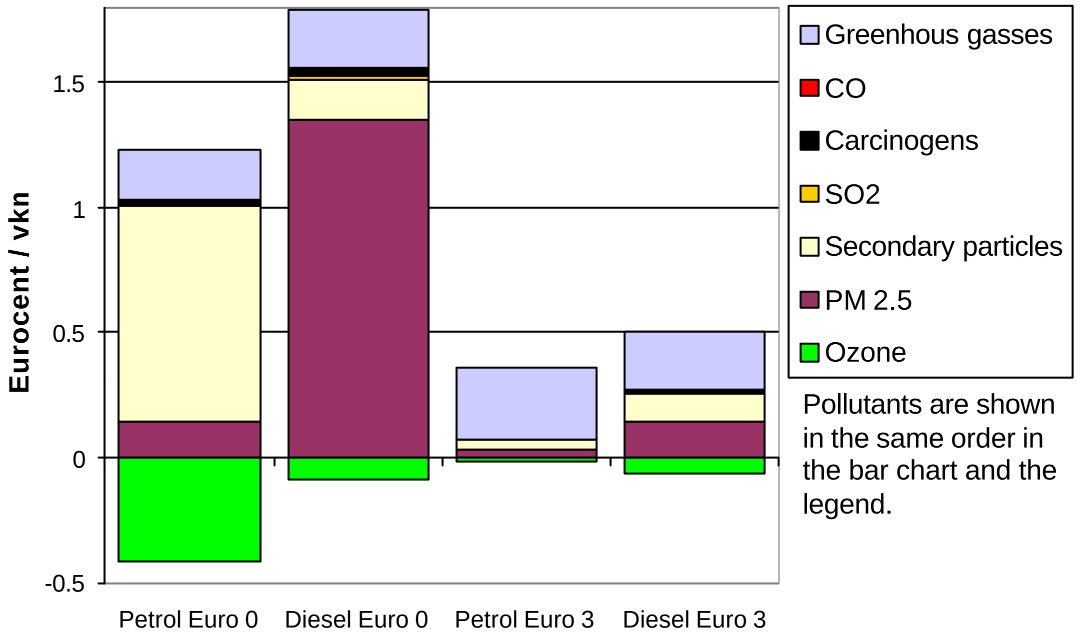


Figure 11: Marginal external costs shown as a subtotal of impacts from different pollutants (rural drive).

The air pollution caused by (pre-catalyst) EURO0 petrol cars was dominated by the health impacts of secondary particles (based on nitrate aerosols). The introduction of three-way catalysts, needed to comply with EURO1 emissions standards, has significantly cut the emissions of NOx and VOC. From the graph it can be seen that this regulation has significantly reduced the impacts of nitrates and ozone from new petrol cars. The reduction in externalities of diesel passenger cars was mainly accomplished by the reduction PM emissions. The resulting decrease of PM impacts explains most of the reduction between EURO0 and EURO3 diesel cars.
Major reductions in CO emissions, although spectacular in terms of tailpipe concentrations, hardly affected the total external air pollution cost, because their impact is much less important than that of other pollutants.

The results discussed above only apply to cars with a specific cylinder capacity and for rural driving conditions. Although results are different for different locations and different engines types we can summarise them by looking at two variables only: fuel and emission standard (EURO type, see paragraph on technical factors 2.3.2.1).

2.3.3.1.b. Marginal air pollution cost of an average Belgian car

The results discussed above only apply to cars with a specific cylinder capacity and for rural driving conditions. It is clearly an impossible task to discuss all damages for all pollutants of every vehicle type at different speeds. It is more useful to weigh these results according to their share in the present day fleet.

Table 6: Comparison of the external costs per vehicle.kilometre in different locations as calculated with different emission factors. Average car (Belgian fleet 1998)

Cost in Eurocents/km	Highway	Large city	Small city	Rural area
Input from INFRAS	0.9	4.5	1.7	0.7
Input from MEET	1.0	6.7	2.5	0.6
Difference	10%	33%	31%	10%

In Table 6 we have made a comparison for an average Belgian car in 4 typical situations. As expected, marginal costs are lowest in rural areas and highest in cities. There is no difference between MEET and INFRAS in this respect.
The external costs for traffic in rural areas and on highways are similar for both sources of emission factors. However, at lower speeds in urban traffic, we find a larger difference. This is due to the combined effect of larger differences between INFRAS and MEET at lower speeds and the multiplicative effect of higher impacts in cities (larger population at risk). The difference between large and small city is exclusively caused by the population density, the emissions are the same in both cases. When we study the outcome of this calculation in more detail, it can be shown that higher emissions of particulates by diesel cars in MEET explain most of the effect. The differences at the level of specific vehicle classes are often much larger than the values reported in Table 6. Because the effect at the fleet level is more important from a policy perspective, we will use this view in the next paragraph on aggregated estimates of external costs.

2.3.3.1.c. Aggregated externalities for all Belgian passenger cars.

By adding all marginal external costs, over all vehicle types, locations and circumstances, we obtain a total cost figure for the entire fleet of passenger cars. For the year 1998, we find that the total is between 1.4 and 1.6 billion Euro (results with INFRAS and MEET emissions respectively). About 80% of all costs are caused by diesel cars. The remaining 20% can be attributed to petrol cars; LPG is a marginal fuel in Belgium. With respect to location it is estimated that city traffic causes two-thirds of the impacts, three times more than highway traffic.

A 10% difference in total external costs is obtained when emissions data from both INFRAS and MEET are used. This is smaller than could be expected from pairwise comparisons of specific vehicles and from literature (Kühlwein and Friedrich, 1999). Apparently, some of the larger differences found at the vehicle level compensate each other, or have a small weight in the overall sum. It is interesting to find that while uncertainty about emission factors contributes significantly to the uncertainty of externalities for specific vehicles, the total is rather insensitive to the choice of an emission model.

The aggregation was repeated for all years between 1993 and 1998. During this period the changing composition of the fleet and the mileage were taken into account, but no changes to the composition of fuels or mobility distribution data were included in this exercise. The results with both emission models indicate that the cost of air pollution from Belgian cars has only slightly decreased between 1993 and 1998. Although the important pollutants such as NOx, PM and VOC from new cars have been cut to very low levels, the spontaneous renewal of the fleet has not ensured these reductions to take their full effect. The rate at which old cars are replaced by new models imposes a time lag of several years before the implementation of new regulations can be observed. However even over a time frame of 6 years, there is no impressive reduction. This is due to the combined effect of three factors:

1. The number of registered passenger cars in Belgium keeps increasing (+12% since 1993).
2. The average mileage per car keeps increasing every year (+15% in total fleet mileage)
3. There is a clear trend in Belgium towards more diesel cars. Very often old petrol cars are replaced by diesel cars that have higher external costs.

In the results based on INFRAS emissions, we only find a very slight decline since 1995, which is mainly caused by the lower annual mileage of older (EURO0) cars. Results based on MEET emissions show a more pronounced decline, but less than what was expected (see Int Panis et al. 2000 for a more detailed discussion).

The evolution of externalities in the future (2000-2010) depends on a whole set of parameters. Some of these, such as future European emission standards, are well known, some depend on economical assumptions and others will be decided by political decisions. In this paper we will only discuss our analysis of the trend scenario from the Flemish TEMAT-model.
By combining estimates of the expected fleet and traffic growth from this trend scenario with the introduction of new European emission standards (EURO3 in 2000 and EURO4 in 2005) and the planned reduction of the fuel sulphur content (50ppb in 2005) we obtain the graph shown in Figure 12.

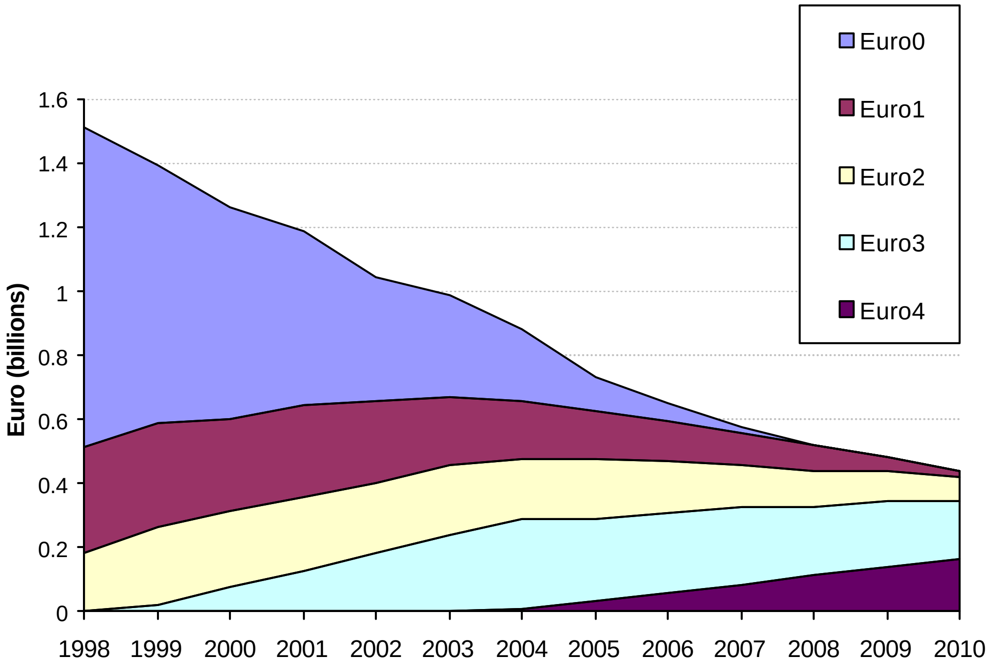


Figure 12: The evolution of environmental externalities. 1998-2010, Belgian Fleet

At this time (1998-2001) there is a distinct decrease of externalities parallel to the introduction of cleaner types of vehicles. However, the driving force behind the decrease is the elimination of the remaining cars without any emissions control (EURO0) rather than the performance of sophisticated EURO3 and EURO4 cars. The old EURO0 cars continue to dominate the air pollution costs until 2001.

The reliability of this prediction depends to a large degree on the assumptions in the TEMAT trend scenario. A Monte Carlo analysis of 23 fleet and mobility parameters showed the 1998-total to be rather insensitive to the assumptions, with a 90% confidence interval of about 0,2 billion Euro wide. Unfortunately a similar analysis 2010 estimate produces very wide confidence intervals. The main trends depend largely on policy decisions which cannot be modelled as uncertainties with a known statistical distribution. Because the TEMAT model keeps all unknown parameters constant, the estimate for 2010 shown in Figure 12 seems a very optimistic one. If the recent trend towards diesel continues for example, the Belgian fleet could have a 50% diesel share by 2010. Any increase in urban traffic or peak traffic would also cause an increase in externalities. Recent work (Int Panis et al., 2001) suggests that the average outcome for the year 2010 is 685 million Euro with a 95% confidence interval between 619 and 754 million Euro. This demonstrates that it is important to take all uncertainty into account. Calculating external costs with our methodology will therefore prove to be an effective tool to compare the different scenarios that are currently drawn up by federal and regional government authorities.

2.3.3.1.d. Motorcycles

Introduction

One vehicle which has recently been very successful as an alternative to passenger cars, is the motorcycle. In this paragraph we compare motorcycles with different cylinder capacities between 50 and 750cc. The consequences for air quality of modal shifts between passenger cars (incl. car-pooling), motorcycles and city buses are evaluated. This evaluation is based on the calculation of environmental external costs.

In contrast to some other, mainly southern European countries, Belgium’s fleet of powered two-wheelers is rather small. Most motorcycles in Belgium (98% of the fleet) are equipped with a four-stroke petrol-fuelled engine. Despite the spectacular increase of the fleet in recent years (+60% between 1990 and 1997), the fleet’s total mileage remains almost constant. The average mileage per vehicle is less than half that of passenger cars. This reflects the fact that motorcycles in Belgium are mainly used for recreational purposes and not for commuting. It is estimated that only about 1% of all passenger.kilometres in road transport can be attributed to powered two-wheelers.
In this report we only address vehicles with 4-stroke engines larger than 50cc. Two-stroke motorcycles with an engine capacity over 50 cc are negligible and are not included in our analysis. Two-stroke engines and mopeds (<50cc) generate emissions with a very different composition. Although their effects (e.g. due to high VOC emissions) are potentially very important, emission factors are neither accurate nor complete enough to attempt a quantification of environmental externalities. Vito has therefore started an explorative study that should eliminate this gap in current knowledge.

We will first discuss marginal external costs and assess which emissions are dominant and which measures could be taken for the abatement of air pollution by motorcycles. Our second objective is to compare the environmental performance of motorcycles with cars and buses and address the question whether motorcycles are an alternative to cars and buses from an environmental perspective. In contrast to other road vehicles, we have not calculated an aggregated estimate of external cost from motorcycles. It was deemed that the necessary data on mobility were lacking. In contrast to data for passenger cars, no official figures are available on yearly activity of motorcycles and mopeds. Mileages from motorcycles as reported in MEET for Belgium (3000 km) are very different from those for Austria (7800 km) and France (6500 or 9000 km). According to European statistics, the average mileage of Belgian motorcycles has decreased by 30% since 1990. Their estimate (6200 km/y) is very close to Vito's assessments of motorcycle mileage (6390 km/y; De Vlieger, pers. com.) which are therefore believed to yield the best estimates of motorcycle mileage. Nevertheless a reliable estimation of the total cost for motorcycles seems impossible. In addition there is no info on two-stroke motorcycles (neither emissions nor data on the fleet and its composition).

Calculation of emission factors for motorcycles

Emission factors for motorcycles we calculated with the speed dependent functions from MEET. MEET distinguishes controlled and uncontrolled motorcycles, but differentiates only the uncontrolled vehicles in three classes according to cylinder capacity (<250cc, 250-750cc, >750cc). The speeds that were used for the calculations in this paper are given in Table 7.

Table 7: Average speeds (km/h) for different vehicle types and road types under normal and dense traffic conditions

Vehicle type	Average speed (km/h)		
	urban dense / normal	rural dense / normal	highway dense / normal
Passenger car	*15 / 22*	*25 / 51*	*25 / 110*
Public bus	*11 / 15*	*25 /45*	*25 / 80*
Motorcycle	*15 / 25*	*25 / 51*	*25 / 110*
Moped	*15 / 25*	*25 /31*	*25 / -*

It is assumed that speeds are similar for passenger cars and motorcycles in rural driving conditions and on highways, but it cannot be ignored that motorcycles have a slightly higher average speed in urban traffic.

2.3.3.1.e. Results

Traditionally motorcycles are considered to be cleaner vehicles than cars. Very often the lower fuel consumption is cited as one of their major environmental advantages. In Figure 13 we have illustrated the environmental damage costs of uncontrolled motorcycles in different driving conditions. It is clear that motorcycles do perform better from an environmental perspective when compared with uncontrolled passenger cars. The main impacts however are not caused by CO_2 or global warming (which is directly linked to fuel consumption). The adverse health effects of nitrates, particles (PM2.5) and ozone are much more important

Nitrates are formed from NOx emissions by chemical reactions in the atmosphere. Their effect is mainly regional (Europe-wide), and therefore little difference is found between the three locations shown in Figure 13. Particles on the other hand have a local effect and are found to dominate in urban locations, where population densities are highest.

Uncontrolled motorcycles have lower emissions of NOx and PM2.5 than uncontrolled (pre-catalyst) cars. This explains why the impacts are generally 20-50% lower. There is an interesting parallel between the emissions and the size of the engine. Heavy motorcycles (that were designed for higher speeds) have lower emissions in highway driving. Smaller engines (<250cc) which are often used to power scooters have lower emissions at low speeds (e.g. in urban traffic)

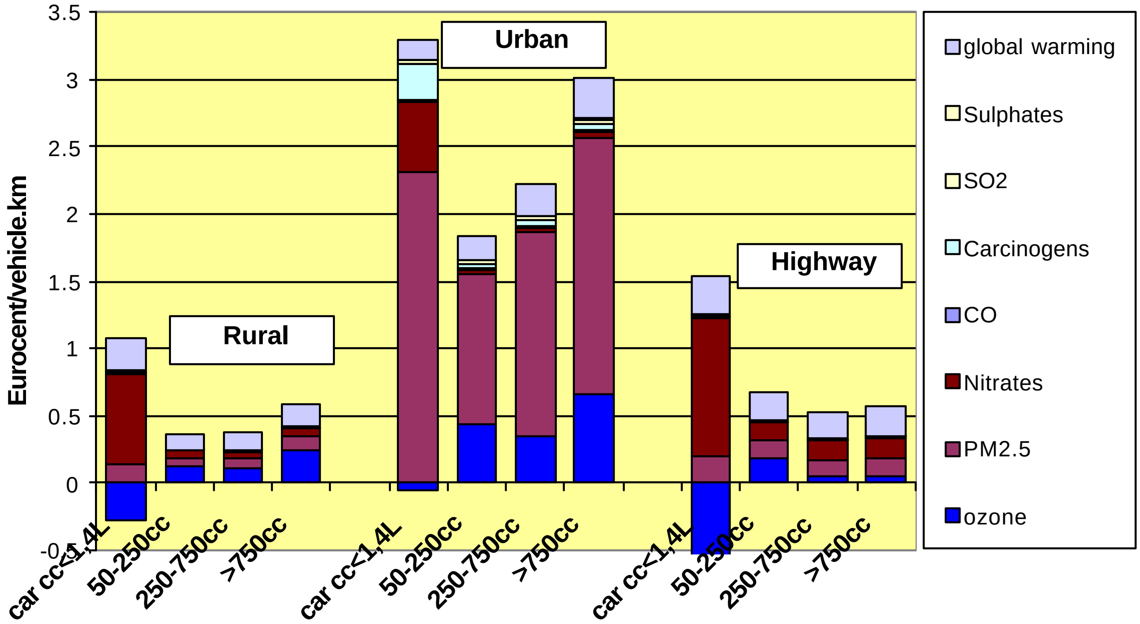


Figure 13 : **Environmental external costs of uncontrolled motorcycles compared to small petrol cars in different locations. Global Warming 16€/ton CO2 as sensitivity.**

Emissions of VOCS from motorcycles are relatively high. This leads to the formation of ozone that can cause significant impacts. Compared to cars, the O3-forming effect of the VOCS is larger than the (local) depressing effect of high NOx emissions on O_3 formation. Therefore ozone impacts are positive for uncontrolled motorcycles.

Figure 14 shows the results for “controlled” motorcycles, compared to common passenger cars. It is clear at first glance that motorcycles have lost their environmental advantage over cars. Only in large cities they have slightly lower externalities than cars.
Although tighter emission standards for motorcycles have been set in 1999, they have not achieved a similar effect on external costs as in passenger cars. Most significant reductions where those of VOCS and particles. However compliance with the emission standards did not necessitate the introduction of three-way catalytic converters as in passenger cars. As a result emissions of NOx from motorcycles have not decreased.

Taking into account the January 1st 2000 introduction of EURO 3 emission standards for cars, the low penetration of controlled motorcycles and the number of passengers per vehicle; we conclude that motorcycles are now environmentally outperformed by cars.

The reduction of NOx from tail-pipe emissions has proven to be the single most important technological achievement in lowering external costs in passenger cars. Despite the success of catalysts in cars, motorcycles with catalysts are rare and implementing them involves technological problems unlike those in passenger cars 10 years ago. It is therefore unlikely that EC directives will impose emissions standards that force the introduction of catalysts in the near future.

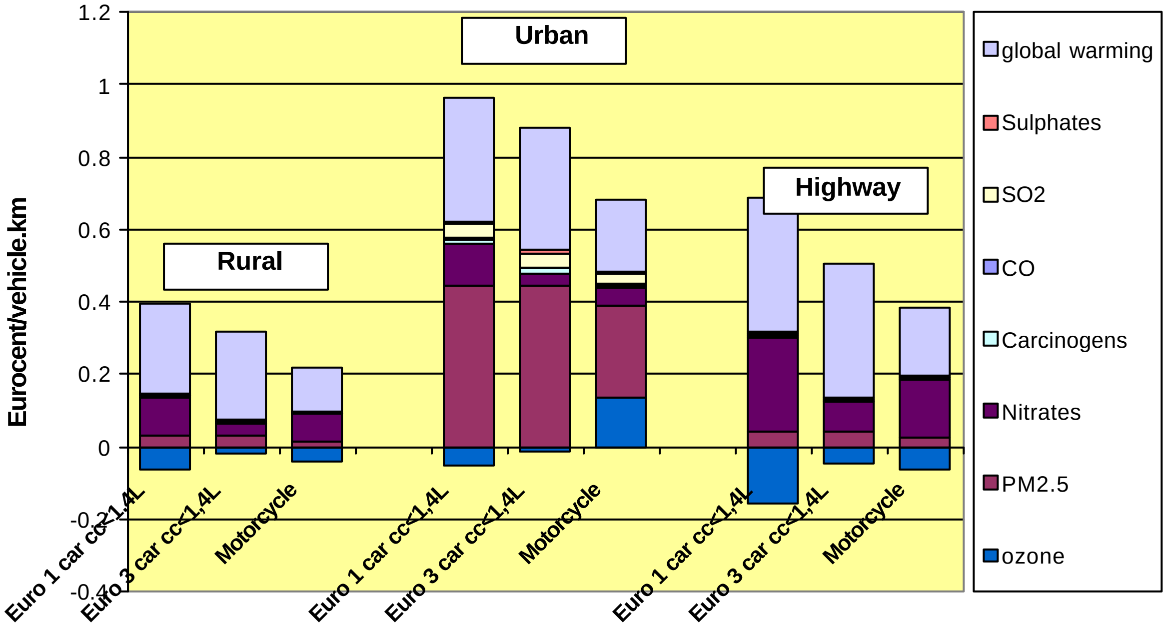


Figure 14: Environmental external costs of controlled motorcycles compared a small petrol car in different locations. Global Warming 16€/ton CO2 for sensitivity.

Only few major manufacturers can offer a motorcycle with a catalyst and some withdrew their model because of disappointing sales. Given the relative importance of nitrate impacts in rural and highway driving, it is expected that motorcycles could regain their favourable environmental image with the mandatory introduction of three-way catalytic converters. At that point, the lower fuel consumption of motorcycles may again make a difference.

In urban peak traffic, when speeds are low, NOx emissions are also much lower. Combined with the large population at risk, we find that local health effects of primary particles dominate externalities (see Figure 13 and Figure 14). The reduction of particulate emissions in controlled motorcycles has been adequate to ensure that impacts per vehicle.kilometre are lower than for passenger cars in this specific case (Figure 15).

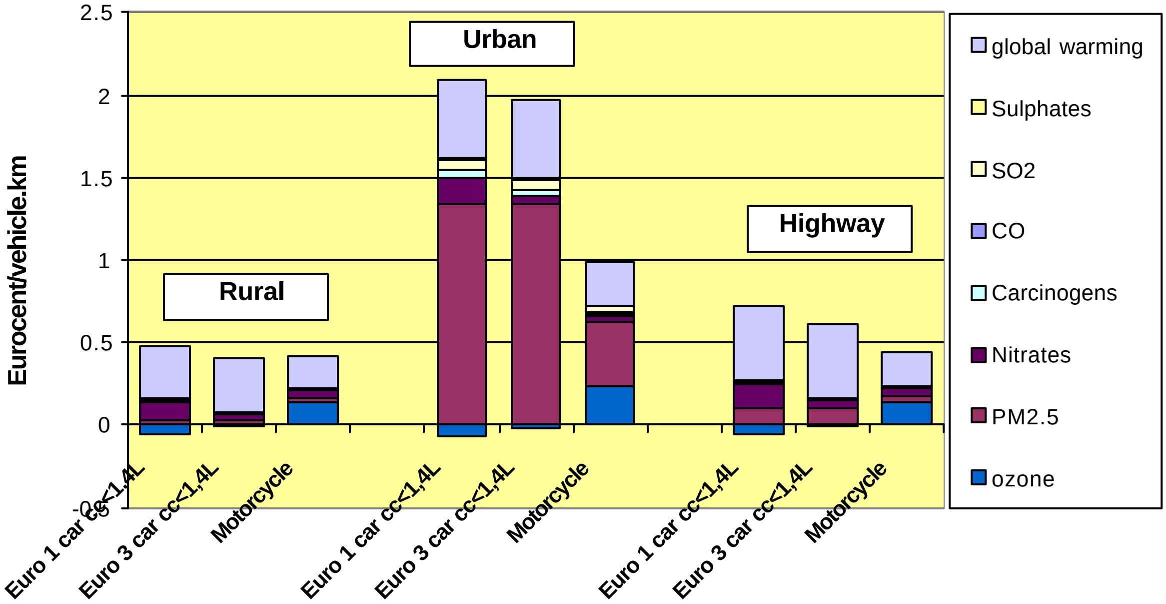


Figure 15: Air pollution impacts from urban peak traffic. Global Warming 16 €/ton CO2

2.3.3.1.f. Comparison with other modes

This apparent advantage of motorcycles in urban peak traffic quickly evaporates when occupancy rates are taken into account. Obviously passenger cars and public buses can carry many more passengers than a motorcycle. Although the average occupancy of cars and buses is low (1.3 and 15 passengers respectively), these could be increased significantly (e.g. in peak traffic). Policy makers may therefore consider that promoting car-pooling and public transport with buses are just as effective for reducing externalities as a modal shift to motorcycles.
It would be misleading to compare motorcycles with other transport modes in terms of impacts by tail-pipe emissions only. In urban peak traffic, air pollution is but one of several causes of external costs. Other possible externalities include Life Cycle Impacts (LCI) as well as impacts from noise, accidents and congestion. Despite the severe lack of useable data from literature, we have attempted to create a graph with preliminary estimates for some of these impacts. Life Cycle impacts and external costs of noise were included in Figure 16 to provide a comparison with use-phase externalities but we refer to paragraphs 2.3.3.3 and 2.3.3.4.b for a full discussion.

Air pollution impacts from tailpipe emissions are usually the most important environmental costs. External costs resulting from the production of motorcycles and their fuel are lower than for cars because of their low weight and low fuel consumption (based on ExternE data from IER, Bickel pers. comm.). An increase of the average occupancy rate of cars (car-pooling) however can achieve a similar reduction in the costs per passenger.km. For buses, air pollution costs are much larger than other externalities.

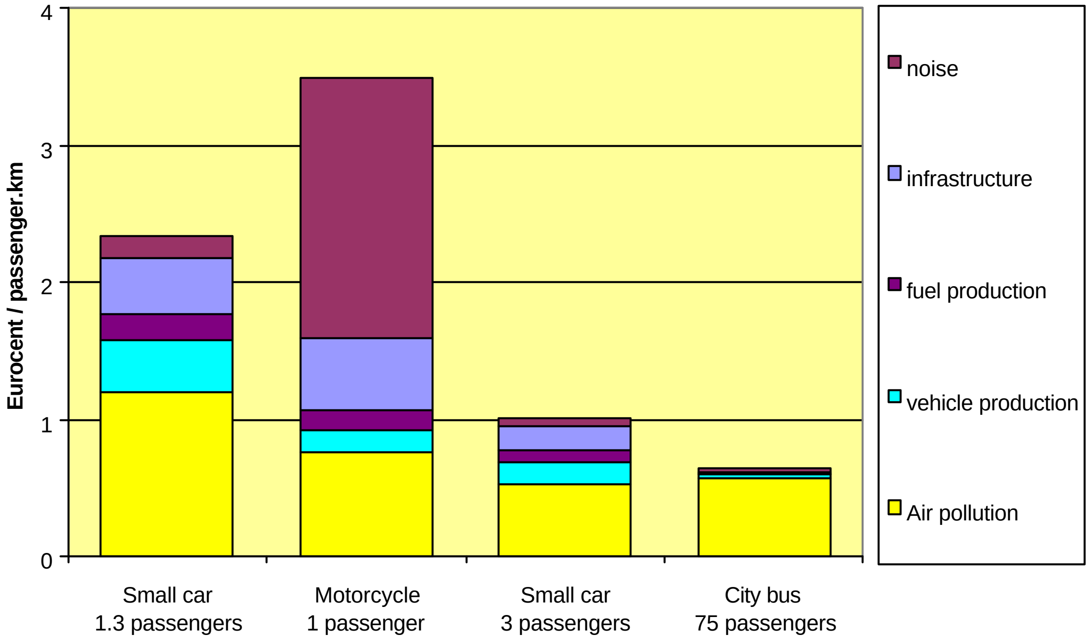


Figure 16: Summation of different types of externalities for different means of urban passenger transport (peak traffic, Eurocent/passenger.km)

We have found no applicable literature on the external costs of noise from motorcycles. European emission standards are now at the same level as for heavy lorries (80 dB). Therefore it can be expected that noise externalities will be at least 10 times higher than for cars. This means that noise may be the major impact from motorcycles in urban traffic (based on data from Mayeres and Van Dender, pers. com.).

Two other types of (non-environmental) external costs are derived from the interaction of motorcycles with other vehicles in real life traffic situations: external accident costs and external congestion costs. The external accident cost (the risk that someone else gets injured) for motorcycles appears to be much higher than for other vehicles. Recent studies shown that this cost may amounts to 0.17 Euro/km, dwarfing the externalities shown in Figure 16 (based on data from Mayeres, pers. com. see elsewhere in this report). The main uncertainty however lies with external cost of congestion. None of the recent studies of externalities have addressed the specific impacts of motorcycles. For cars, these costs are by far the most important category (up to 30 times as high as all other costs combined in urban peak traffic). Unfortunately, there is no obvious relationship between the external congestion costs of cars and motorcycles.

2.3.3.1.g. Buses

Introduction

Diesel fuelled buses have fairly high externalities per kilometre. This is due to the low average speed and accentuated by the fact that urban driving accounts for 65% of the mileage in public buses. On the other hand the yearly mileage of buses is quite low compared to other vehicles (100 times less than passenger cars) so that their contribution to the total transport externalities is insignificant at this time.

Marginal externalities of buses compared to passenger cars

The external costs per vehicle.km of diesel buses vary from 95 Ect/vkm for EURO0 buses to 30 Ect/vkm for present-day buses and 21 Ect/vkm for future EURO3 buses. These figures apply for big cities and normal urban traffic conditions. The positive trend is due to a decrease in particulate emissions and, to a lesser extent, lower NO_x emissions. As particulate emissions are responsible for a major portion of the externalities, it is expected that the introduction of PM-filters or alternative fuel buses (CNG, LPG) would lead to significantly lower results. For urban transport, it is interesting to compare results for passenger cars and public transport. At this point we limit the comparison to large diesel busses, as they take up the major part of the present-day fleet of public transport vehicles.
In order to compare different transport modes, external costs should be expressed in terms of passenger.kilometre instead of vehicle.kilometre. Consequently, as shown in Figure 17, seat occupancy rates are a predominant factor of public transport externalities. The straight dotted line indicates the fixed external costs of an average passenger car with an occupancy of 1.3 persons per car. Early in the project we have shown that present-day EURO2 buses perform better with respect to recent diesel cars if the occupancy rate is more than 15%. In order to have lower externalities than recent petrol cars, occupancy rates should at least be 50%. However widely different rates (between 5% and 60%) were obtained when comparing different engine technologies. Therefore no unequivocal result could be presented and the resulting data were not useful for policy makers. By combining data on external costs and the actual composition of the fleets of passenger cars and city buses, we created the graph shown in Figure 17. If the occupancy rate of diesel busses is higher than 25%, the environmental costs of public transport are lower.

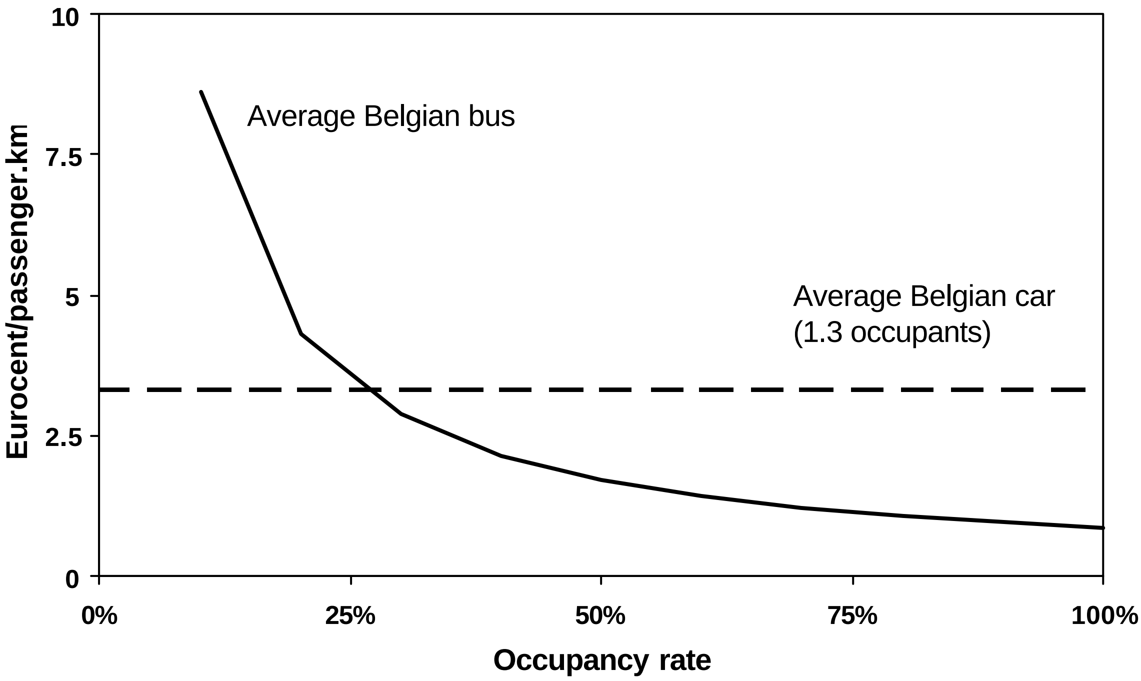


Figure 17: 1998 comparison of public and private passenger transport in a large Belgian city

A similar comparison in a situation with congested traffic would lead to a similar conclusion, as emissions for bus and car approximately increase with the same factor with respect to urban non-peak driving. As occupancy rates of public transport are usually quite high during

peak hours, modern buses should perform quite well in peak periods. However, at some times occupancy rates are well below 25% (the average being about 15%) so that the environmental cost of public transport with diesel buses in Belgian cities is higher when compared to private transport.
At this moment we have no reason to believe that new technologies are being introduced faster in the fleet of city buses than in the fleet of private passenger cars. Therefore we don't expect the relative position of both curves to change over time. In addition the conclusion proved to be robust to the changes in the methodology described in paragraph 2.3.1.2 (see De Nocker et al., 1999).

2.3.3.1.h. Total externalities

The necessary data to aggregate marginal externalities to the national level is much harder to find than for passenger cars. The mileage of buses for public transport is based on information of the Brussels and Flemish bus operating companies. Values of the Flemish company 'De Lijn' show an average mileage of about 43000 km/y for there own buses. The mileage of buses of operators driving for De Lijn is not known (De Lijn, 1996). The mileage driven in Belgium by coaches is known from the Institute for coaches and buses (ICB). For coaches the mileage driven in Belgium for the years 1990 and 1998 is 24710 km and 22940 km respectively. While figures for the Flemish Region show that the total mileage of public buses is more or less constant during the period 1990-1996, an average yearly change in activity of -0.7% was found for coaches. These mileages were distributed over the different locations as shown in Table 8.

Table 8: Overview of preliminary mileage distribution over different road types

Vehicle type	Mileage distribution %		
	urban	rural	highway
Public bus	*65*	*35*	*0*
Coaches	*25*	*37,5*	*37,5*

The total external cost caused by the combined fleets of public buses and private coaches is 172 million Euro per year. Given a total mileage of 610 million kilometres the average impact is 28 Eurocent/vehicle.km. The precise impact per passenger kilometre depends on the seat occupancy rate in different locations. At the aggregated level it is difficult to make assumptions about this rate. However it can be shown that public buses perform better (environmentally) than passenger cars in large cities when seat occupancy rates are at least 25% (De Nocker et al., 1999).
Buses and coaches account for 1 % of the total mileage and a 7 % share in the total externalities in 1998. The evolution of the total external cost and mileage is shown in Figure 18. Despite the enormous mileage growth, externalities have begun to decrease since 1996. The turning point coincides with a reduced growth of the fleet mileage in 1997 and 1998. Therefore the pattern of changing external costs completely reflects the important changes in new bus sales.

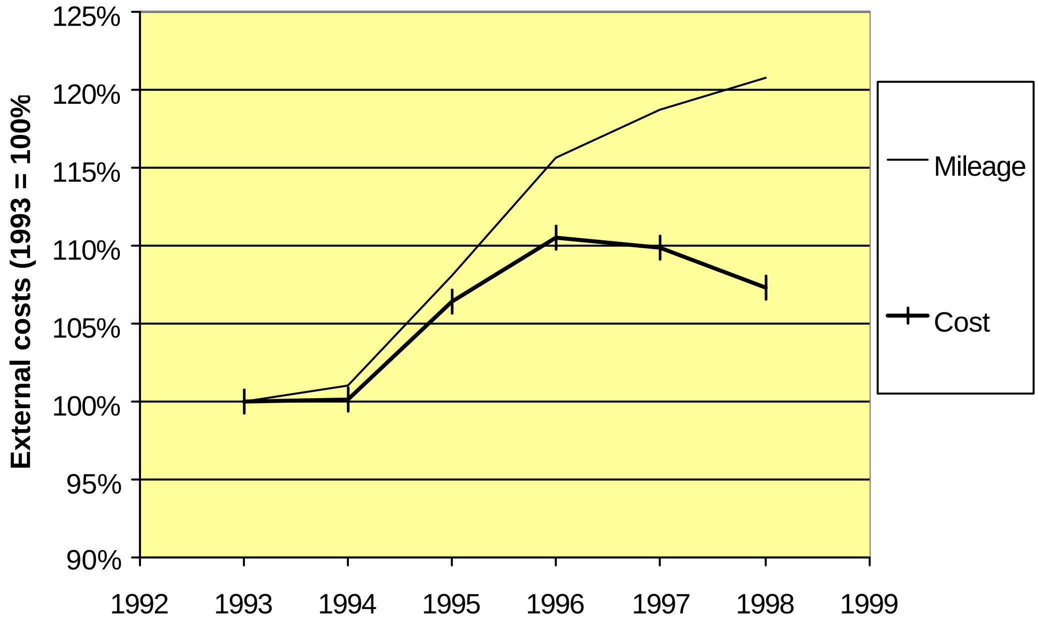


Figure 18: Evolution of total mileage and external costs of buses and coaches

2.3.3.2. Use phase impacts from freight transport

2.3.3.2.a. Introduction

In this paragraph we present estimates for the external costs of heavy-duty vehicles. The external costs of different types of vehicles and technologies are compared while taking differences in capacity into account. We will first present marginal estimates for Heavy Duty Vehicles under different driving conditions. We will discuss which pollutants cause the dominant impacts and which technological options may be chosen to improve tailpipe emissions. Then we discuss an attempt to estimate the total external cost, which highlights possible policy measures aimed at fleet and activity related variables for the abatement of air pollution.

2.3.3.2.b. Marginal externalities

A summary of environmental damage costs for heavy-duty vehicles in rural areas can be found in Figure 19. Despite the fact that large lorries have higher tailpipe emissions than light trucks, their impact per tonne of freight transported is lower. Therefore it seems most efficient to use large trucks for long range transport. Impacts of HDV on highways are comparable to those shown in Figure 19. A similar pattern is also found in urban areas, but externalities are more than 10 times higher in cities.

Preliminary results for freight trains and inland ships were included for comparison. They may provide an environmentally preferable alternative for road transport. Results in paragraph 2.3.4.2.d focus on comparing these modes with trucks by using existing railway and canal trajectories as alternatives for certain highways.

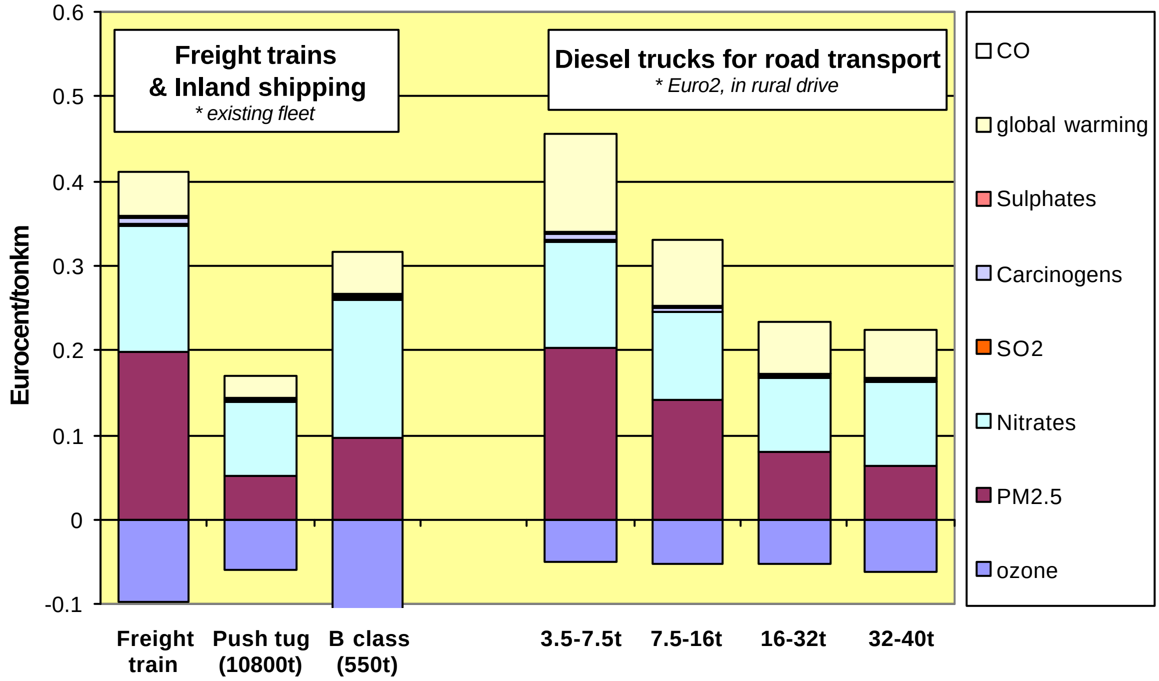


Figure 19: Marginal external costs of different vehicles for freight transport. Global Warming based on 16€/t CO2 as a sensitivity.

Impacts are dominated by the adverse health impacts of particulate emissions for the local population (hence the important location impact). This is typical of diesel engines. The second most important impact is caused by the long-range health impacts of nitrate aerosols. The effects of PM2.5 and NOx emissions dwarf all other impacts including global warming. Marginal impacts of ozone are thought be negative in Belgium because of the (local) depressing effect that high additional NOx emissions currently have on ozone formation.

2.3.3.2.c. Aggregated results

The NIS statistics show that the average mileage driven in Belgium for HDV (all weight classes) was 38078 in 1996 and this number increases by 2.3% annually. Vito's assessment of mileage per weight class is based on the NIS statistics. Because there is no uniformity in the category split of different statistics only rough estimations can be made. In addition, our mileage data only refers to Belgian trucks on Belgian roads. Despite the obvious fact that a significant proportion of trucks on Belgian roads is foreign, there is no information on the fleet composition, let alone a road type distribution of their mileage.
The mileage distribution of heavy-duty vehicles and buses over different road types was taken from Vito's assessment because other sources gave widely different estimates. Most important for the comparison with passenger cars is the fact that HDV drive about half of their mileage on highways and only 10% in cities.

The total external cost of Belgian trucks in Belgian traffic is approximately 640 million Euro (Table 9). Although heavy-duty vehicles only account for 7% of total mileage, they cause 27% of the annual impacts from road transport. Other heavy-duty vehicles such as buses and coaches only account for 1% in total mileage and a 7% share in the total externalities. From this result it is clear that policy measures, often aimed at (some types) of passenger cars should not ignore impacts from heavy-duty vehicles for freight.

Table 9: Total external costs of the use phase for different segments of the fleet (in billion Euro)

	External costs			Mileage		
Year	Pass. cars	HDV	Buses	Pass. cars	HDV	Buses
1993	1.96	0.62	0.16	67.03	5.3	0.51
1994	1.91	0.64	0.16	69.14	5.5	0.51
1995	1.86	0.65	0.17	70.87	5.7	0.55
1996	1.77	0.65	0.18	72.53	5.8	0.58
1997	1.68	0.65	0.18	74.76	6.1	0.60
1998	1.59	0.64	0.17	77.10	6.3	0.61

2.3.3.2.d. Evolution of the total environmental costs of HDV in Belgium

We have repeated the aggregation for all years between 1993 and 1998 (Figure 20). In this way the net result of all transport related policy measures (European and national) in this time frame can be evaluated. During this period the changing composition of the fleet and the mileage were taken into account. The composition of fuels and mobility distribution data were kept constant.

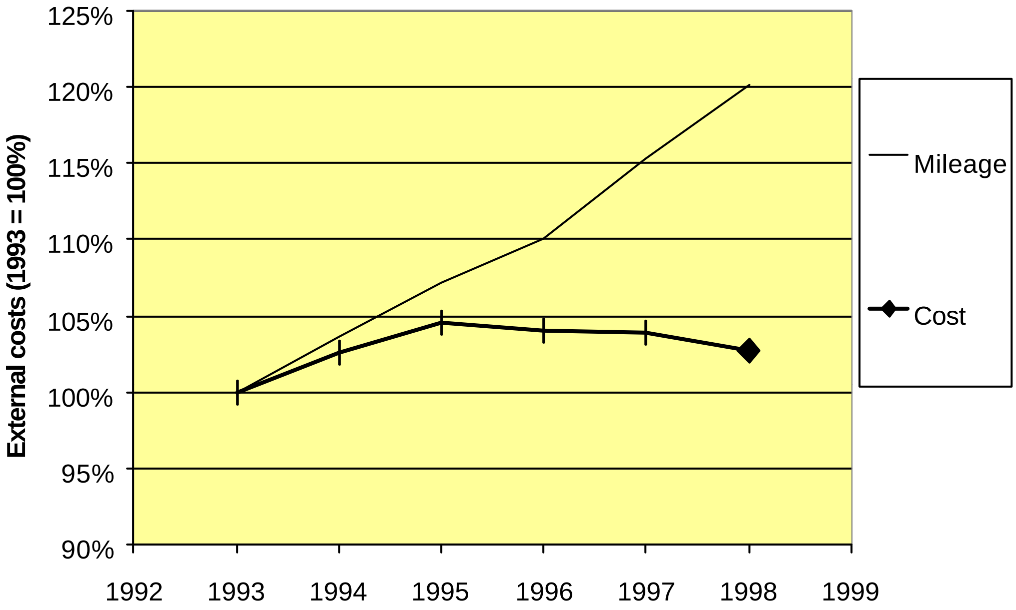


Figure 20: Evolution of total mileage and external costs of HDV

Although the cost of air pollution from Belgian road transport has been decreasing every year between 1993 and 1998, this is almost entirely due to the reduction of environmental impacts from passenger cars. Total externalities from HDV have only begun to decrease recently. This trend is clearly different from passenger cars. There has been no steady decline. An initial rise has been followed by a slight decline, which first started in 1996. Whereas the technological progress for passenger cars appears sufficient to compensate for the increased mileage, growth of the fleet and trend towards diesel, this was initially not the case for trucks or buses. Eventually, the interaction between fleet size and fleet composition has resulted in a divergence between the trends of externalities and fleet mileages.

For trucks there has been an important growth, both in terms of the number of vehicles as in average mileage per vehicle. This combined increase leads to a rise of the total fleet mileage (approximately 20%) that is much higher than for passenger cars (15%). Two phases can be distinguished in Figure 20. Between 1993 and 1996 the growth of the fleet was rather slow, but external costs were rising. In 1997 and 1998 the growth of the fleet was much more important, explaining the accelerating increase of the fleet mileage. On the other hand, the fast growing fleet now has acquired a significant proportion of young, environmentally better, EURO2 trucks. And since 1996 this technological progress seems to compensate for the growth in the road transport sector. Nevertheless the environmental performance of the Belgian fleet seems disappointing when compared to the UK (Int Panis et al., 2000). Externalities have fallen consistently since 1990 in the UK and fallen by 35% between 1993 and 1998. In contrast, externalities have risen in Belgium. A small part of this difference can be attributed to the way that traffic growth is distributed across urban, rural and motor-way locations, but the fact remains that externalities in Belgium have remained relatively constant at best. When looked at within Belgium, the obvious conclusion is that this arises because of the very large growth in total HDV mileage. However the total mileage was very similar in both countries and UK-mileage growth was only marginally lower than in Belgium. The answer to the differences must therefore lie in differences in the fleet composition and evolution.

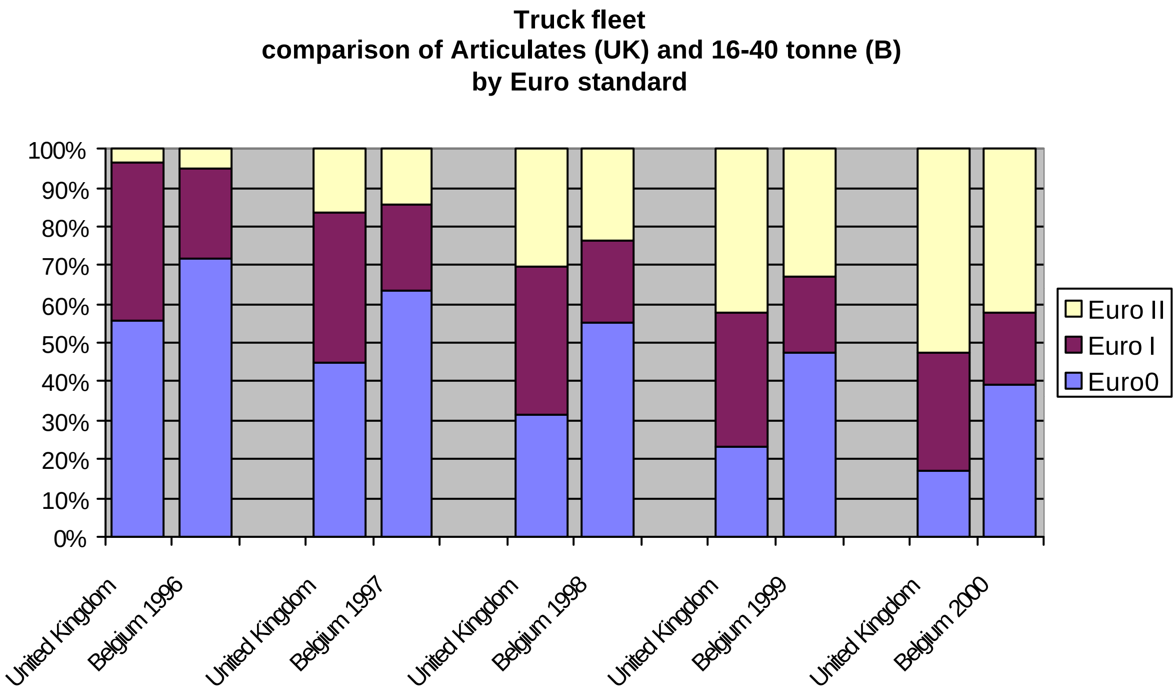


Figure 21: **Introduction of new (EURO1 and EURO2) heavy trucks in the UK fleet as compared to Belgium (1996-2000)**

In both countries there has been a move to heavier goods vehicles in recent years due to major changes in the haulage industry. This has led to differences in the environmental performance between different weight classes of goods vehicles. Most recent sales of goods vehicles have been towards larger articulated vehicles. A greater proportion of these heavier vehicles therefore complies with EURO 1 or EURO 2 emissions standards. However, the fleet turnover and the degree of switch to heavier vehicles has been greater in the UK. This can be seen in Figure 21. Newer types of trucks are introduced much faster in the UK. By extrapolation it is found that over 50% of all UK articulate trucks will comply with EURO2 emission standards in 2000. Moreover, older, uncontrolled, trucks (EURO0) are scrapped much faster in the UK and as these vehicles account for the bulk of externalities, it is their fraction that is most important for the decrease. These differences result in the differences in total externalities over time. It can be concluded that emissions legislation has been important in driving the reductions in externalities in the transport sector in both countries, though the effect of operating heavier vehicles (driven by economics) is more important. Stricter emission standard do not always lead to externalities reductions per se in all European countries, as the Belgian fleet shows.

2.3.3.3. Life Cycle costs

Studies on transport externalities have focused primarily on estimating externalities from the actual use of the vehicle. This is in accordance with the generally accepted view that the use phase has the most severe environmental consequences. Up- and downstream processes however, may also place heavy burdens on human health as well as on the natural and man-made environment. From a scientific point of view this necessitates the study of the whole life cycle of transport in the analysis. More specific: the fuel cycle, the life cycle of the vehicle and its infrastructure have to be examined as well.

Earlier studies indicated that non-use impacts, although not predominant compared to the use of the vehicle, are certainly not negligible. Secondly, society and governments are concerned about some non-use environmental burdens, like the disposal of vehicles, which is e.g. subject of European legislative initiatives (CEC 97/0194). Also, the importance of the up- and downstream processes varies over different transport modes, so a correct intermodal comparison should include these differences. Another reason for looking into these up- and downstream processes, is that, in the near future, environmental impacts of the use-phase could decline due to technical an regulatory innovation (e.g. EURO 2 and 3, lower fuel consumption,…) with respect to the other stages of the life cycle. In this perspective, up- and downstream processes could get relatively more important from an environmental point of view. By examining the whole life cycle, it becomes feasible to review trade-offs between different life cycle stages and/or different impact categories.

2.3.3.3.a. Overview of existing studies (1997)

As a first step a literature survey has been performed concerning LCA studies in the transport sector. The following conclusions can be drawn from the examination of the study ‘Overview of LCA studies in the automotive sector’ (IPTS, 1996).

- since 1990 more than 50 studies (in particular in Europe) were performed; all car manufacturers are involved in this kind of activity
- the most ‘popular’ topics are, in descending order of importance :
  - fuel (life cycle comparisons of diesel, gasoline, biofuels, LPG, natural gas,…)
  - materials and car parts (e.g. bumpers, Al. vs. Steel body)
  - engines and cars (few for the latter)
  - ‘end of life’
- quite some studies are not publicly available (owned by car manufacturers)

VITO has performed a comparative LCA of biodiesel and fossil diesel fuel (Spirinckx et al., 1999; De Nocker et al., 1998). The environmental profiles of the 2 automotive fuels are compared for the different impact categories. A remarkable result of the study is that the biodiesel life cycle only scores better on the impact indices ‘use of fossil fuels’ and ‘greenhouse effect’.

Large scale LCA work has been performed by the IKP of the University of Stuttgart (Eyerer et al.,1996) and in the EUCAR-LCA project (Kaniut et al., 1996), which was launched in 1993 by several European car manufacturers. Objectives of the latter project are:

- to work out a common LCA methodology
- to define practical ways for ‘Design for environment’ for new car parts
- to build a reliable database

Phase 1 (case study on a car bumper) ended in 1995 but the results are very disperse due to methodological and implementational differences. As a result, priorities for future work are focused on methodology and database harmonisation, the influence of weight on fuel consumption and a joint model on fuel emissions of engines.

It is important to note that the stated studies are concentrated on either particular life cycle stages, particular car parts or particular technologies, which limits their usability in the present study. They can be used as an additional information source when very detailed information is relevant. For the moment, no European LCA program on a generic car is foreseen, whereas 2 similar projects are set up in the US.

Concerning the life cycle of railway and waterway life cycle, no specific studies were found. This is probably due to a lack of data availability and to the fact that they are generally acknowledged as environmentally friendly. Due to the problems regarding data availability and data complexity only few studies to date consider construction and maintenance of infrastructure. This appears to be an important shortcoming, as the studies, which did analyse these process steps, suggest that the emissions are far from being negligible, particularly for rail transport (cf. next paragraph).

Few studies consider the whole process chain of transport. ISV (1997) analysed the whole process chain for specific goods transport relations in Baden-Würtemberg. Ökoinventar Transporte (Maibach et al., 1995) is a general emission inventory of all transport life cycle stages. It can be considered as an important exception to the general consideration that few useful information, in particular for railway and waterway transport and infrastructure, is available.[3]. In order to fill a gap in data, we did a separate LCA study for inland waterways.

This study also covers the emissions to the water and soil, and a detailed analysis has shown that the non-use phase can be important for these impact categories. However, it was not possible to fully apply the impact pathway to these emissions, and thus not possible to calculate the external costs.

A further analysis has focussed on the relative impacts of site specificity to assess impacts from the fuel production life cycle. This analysis has shown that it is important to separately take into account the impacts from emissions outside Europe, and especially offshore. (Torfs,

[3] Also, it is noted that the ETH database (Frieschknecht et al., 1996), which is used for providing basic emission data in LCA studies, incorporates data on transport tasks, which are the result of Ökoinventar Transporte.

1999) However, the results of this case study cannot be generalised for all the emissions from the non-use phase of transport.

For reasons of consistency, the discussion in this report is limited to the external costs of air borne pollutants from the non-use phase, based on emissions from the ExternE network.

2.3.3.3.b. Road Vehicle production and fuel cycle

Emissions resulting from the production of road vehicles and several fuels are shown below. For manufacture, estimates for LDVs, HDVs and buses are scaled from the data on car manufacture based on weight and assume the following lifetime vehicle km (based on MEET estimates).

Passenger Car	150 000 km
LDV	200 000 km
HDV	600 000 km
Bus	390 000 km

The values for fuel production are based on the following fuel consumption assumptions.

	Energy (MJ/km)			
	Car*	**LDV***	**HDV****	**Bus****
Gasoline	2.70	4.60	-	-
Diesel	2.10	3.80	9.52	14.92
LPG	2.67	4.45	9.57	16.43
CNG	2.73	4.60	10.15	15.83

* based on bi-fuel TWC. ** based on lean burn diesel conversion.

Passenger Car		**g/km**						
		CO_2	**CO**	**NO_x**	**NMHC**	**SO_2**	**CH_4**	**PM**
Gasoline	Manufacture	66.31	0.025	0.195	0.10	0.60	0.092	0.019
	Fuel prod	25.11	0.0138	0.114	0.57	0.181	0.047	0.0065
	Total	**91.42**	**0.039**	**0.31**	**0.67**	**0.78**	**0.14**	**0.026**
Diesel	Manufacture	66.31	0.025	0.195	0.10	0.60	0.092	0.019
	Fuel prod	14.28	0.0097	0.076	0.18	0.100	0.033	0.0023
	Total	**80.59**	**0.035**	**0.271**	**0.28**	**0.70**	**0.14**	**0.021**
CNG	Manufacture	68.59	0.025	0.198	0.10	0.61	0.092	0.019
	Fuel prod	11.19	0.0038	0.029	0.075	0.046	0.61	0.0022
	Total	**79.78**	**0.029**	**0.227**	**0.172**	**0.657**	**0.70**	**0.021**
LPG	Manufacture	66.68	0.025	0.195	0.10	0.60	0.092	0.019
	Fuel prod	16.55	0.011	0.089	0.15	0.094	0.045	0.0043
	Total	**83.24**	**0.036**	**0.285**	**0.25**	**0.695**	**0.14**	**0.023**

Light Duty Vehicle		**g/km**						
		CO_2	**CO**	**NO_x**	**NMHC**	**SO_2**	**CH_4**	**PM**
Gasoline	Manufacture	99.47	0.0378	0.2921	0.1451	0.90	0.1377	0.029
	Fuel prod	42.78	0.023	0.20	0.97	0.31	0.080	0.011
	Total	**142.25**	**0.061**	**0.49**	**1.12**	**1.21**	**0.218**	**0.040**
Diesel	Manufacture	99.47	0.0378	0.2921	0.1451	0.90	0.1377	0.029
	Fuel prod	25.84	0.017	0.14	0.33	0.18	0.060	0.0042
	Total	**125.31**	**0.055**	**0.43**	**0.48**	**1.08**	**0.198**	**0.033**
CNG	Manufacture	102.88	0.038	0.297	0.15	0.92	0.138	0.029
	Fuel prod	18.86	0.006	0.049	0.13	0.078	1.03	0.0037
	Total	**121.74**	**0.044**	**0.35**	**0.27**	**0.99**	**1.167**	**0.032**
LPG	Manufacture	100.02	0.038	0.293	0.15	0.90	0.138	0.029
	Fuel prod	27.59	0.018	0.15	0.25	0.16	0.075	0.0071
	Total	**127.61**	**0.056**	**0.44**	**0.40**	**1.06**	**0.212**	**0.036**

Heavy Duty Vehicle		**g/km**						
		CO_2	**CO**	**NO_x**	**NMHC**	**SO_2**	**CH_4**	**PM**
Diesel	Manufacture	198.94	0.076	0.58	0.29	1.80	0.28	0.057
	Fuel prod	64.74	0.044	0.34	0.84	0.45	0.15	0.010
	Total	**263.68**	**0.119**	**0.93**	**1.13**	**2.25**	**0.43**	**0.068**
CNG	Manufacture	205.76	0.076	0.59	0.29	1.83	0.28	0.057
	Fuel prod	41.62	0.014	0.11	0.28	0.17	2.27	0.008
	Total	**247.38**	**0.090**	**0.70**	**0.57**	**2.00**	**2.55**	**0.066**
LPG	Manufacture	200.05	0.076	0.59	0.29	1.80	0.28	0.057
	Fuel prod	59.33	0.039	0.32	0.54	0.34	0.16	0.015
	Total	**259.38**	**0.115**	**0.91**	**0.83**	**2.14**	**0.44**	**0.073**

Large Urban Bus		**g/km**						
		CO_2	**CO**	**NO_x**	**NMHC**	**SO_2**	**CH_4**	**PM**
Diesel	Manufacture	255.05	0.097	0.75	0.37	2.31	0.35	0.074
	Fuel prod	101.46	0.069	0.54	1.31	0.71	0.24	0.016
	Total	**356.51**	**0.166**	**1.29**	**1.68**	**3.02**	**0.59**	**0.090**
CNG	Manufacture	263.80	0.097	0.76	0.37	2.35	0.35	0.074
	Fuel prod	64.90	0.022	0.17	0.44	0.27	3.54	0.013
	Total	**328.70**	**0.119**	**0.93**	**0.81**	**2.62**	**3.89**	**0.086**
LPG	Manufacture	256.47	0.097	0.75	0.37	2.31	0.35	0.074
	Fuel prod	101.87	0.067	0.55	0.93	0.58	0.28	0.026
	Total	**358.34**	**0.164**	**1.30**	**1.30**	**2.89**	**0.63**	**0.100**

These emissions from all three upstream processes were multiplied by country specific damage costs. These costs were estimated for runs with the multisource EcoSense model for different European countries. Since the location, stack-height etc. of the upstream emissions is unknown we have tried to estimate averages in a consistent way for all European countries. Values for Belgium are presented in Table 10.

Table 10

Euro/t	Health	Crop losses	Material damage	Total
NOx	3459	-660	73	2871
SO2	5942	-9	224	6157
PM10	13313			13313
NMVOC	888	829		1717

Costs per tonne for greenhouse gases are identical to those used for the marginal use-phase impacts.

2.3.3.3.c. LCA emissions for rail vehicles

Emissions for the manufacture of trains are slightly more complex than for road vehicles. For trains, emissions have to be summed up for the number of coaches plus locomotive. To this end data from the national railroad company NMBS/SNCF were used to estimate the number of coaches from the trains total average weight. All figures below are based on a process chain analysis, based on following specifications:

	Capacity	Kilometres per year	Lifetime in years	Weight (tonnes)
Electric locomotive	-	200000	40	87
Diesel locomotive	-	200000	40	72
Passenger coach	60	150000	40	48
Tram	242	90000	30	56

Manufacture Rail Veh.	**g/km**						
	CO_2	**CO**	**NO_x**	**NMHC**	**Benzene**	**CH_4**	**PM**
Electric locomotive	112	0.79	0.22	0.51	7.5e-4	0.36	0.093
Diesel locomotive	85	0.63	0.17	0.39	6.0e-4	0.28	0.077
Passenger coach	84	0.55	0.20	0.47	2.9e-4	0.27	0.070
Tram	219	1.89	0.44	1.06	5.7e-4	0.79	0.242

Fuel production values for diesel used in trains are identical to those of road vehicles. They are based on European average emissions from diesel fuel manufacture and transportation. The emissions are presented in terms of g/GJ (not g/km) because of the variation in fuel consumption for rail in different European countries.

CO_2 kg/GJ	**CO** g/GJ	**NO_x** g/GJ	**NMHC** g/GJ	**SO_2** g/GJ	**CH_4** g/GJ	**PM** g/GJ
6.8	**4.6**	**36.1**	**87.8**	**47.6**	**15.8**	**1.1**

2.3.3.3.d. Infrastructure provision

Road Infrastructure

We could find no relevant data for Belgium. Therefore we have followed other European teams in using emissions that were calculated based on data for specific roads in Baden-Württemberg (side roads, main roads and federal roads) and projected to Germany. The following materials and processes were included:

Earth work	energy input
Road crust	material input different road layers energy input construction of upper road layer
Supplies	material input crash barrier, lamp posts, sewerage, road signs, noise protection
Operation and maintenance of road infrastructure	material input constructional maintenance energy input traffic lights, street lighting, fuel consumption of service vehicles

As the processes involved mainly produce coarse particles, the share of fine particles (PM_{10}) was estimated from the process chain. The resulting average emissions per vehicle category are:

Infrastructure Road	**g/km**						
	CO_2	**CO**	**NO_x**	**NMHC**	**Benzene**	**CH_4**	**PM**
Car	20.72	0.129	0.047	2.96	0.7e-4	0.071	3.7e-3
Bus	33.39	0.207	0.075	4.76	1.1e-4	0.114	6.1e-3
HDV	77.39	0.476	0.175	10.94	2.6e-4	0.263	0.014

Railway Infrastructure

The Belgian national railroad company could not provide data on emissions resulting from the building or maintenance of tracks. Therefore we have again used emissions for Germany that were estimated for a mix of track types (wood, metal and concrete sleepers). It proved to be impossible to distinguish different types of tracks. The following materials and processes were included:

Earth work	energy input
Track	material input sleepers, rails, metal material (bolts etc.), ballast energy input rail installation
Supplies	material input signal and communication equipment material input catenary wire for electrified routes
Operation and maintenance of rail infrastructure	energy input heating of points energy input track maintenance for wood sleeper type

As the processes involved mainly produce coarse particles, the share of fine particles (PM_{10}) was estimated from the process chain. The resulting average emissions per vehicle category are:

Infrastructure Rail	g/km						
	CO_2	CO	NO_x	NMHC	Benzene	CH_4	PM
Passenger Train	319.3	4.27	1.37	2.56	0.9e-3	1.50	0.043
Goods Train	1663.7	22.26	7.12	13.34	4.8e-3	7.82	0.222

2.3.3.3.e. Electric powered vehicles

An average cost per kWh was derived for electrical energy used by trains and trams. This report contains estimates for electric rail vehicles only. The external cost of electricity stored in batteries with a limited life is not included since this would entail a detailed LCA study of different battery types. Given the low penetration and low expectations for electric road vehicles this would result in few, if any, useful policy recommendations. Therefore this was considered to be beyond the scope of this study.
For classical electricity use, the external cost depends on the assumed fuel used for electricity generation in the production plant. The costs used here were calculated with an update of the European tool for electricity plants (Table 11). The methodology is therefore consistent with the details presented in paragraph 2.2.

Table 11

	Euro/MWh	Ct/kWh
Fossil Mix	26.7	2.67
STEG (natural gas)	7.8	0.78
Nuclear	1	0.1
Belgian Mix	16	1.6

For most purposes (figures) we have used the fossil mix value. Clearly this should be regarded as an upper value. Nevertheless electric vehicles often perform very well compared to others even when this upper value is used.

2.3.3.3.f. Inland shipping

Introduction and methodological issues

To complement the calculations of marginal externalities (see paragraph 2.3.4.2.b) we also present estimates of Life cycle costs. Given the general lack of data on this subject and the importance that some policy makers attach to inland shipping, a lot of time was devoted to this part of this project. The scope of this study is schematically given in Figure 22.

Very often, no country-specific values are available. Vito therefore proposes European wide average values are which are based on a joint study of literature with IVM (The Netherlands). Even though the reference area for the analysis may be different from the country for which the figures are applied, the use of these values give an order of magnitude estimation which can be improved in later projects e.g. on modal shift.

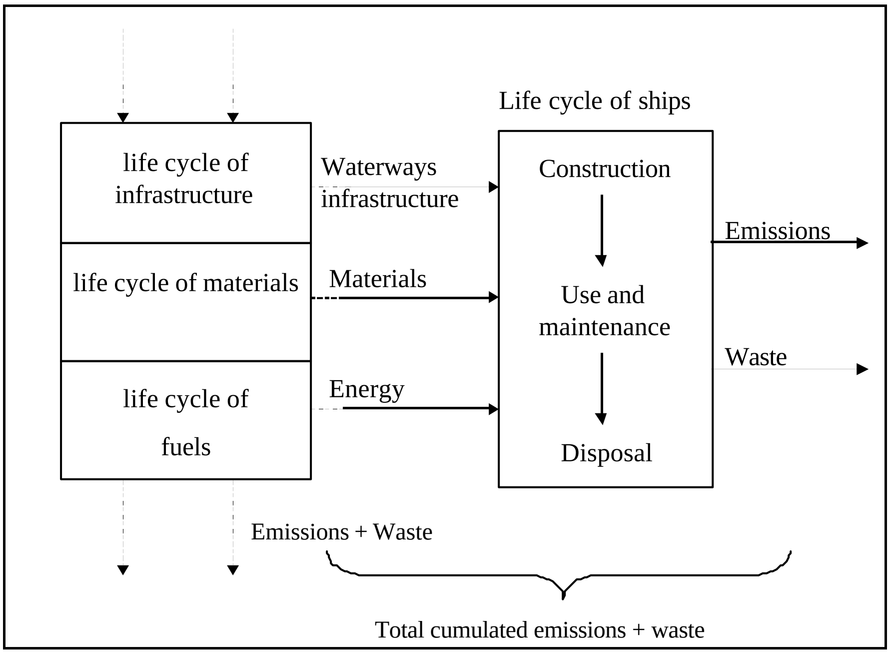


Figure 22: Life cycle inventory of ships for inland waterways transport.

Emissions and energy use for the life cycle stages: construction, maintenance, disposal, infrastructure, fuel production and operation of inland waterways ships are inventoried. Of each stage of the life cycle the direct and indirect emissions and energy uses are given. Indirect emissions due to production of materials are limited to process emissions due to material production and energetic emissions due to the use of fuels and electricity in the production process. Infrastructure, energy precombustion and materials used in the process are not considered. Infrastructure used by inland waterways transport is also taken into account. In this too, the direct emissions due to the construction of canals, locks, etc. is given, next to indirect emissions (defined in the same way as above).The functional unit is 1 tonne-kilometre. A lot of material was collected in tables and figures for this part of the project. To keep this paragraph readable, most of the basic information as well as specific references in literature dedicated are given in an annex to this report (see Appendix 1).

Overview of Inland shipping in Europe

Europe has approximately 30 000 km of navigable waterways and a total of about 110 000 million tonne-kilometres (tkm) of goods traffic was recorded. Germany, the Netherlands, France and Belgium are the most important countries with respect to inland waterways transport. The total amount of goods transport by inland waterways transport was about 500 million tonnes in 1995. Compared to the estimated 1 160 000 million tonne-kilometres of goods transport by road in 1996, inland waterways transport has an important potential as a substitute for road transport.

It is astonishing to see that there are a lot of ships older than 40 years still in operation. The average capacity of the 12 300 ships in operation in the EU is about 750 tonnes (there are no current EU data for the nineties available). The newer ships have a larger capacity though, up to 2 000 tonnes per ship built in the eighties. It must be noted that this is important for the emissions from ships in the use phase: older technologies, having smaller capacities will have significantly higher fuel consumption per tkm than newer and larger ships. In Belgium the average age of the fleet was 41 year in 1996. 17.6 % of the load capacity is coming from ships that have originally been constructed before 1940 (0.3 % even before 1900!).

So on average a ship transports about 9 million tkm per year. Having an average capacity of 750 t, this comes down to an average yearly transport distance of about 12 000 km, at 100% load. At a more realistic average load of 85% (based on Belgian data) this comes down to a yearly transport distance of 14 000 km.
Different types of ships are used in different parts of Europe (Figure 23). In France the "Spits", with a capacity up to 350 tonne, is suited to navigate through the small waterways and locks in the north. In Belgium and South Holland a type called the "Kempenaar" (600 to 700 tonne capacity) is used for transportation of goods. "Dortmunders" (900 t capacity) originally were used on the north German canals. They now also sail in the north of France, Belgium and Switzerland. Rhein-Herne ships with a capacity of 1350 tonne were conceived for the Rhein-Herne canal in Germany. A lot of these ships have been rebuild and no longer have standard dimensions. "Euroships" (containerships up to 1500 t capacity) are now being used on most large navigable waterways throughout Europe. Rhine ships (3600 t) travel along the Rhine from Rotterdam to Basel. They too are very well suited for container transport. Push tugs have been navigating the Rhine since the 1960s. They have an enormous cargo capacity for transporting raw materials from the Dutch sea ports to the industrial areas in the German Ruhr area: Four-barge convoy set: capacity of 9600 tonnes, 4 x (76.50 x 11.40 m) or a six-barge convoy set: capacity of 14400 tonnes, 6 x (76.50 x 11.40 m)

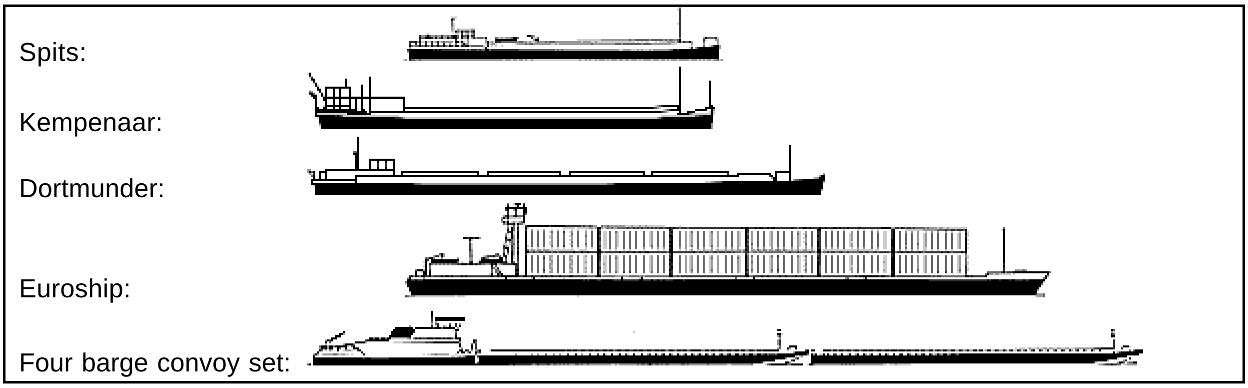


Figure 23: Common types of ships for inland waterways transport

Life cycle inventory : Up and downstream processes

In Ökoinventar Transport the emissions due to inland freight vessels are given based on the German situation for container transport, and based on ETH for bulk transport. ETH discusses tanker transport as well.

Bulk transport
Inland Waterways Transport data for bulk transport in ETH is based on (Western-) German data. The average load of an 1100 t capacity ship is around 70%, i.e. a load of 790 t; the average yearly distance is 12000 km. The lifetime is estimated between 30 and 35 years. Emissions due to the construction of ships out of materials can be estimated based on the figures in Table 12. For the construction of ships it is estimated that about 50% of the energy needed to produce the materials is used. 536 t of the 650-t weight of the ship is accounted for in Table 12. These assumptions fit quite well with the average ship in operation in Europe.

Table 12: Materials used for the construction of ships for bulk transport.

	material use (t)
Steel	445
High alloy steel	11
Cast iron	52
Copper	7
Cement	12
Mineral wool	9

Direct emissions due to the assembly of these materials into one ship are given in Table 13. Conversion factors for electricity and oil used in ETH are given in Appendix 1. ETH mentions the use of lubricating oil and paint for the maintenance of ships. Roughly estimated the use of 400 kg of synthetic resin paint equals the emission of 25 mg NMVOC/tkm for a small (650-t dead weight) inland waterways ship. Disposed materials are considered being re- or downcycled. The emissions due to disposal will be small compared to other stages. Infrastructure for inland shipping consists mainly of canals and locks. All these impacts are summarised in Table 13. The emissions and energy use during the use or operational phase are also given for comparison.

Table 13: Direct material use, energy use and emissions for inland waterways transport (bulk transport).

			Construction per tkm	Maintenance per tkm	Disposal per tkm	Infrastructure per tkm	Use per tkm
Materials							
	Steel	kg	0.0014			7.20E-05	
	High alloy steel	kg	3.50E-05				
	Cast iron	kg	1.65E-04				
	Copper	kg	2.20E-05				
	Cement	kg	3.80E-05			1.30E-04	
	Mineral wool	kg	2.80E-05				
	Paint	kg		5.00E-05			
	Bitumen	t				2.50E-07	
	Concrete (sand and grind)	kg				5.00E-03	
Energy							
	Electricity UCPTE (mid)	kJe	4				
	Heavy Fuel in Burner 1MW	kJ	40				
	Diesel	t				2.48E-06	1.10E-05
Emissions							
	CO2	kg	0.0036			7.52E-03	0.0353
	CH4	kg	1.29E-07			4.60E-07	2.79E-05
	N2O	kg	6.79E-08			2.90E-07	
	CO	kg	6.00E-07			3.84E-05	0.000158
	SOx	kg	5.10E-05			6.28E-06	2.88E-05
	NOx	kg	7.13E-06			1.04E-04	0.000279
	NMVOC	kg		2.50E-05		1.68E-05	6.04E-05
	particles	kg	2.12E-06			1.54E-05	3.00E-05
Waste							
	Bilge oil	kg					6.00E-05

The production of materials, used to construct ships and infrastructure can also be estimated (Appendix 1). The resulting emissions in Table 14 can be compared to the direct emissions in Table 13.

Table 14: Indirect emissions due to production of materials.

			Construction, maintenance and disposal per tkm	Infrastructure per tkm
Energy				
	Fossil fuels	TJ	4.31E-09	8.31E-10
	Electricity	TJe	1.93E-09	1.73E-10
Emissions				
	CO2	kg	6.18E-04	1.71E-04
	CH4	kg	1.35E-08	1.43E-09
	N2O	kg	5.97E-09	8.10E-10
	CO	kg	2.16E-06	2.07E-07
	SOx	kg	7.24E-06	2.78E-07
	NOx	kg	8.57E-07	4.47E-07
	NMVOC	kg	4.94E-08	3.34E-08
	particles	kg	2.14E-07	5.13E-08

When comparing the energy use, either between the direct and indirect inputs, or between up- and downstream processes and the operational phase, it can be seen in Figure 24 that the use phase is more important by almost a factor 10 to the construction, maintenance and disposal stages and by a factor of five with infrastructure. Indirect electricity use for construction is comparable to direct electricity use. The same conclusions apply to emissions (Figure 25, Figure 26). The infrastructure used by inland waterways transport cannot be neglected.

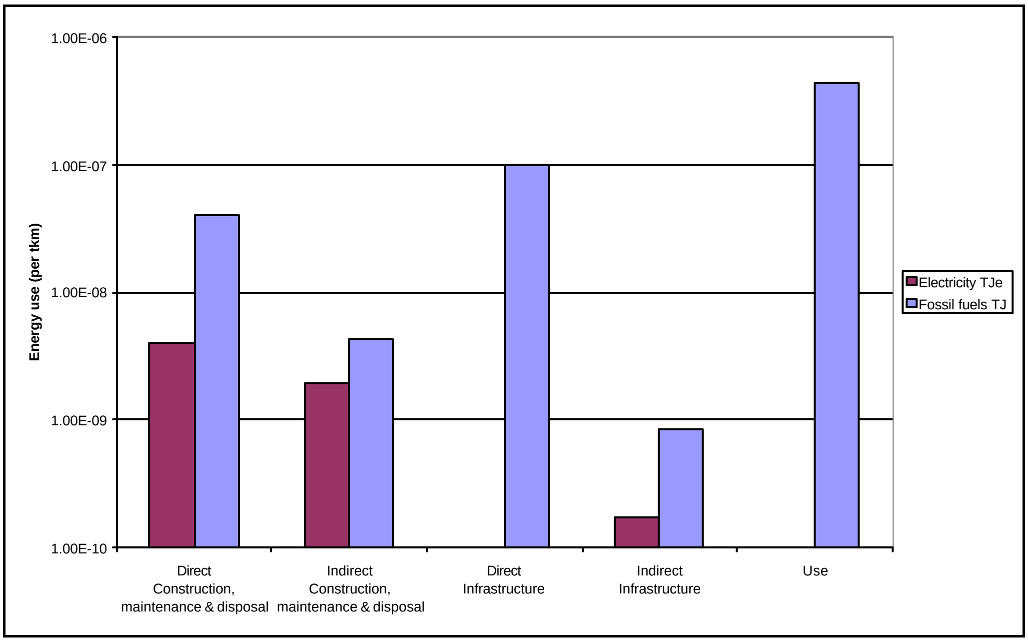


Figure 24: Energy use for use of ships compared to other life cycle stages and infrastructure

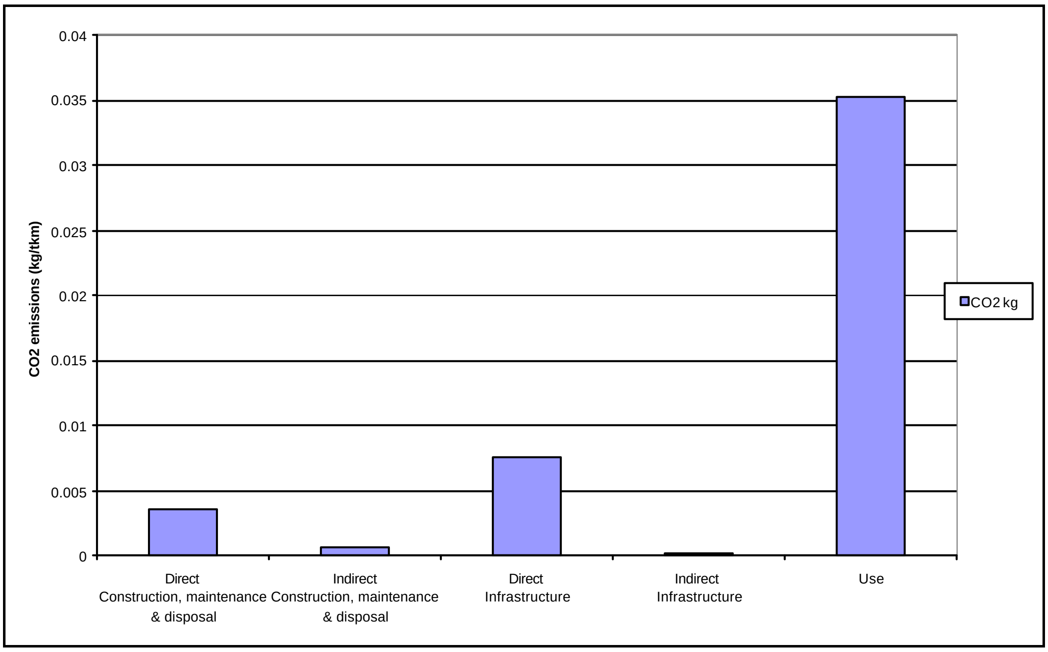


Figure 25: CO2 emissions due to use, construction and maintenance and infrastructure

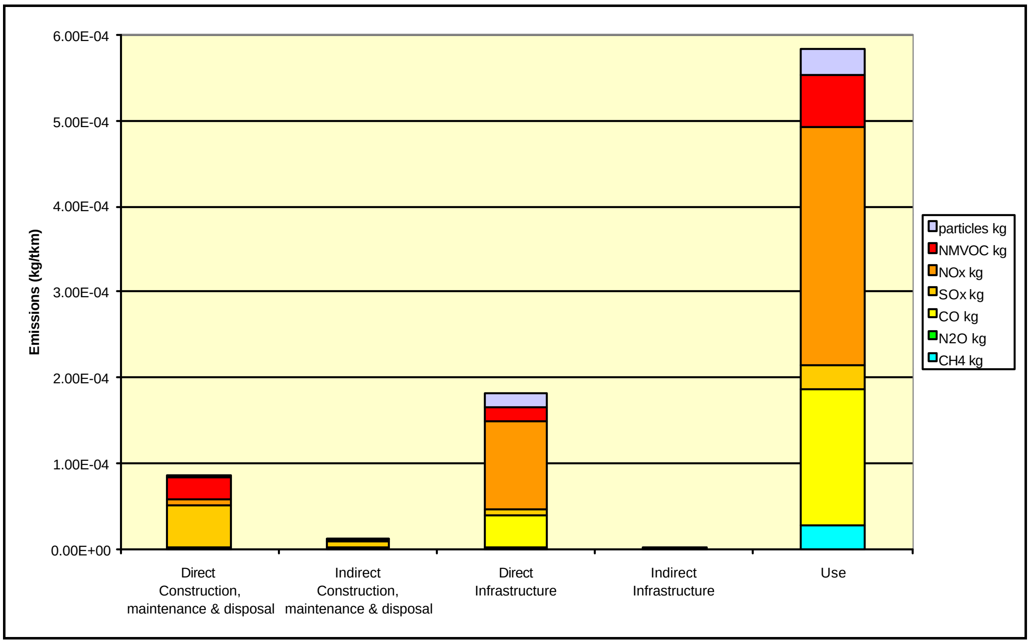


Figure 26: Other emissions due to use, construction and maintenance and infrastructure

Inland tanker

Inland tankers are calculated separately in ETH, based on Swiss and German data for Rhine transport. The average capacity of tankers is 1900 t, the load about 65% (1200 t). The material list for tankers is very limited: about 500 t of steel and 5 t of copper. The same assumptions for maintenance and infrastructure are made. The results are summarised in Table 15.

Table 15: Results for inland tankers.

			Construction, maintenance and disposal		Infrastructure		Use
			Direct (per tkm)	Indirect (per tkm)	Direct (per tkm)	Indirect (per tkm)	(per tkm)
Energy							
	Fossil fuels	TJ	2.25E-08	7.60E-10	9.91E-08	8.31E-10	4.80E-07
	Electricity	TJe	1.00E-09	2.19E-09		1.73E-10	
Emissions							
	CO2	kg	1.87E-03	3.17E-04	7.52E-03	1.71E-04	0.038
	CH4	kg	6.98E-08	6.05E-09	4.60E-07	1.43E-09	3.00E-05
	N2O	kg	3.70E-08	2.40E-09	2.90E-07	8.10E-10	
	CO	kg	3.38E-07	9.04E-09	3.84E-05	2.07E-07	1.70E-04
	SOx	kg	2.78E-05	4.07E-06	6.28E-06	2.78E-07	3.10E-05
	NOx	kg	3.78E-06	4.31E-07	1.04E-04	4.47E-07	3.00E-04
	NMVOC	kg	2.50E-05		1.68E-05	3.34E-08	4.00E-05
	particles	kg	1.16E-06	7.77E-08	1.54E-05	5.13E-08	3.00E-05

Container transport
In Ökoinventare Transporte data for construction, maintenance and disposal of containerships are given, as well as estimates for the infrastructure. It is not clear whether the system boundaries are the comparable for construction of ships and infrastructure. The cumulated emissions (direct + indirect) for these life cycle stages are significantly higher than for the assessment of bulk and tanker transport above. Allocation is made for harbours that are shared by sea transport ships, inland waterways transport ships and other means of goods transportation. It is noted that the emissions due to harbour specific industry are included in the overall infrastructure emissions. Natural rivers and the maintenance thereof are excluded. In this study we chose to use the material list and other direct inputs from Ökoinventare Transporte, but apply the same methodology as we did for bulk and tanker transport. The resulting direct and indirect emissions are then consistent with the other types of ships.
The average capacity of a container ship is 2500 t. The average load is 45% or 1139 t, while the average yearly distance is 26667 km (400000 km in a lifetime, or 15 years in total is accounted for). The material list for this average container ship of 855 t dead weight is more detailed than in ETH (Table 16). Here also it is estimated that about 50% of the energy needed to produce the materials is used for the construction of ships. NMVOC emissions due to painting during maintenance are about 14.5 mg/tkm. Disposal of non-metal components of ships is of minor importance.

Other burdens

PIANC (the Permanent International Association of Navigation Congresses) is currently working on the environmental aspects of inland waterways (PIANC working group 14). A draft document of this working group concludes that the external costs due to noise, accidents, surface occupation and barrier effects are close to zero.. Another project, Immunity, funded by the EC, is aimed at the identification, development and selection of concepts, methods and tools to reduce the potential negative impacts due an expected increase in the volume and multipurpose use of inland navigation. Results aren't published yet.

Table 16: Materials and energy used for the construction of container ships.

	material use
Steel	738 t
High alloy steel	9.6 t
Cast iron	48.4 t
Copper	6.11 t
Cement	10.5 t
Mineral wool	7.86 t
Aluminium	0.436 t
Plastics (PP)	2.18 t
Rubber EPDM	1.31 t
Ceramics	3.06 t
Paint	4.364 t
PE	4.36 t
Wood	1.75 t
Electricity UCPTE for construction	1.05 TJe
Fuel (heavy oil) for construction	9.45 TJ

Table 17: Results for container ships.

			Construction, maintenance and disposal		**Infrastructure**		**Use**
			Direct (per tkm)	**Indirect (per tkm)**	**Direct (per tkm)**	**Indirect (per tkm)**	**(per tkm)**
Energy							
	Fossil fuels	TJ	2.07E-08	2.84E-09	9.91E-08	8.31E-10	4.40E-07
	Electricity	TJe	2.30E-09	1.98E-09		1.73E-10	
Emissions							
	CO2	kg	1.89E-03	4.90E-04	7.52E-03	1.71E-04	0.035119
	CH4	kg	6.75E-08	1.07E-08	4.60E-07	1.43E-09	1.65E-06
	N2O	kg	3.54E-08	4.99E-09	2.90E-07	8.10E-10	
	CO	kg	3.11E-07	1.33E-06	3.84E-05	2.07E-07	2.20E-04
	SOx	kg	2.66E-05	5.16E-06	6.28E-06	2.78E-07	2.81E-05
	NOx	kg	3.74E-06	8.73E-07	1.04E-04	4.47E-07	3.86E-04
	NMVOC	kg	1.45E-05	2.12E-07	1.68E-05	3.34E-08	5.35E-05
	particles	kg	1.11E-06	1.74E-07	1.54E-05	5.13E-08	3.86E-05

Additional burdens are mentioned in a range of publications (see Appendix 1). Due to the use of antifouling paints containing copper during maintenance about 22 ton copper per year is released to water in the Netherlands.

Spilling due to loading/unloading of ships causes small, uncontrollable emissions to air and water. The absence of a vapour return causes the release of VOC and other gases transported by tankers. Cleaning of ships is another possible source, especially of emissions of bilge oil, grease and heavy metals to water. No numbers are available however. In the Netherlands an estimated 790 to 1580 tonnes of lubricant grease is lost due to leakage every year.

An overview of waste products from ships is given in
Table 18. They all present potential burdens to water or air.

Table 18: Overview of other potential burdens to water and air.

Type of waste generated during maintenance or use of the ship	Average or median(*) quantity per ship and per year
Used lubricant oil or insulating oil	0.3 m³
Fuel residue	0.06 m³
Grease	0.06 m³
Bilge oil	0.76 m³
Bilge water	8 m³
Bilge oil/water	8 m³
Hazardous wastes	0.02 t
Household waste	2.3 t
Solid chemical residues	0.2 t
Other residues	0.5 t
Liquid chemical residues	17 m³
Liquid diluted oil residues	14 m³
Wash water containing chemicals	30 m³
Wash water containing oil	240 m³
Ballast water containing oil	290 m³
Spray painting, resin application generating VOC emissions	na.
Electroplating/metal finishing generating metal sludges	na

na: not available

Operational phase

This aspect of the life cycle of inland waterways transport has been studied before in ExternE Transport. A summary of fuel consumption and emissions per unit of fuel is given in Table 19. Precombustion emissions are not included.

Table 19: Emissions from ships for inland waterways transport.

	CO2 (g/kg)	**SO2 (g/kg)**	**NOx (g/kg)**	**Particles (g/kg)**	**CO (g/kg)**	**HC (g/kg)**	**VOC (g/kg)**	**fuel consumption (kg/tkm)**
Inland shipping Germany (ABL-1988)	3175	3	60	2 (2)	12	5		0.0112
Inland shipping Germany (NBL-1988)	3175	12	60	4 (2)	24	10		0.0112
Inland shipping Germany (BVWP'92-2010)	3153	2	27	0.8 (2)	5	3.4		0.0112
Inland shipping Germany (DIW/IFEU-2010)	3175	0.9	60	1.2 (2)	9.6	4		0.0112
MEET "Inland navigation"		8	76	13 (3)	31		9.1	21.27 (6)
MEET "diesel motor ships - river"		19*S (1)	39		14		7	21.27 (6)
ETH "Inland bulk transport"	3209.1	2.62	25.36	2.73 (2)	14.36		5.49	0.011
ETH "Inland tanker"	3166.7	2.58	25.00	2.50 (2)	14.17		3.33	0.012
ExternE 1997 "small ship"	3130	4.00	36.60	0.30 (4)	6.60		4.00	0.01
ExternE 1997 "medium ship"	3130	7.10	57.00	0.30 (4)	9.50		5.50	0.006
ExternE 1997 "push vessel"	3130	7.10	57.00	0.30 (4)	10.10		5.70	0.008

(1): Sulphur content of fuel
(2): Soot
(3): PM
(4): PM10
(5): only NMVOC
(6): in t/day; average consumption function: C= 9.8197+0.00143*GT (GT= gross tonnage)

Discussion and summary

For ships up to about 1500 t capacity (types like the Spits, Kempenaar and Dortmunder) it seems reasonable to use the ETH material list, by scaling it to the right weight of the ship. Given the lifetime, the average load percentage, and the yearly transport in tkm, the specific material and energy list can be calculated, and consequently the accumulated emissions due to construction and maintenance. Disposal data are not available, but will be small.
Infrastructure data have to be used with great caution.
For container ships the Ökoinventar material data are suitable. Also here the material and energy list for construction can be scaled according to the weight of the ship.

The emissions for different types of ships are summarised in Appendix 1. Based on these emissions we have calculated the LCA-costs that are discussed in paragraph 2.3.4.2.c Comparison of traditional and alternative transport modes.

2.3.3.4. Other impacts

2.3.3.4.a. Brake and tyre wear

These emissions were not included in the ExternE project and were not included in this project either. Although models exist that provide emissions for brake and tyre wear (e.g. the recent Motor Vehicle Emission Inventory 7G of the California Environmental Protection Agency Air Resources Board), it is clear that these particles are of a different nature than primary and secondary particles resulting from the combustion of fuels. It would therefore be questionable to assign them the same types of impacts.

Brakes of cars can contain between 10 and 70 % asbestos, especially of the “chrystotiele” type. The amounts of asbestos that are released from the brakes of vehicles have been shown to be too low to pose a cancer threat to the population, Although they do contribute to overall pollution of asbestos. In addition most brakes contain the “chrysotiel” form of asbestos, the least dangerous form (source: prof. A. Bernard en W. Hecq).
Estimates available in the US (MVEI7G) are also based on data from asbestos brake pads.
A number of 0.0078 g PM10/km is used for all model years and all vehicles (cars, motorcycles, trucks, and buses). However, since asbestos brake pads were phased out in the late 1980's, actual US emission factors may differ from those found in the model.

On road and tyre wear emissions MVEI concluded that there are inconsistencies and substantial differences in the tyre wear EFs reported. But there is little doubt that substantial portions of PMTW exceed 10 microns. The fraction below 10µm is estimated at 0.005 gPM10/km for all types of passenger cars. Motorcycles are attributed half this value, for trucks and buses this is multiplied by the number of tyres or axles.

Vito performed a limited number of test runs with the Californian MEI7G emission model. For modern American petrol cars many emission factors are similar to those predicted by MEET and INFRAS in Europe at similar speeds. Emission factors of PM10 tail pipe exhaust were broadly similar to those from INFRAS but up to 2 times lower than those of MEET and Klein, especially at urban and highway speeds.

PM10 from brake and tyre wear are relatively small for diesel cars (5-13% of total PM10 emissions) and much smaller than the differences between INFRAS and MEET for PM exhaust emissions.

In petrol cars PM10 from brake and tyre wear range from 33-46% of total PM10 for non-catalyst cars and between 72 and 83% for modern cars. This makes them potentially important. However even PM10 exhaust emissions from petrol cars are poorly known. For Europe we have only fragmentary data from Finland and the Netherlands (Klein). The American estimate in MVEI7G has not been changed since 1980.

Given the large numbers of uncontrolled vehicles and diesel-fuelled vehicles in the Belgian fleet, we feel that PM10 from brake and tyre wear is less important than tail pipe emissions. In addition, PM emissions from brake and tyre wear from the MEI7G model are not speed dependent and identical for diesel and petrol cars. They are therefore not of use in discriminating between car types or locations, one of the main objectives of this study.

2.3.3.4.b. Noise

Other studies (e.g. the recent MIRA-S report) have made it sufficiently clear that noise is an important issue. Transport is the single most important, and most widespread source of noise nuisance. While noise from road traffic has increased with traffic volume, it is not clear if emissions from individual vehicles have decreased as a result of more stringent standards (de Graaff, 2000). Therefore the number of people exposed to high levels of traffic noise remains quite high (over 20% in many scenarios).

Recently we have added a module for the calculation of traffic noise impacts to the ExTC-tool. External cost estimates from different sources (TRENEN) can be presented together with air pollution externalities. A separate module allows a more detailed look at the effects of speed (individual and traffic) and traffic density. However the use of a more detailed model also necessitates a number of difficult assumptions that strongly influence the results. For this reason Delucchi (1998) states that it is impossible to calculate noise externalities with an accuracy better than 2 or 3 orders of magnitude.
Therefore our results can not be presented yet. Wherever we have presented estimates of noise related externalities in this report, we have ensured that they are from a single source and based on identical assumptions so that they may be compared in the context they are presented in.

While it is possible to estimate the local change of average sound levels due to traffic, literature reviews show that it is impossible to quantify the health impacts. In a recently published report the British DETR was advised that:

"it is not possible at this time to establish health effect based assessment methods."

(DETR, 1998) among the reasons given for this problem are :

- Although there is evidence to support the existence of a cause-effect relationship for annoyance, and some evidence for activity interferences such as sleep disturbance and speech interference, we do not have convincing evidence whether other measurable effects exist at all at typical levels of exposure to community noise. In addition we do not know what proportion of the population might be affected and to what extent.

- Although there have been numerous scientific studies on the health effects of noise, we cannot at present define robust exposure-response relationships for all the potential effects. Existing dose-response relationships are confounded by a number of variables which serve to scatter the data points around these cause-effect curves. These include non-acoustical exposure variables which can have major effects on attitudes and opinions.
- The scientific evidence suggests threshold levels below which no effects are expected. Again, since these are based on fragmentary and unconvincing evidence, the levels suggested cannot be taken as definitive at this time.
- There are uncertainties in assessing the overall impact of noise on health. These relate to the treatment of more than one effect, the role of modifiers, the cumulative exposure of different time periods, the handling of vulnerable or susceptible groups, and the role of other risk factors in assessing conditions of multi-factorial origin. These need to be better understood before effect-based assessment methods can be established.

Nevertheless, there are ongoing attempts to quantify health impacts from noise more accurately in the near future (e.g. TNO in collaboration with IER in a Unite project called Unite which is monitored by Vito).
A recent report to the CER/UIC (2000) quantifies the external costs of noise based both on health impacts and WTP. Results from this study are shown in paragraph 2.3.4.2 because these estimates are the most up to date for rail transport and strictly comparable figures for road transport are also given.

2.3.3.4.c. Evaporative emissions

Evaporative emissions occur in significant quantities for gasoline vehicles in the form of NMVOC emissions. Evaporative losses contribute substantially to total road transport related VOC emissions. In Belgium evaporative losses account for about 37% of total VOC emissions from road transport in 1995 (ref. MIRA). Since the method of calculation differs from combustion emissions and the impacts are hard to attribute to specific traffic situations or locations, they have been omitted in this study.
It should be noted that also refuelling losses exist. These are not included in this paragraph as they are emitted at petrol stations and therefore integrated in the LCA of the fuel cycle. Estimates were based on the emissions quoted in Meulepas et al. (1999)

2.3.3.4.d. Emissions to soil and water

ExternE has traditionally kept a strong focus on airborne pollutants from combustion and has been criticised for this. A new European research project (NewExt) will now look at other pathways for exposure to pollutants, including soil and water. Therefore impacts from these pathways are not included in the present analysis. Nevertheless the LCA module of ExTC model has been extended to include abatement costs for soil remediation in the vicinity of filling stations. The calculations are based on the levy of 0,13 BEF/l petrol and 0,08 BEF/l diesel that the three Belgian communities plan to use to pay for the remediation of historic pollution. Although the total accumulated cost of this pollution is quite large (up to 18 billion BEF) and marginal costs (per km) are of the same order of magnitude as the health related impacts of the use-phase in modern cars. Further elaboration of the impact-pathway approach is therefore warranted.

2.3.4. Alternative means of transportation

2.3.4.1. Alternative technologies and fuels (2005) for road transport

2.3.4.1.a. Introduction

In addition to the numerous numerical results that were obtained for the vehicles and fuels that are currently dominant in Belgium, we have also studied some alternative vehicles and fuels. Since most of these technologies are not in extensive use it is difficult to obtain reliable emission factors. Nevertheless it should be possible to identify the most promising technologies (in terms of air pollution).

To maintain a clear perspective it was decided not to study complicated experimental systems or to go into technical details. In order to provide useful information to policy makers, we have concentrated on existing or emerging (in the market) technologies. It is therefore possible to promote the use of these vehicles or fuels by 2005.

2.3.4.1.b. Gaseous fuels: LPG and CNG

Emissions

The only alternative fuel which is already extensively used for passenger cars in Belgium is LPG, a mixture of propane and butane. The emission functions selected are those from MEET for the classic pollutants. The emission factors of LPG were already discussed in Chapter 5. As in ExternE, additional emission factors were added for particles and carcinogenic compounds (that were missing in MEET). The 1993 EC standard set the sulphur content of LPG at a maximum of 200 ppm. However specialists at Vito claim that actual sulphur contents are much lower and SO2 emissions are effectively zero.
As was mentioned in Chapter 5, modern LPG cars have lower externalities than petrol cars, but old LPG cars are no match for modern petrol cars and may even be worse than modern diesels in rural areas.

CNG is compressed natural gas and consists mostly of methane (CH4). Based on stoichiometric calculations, complete natural gas combustion should result in 19% lower engine-out (i.e. untreated exhaust) carbon dioxide emissions compared with petrol. Carbon monoxide is a product of incomplete combustion and is formed far more readily from liquid fuels with comparatively large molecules. It is estimated that reductions in carbon monoxide of up to 90% could be attained with natural gas relative to petrol (Whiteman, 1994).
The actual level of unburned volatile organic compounds (VOC) from engines is very difficult to predict due to the large number of physical processes that cause incomplete combustion. Very little fundamental research has been carried out with dedicated gas engine systems, thus reliable data are hard to come by. However, it is expected that gaseous hydrocarbons would oxidise in the catalyst far better than liquid droplets. There is a potential reduction of 15 to 40% in engine-out VOCs relative to petrol engines (Whiteman, 1994). It should be noted that the VOC content of exhaust from an NGV will be predominantly methane.
Oxides of nitrogen (NOx) are formed when the normally inert nitrogen molecule is dissociated and oxidised at the high temperatures found in engine combustion chambers. With optimised natural gas engines the higher compression ratios increase combustion temperature

and tend to increase NOx production. This effect is compounded by the absence of the cooling effect of fuel droplet evaporation. On the other hand, natural gas engines give rise to lower flame speeds and more controlled turbulence which reduce NOx production.
MEET provides emission function for the classical pollutants for CNG cars. In urban areas, NOx emissions are about three times lower than those of an LPG car equipped with a catalytic converter. VOC emissions are almost 10 times lower and with most of this fraction (90%) being methane, NMVOC emissions should be a 100 times lower. However the emission functions in MEET for alternative vehicles are very unreliable and specialists at Vito's vehicle technology research group feel that it is better to assume the same emissions as for LPG for CO, NOx and VOC. Data from Rijkeboer (1994) and Nymund (1996) seems to partially support this view.
No reliable information could be found on the emissions of particles and carcinogenic compounds. Rijkeboer et al. (1994) measured PM emissions from CNG cars that were in the same range as petrol cars and almost always higher than for LPG. Stephenson (1997) blamed this on "minor deficiencies" that caused the burning of lubricating oil. Some specialists feel that emissions of PM and carcinogens, if they exists, must be at least 50% lower than for LPG. Since it was shown in paragraph 2.2 that this pollutant is very important for the outcome of the external costs estimate, calculations for CNG are highly uncertain and will remain uncertain until better measurements of PM become available.
The fuel consumption of CNG vehicles (expressed as kJ) is approximately the same as in petrol cars (or slightly higher because of the extra weight of the fuel tanks). In practice the energy content of 1 Nm^3 natural gas is equal to 1 litre of petrol. Disregarding the specific properties of the gas (in view of the other uncertainties) we assume that the fuel consumption of CNG cars in (Nm^3/100km) is equal to that of petrol cars (in litre/100 km). When compared to diesel vehicles, their energy use is 20 to 30% higher, but the CO2 emissions are the same (LP, personal comm.).
Although most H2S is removed from natural gas before is it is distributed, some traces remain in the burned fuel resulting in SO2 emissions during the use-phase. The sulphur content of NG-fuel was assumed to be 0,002% or 20 ppm. This is very low, but not negligible when compared to very clean petrol and diesel that will be used from 2005 onwards.
The emission factors that were used for CNG cars are shown in the table below.

Table 20 Emission factors for CNG fuelled passenger cars

Urban drive	Vito (TEMAT)	MEET (extended)	TNO
CO	1.8102	1.575101288	0.71
NMHC	0.2767	0.00308353	0.061
NOx	0.38276	0.130267971	0.14
PM	0.003	0.001	0.012
SO2	0.002118802	0.002935482	0.002935482
NH3	0.075483669	0.02569	0.02569
CO2	168.7072	201.8143734	201.8143734
N2O	0.053916906	0.016969744	0.016969744
CH4	0.08	0.027751772	0.549
BENZENE	0.0011	4.77947E-06	0.0009
1,3 butadiene	0.0013835	6.01288E-06	0.0005

2.3.4.1.c. Results for LPG and CNG

Table 21 shows the external costs in Eurocent/km for the gaseous fuel as compared to recent petrol and diesel cars. Because of widely different emission factors found in literature we have chosen to present three estimates for CNG cars based on:

1. Emissions based on Rijkeboer (1994) and Nylund (1996)
2. The Vito estimate (MEET emissions for LPG scaled down for PM, benzene and butadiene
3. MEET emissions functions complemented with Vito estimates for PM, benzene and butadiene

Table 21: Results for LPG and CNG fuelled cars compared to current technology

	Diesel EURO3	Petrol EURO3	LPG controlled	CNG TNO	CNG MEET+	CNG Vito
Urban	3.43	0.57	0.44	1.19	0.16	0.25
rural	0.22	0.09	0.10	0.09	0.06	0.08
highway	0.59	0.14	0.13	0.11	0.08	0.10

One possible advantage of gaseous fuels that is not reflected in the results shown above, is the fact that gaseous are little affected by low temperatures. There are fewer problems with cold start emissions, enrichment requirements and evaporation.

2.3.4.1.d. Biodiesel

Emissions

Emission functions for diesel cars are essentially the same, whether they are fuelled with biodiesel or fossil diesel fuel. For CO, PM and VOC the same functions were used but they were scaled down (-20% for CO and VOC and –33% for PM) (source: TEMAT). Although more optimistic (for passenger cars) this is essentially comparable to other sources such as (ETSU, 1994). ETSU claim NOx emissions to be 15-20% higher relative to vehicles driving on fossil diesel.
In view of the dominance of PM in the externalities of diesels, it is clear that the air pollution of the use-phase of biodiesel cars will be lower. This effect will be most evident in urban areas. In addition CO_2 emissions are essentially zero because they are derived from photosynthetically assimilated carbon. While older publications state that SO_2 emissions are virtually zero, this should be compared to the fossil diesel fuels that were used a decade ago. When compared to recent and future (low sulphur) city diesel, the sulphur content is no longer negligible. Sams and Tieber (1995) estimate the sulphur content of RME at 0,002% or 20 ppm while one internet source (www.biodiesel.de/umwelt.html 19/10/1998) claims that the sulphur content is less 0,001% or 10ppm. We have used the 20ppm estimate in our calculations. This means that SO_2 emissions from biodiesel would effectively be less then half those of highly purified petrol and diesel and about the same as those of CNG.

Results for biodiesel

The external costs (use-phase) from diesel cars fuelled with biodiesel are lower compared to fossil diesel fuel (Table 22). In urban areas this is mainly the consequence of the lower emission of particles. (we implicitly assume that PM emissions from biodiesel have the same health impacts as from fossil diesel, although there is some discussion that these emissions may have different characteristics. In rural areas, the reduction of greenhouse gasses may be the dominant difference. Nevertheless, impacts from biodiesel remain high, especially in urban areas, when compared to other alternatives. If global warming impacts are important, biodiesel may be better than diesel cars equipped with PM-filters on rural and highway trajectories. Biodiesel is especially interesting if you include the global warming impacts from the fuel cycle, but the fuel cycle has important impacts from emissions to water and soil. De Nocker (2000) and Spirinckx (1999) cover this topic in detail.

Table 22: External costs of use-phase in Eurocent/vkm (classic pollutants / Global Warming high estimate)

	EURO1	Biodiesel	FAP
Urban	3.38 / 0.32	2.25 / 0.01	0.45 / 0.32
Rural	0.18 / 0.23	0.13 / 0.01	0.08 / 0.23
Highway	0.55 / 0.3	0.38 / 0.01	0.1 / 0.3

2.3.4.1.e. Alcohols: methanol and ethanol

Emissions

Other fuels that can be made from biomass are the alcohols methanol and ethanol. Although they have not been used in Belgium, a large amount of information on these fuels was collected in the Americas (esp. Brazil and the US). In spark ignition vehicles the alcohols are typically blended with 15% petrol (indicated as E85, for ethanol and M85 for methanol) which allows use of the fuel in a conventional petrol engine with little adjustment.

To estimate emission factors for these fuels, we have used the scaling factors shown in Table 23 (sources: ETSU, 1994 and ExternE, 2000) on the standard MEET function for controlled petrol cars (EURO 1). ETSU assume no difference in the PM emissions when compared to petrol fuelled cars (indicated with an * in the table. Similarly, we have found no information on the sulphur content of bioalcohols and have therefore assumed it to be equal to that of regular petrol as to allow a like-with-like comparison between each of the fuels. If alcohols are produced biologically, CO2 emissions should be ignored.

Table 23

Vehicle type		Fuel	Source	Scale factor					
				CO_2	CO	HC	NOx	PM	SO2
Car	TWC	E85	ETSU	1.00	0.40	1.00	0.35	*	*
			ExternE	0.94	1.43	1.3	1.03	*	*
Car	TWC	M85	ETSU	0.95	0.69	0.78	0.83	*	*
			ExternE	0.92	0.91	0.6	1.14	*	*

As can been seen from the numbers in the table, it is unclear for most pollutant whether they are higher or lower than in standard petrol fuelled cars. It was also the feeling of the European ExternE-team that emission factors for these fuel are very unreliable. Results in figure have been calculated with ExternE Scaling factors.

2.3.4.1.f. Results

Given that :
- there is no information about the emission of particles
- emissions of NOx compared to petrol fuelled cars are very uncertain
- there is no information on the use of biofuels in combination with sophisticated engine technologies.

it is unclear which benefit could result from substituting petrol with alcohols. Predicted externalities from the use phase can hardly been distinguished for regulated pollutants (based on ExternE scaling factors). The only benefits may lie in the photosynthetic origin of the CO2 emissions when bioalcohols are used (Figure 27, CO2-cost are hatched).

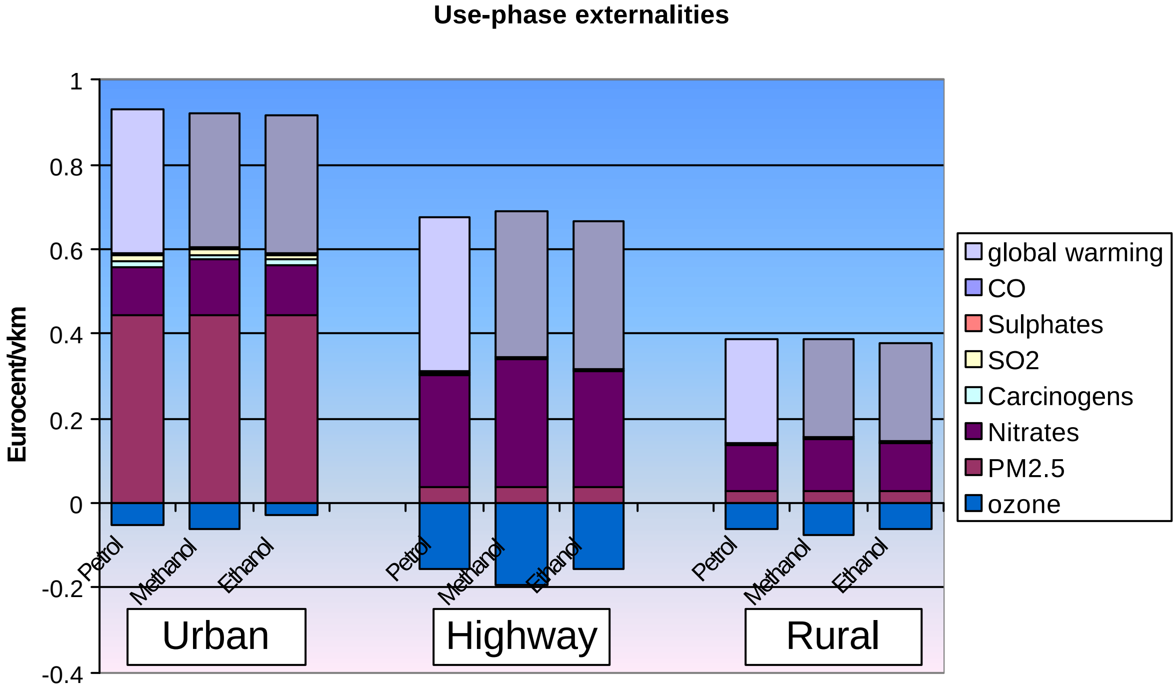


Figure 27: External costs of Euro1 petrol cars with fossil petrol and two alcoholic substitutes

2.3.4.1.g. Diesel with particulate trap

Emissions

From the overwhelming dominance of PM impacts in the externalities in diesel cars, it is clear that any measure aimed at reducing the PM emissions will also have an effect on the external costs. Because of the local nature of primary PM impacts, the effect will be much more important in cities then in rural areas.
Some producers (PSA) claim that particulate filters have PM emissions as low as 4 mg/km (more than 6 times lower than the EURO4 standard and similar to petrol cars). We have linearly applied this reduction to the PM-impacts, which implies that:

1. mass is the correct indicator to evaluate the effects of particles and
2. particulate filters and trap remove particle of all sizes with the same efficiency and
3. do not changes their properties (e.g. adsorbed PAKS)

Before advocating the widespread use of particulate filters, further scientific study should confirm these implicit assumptions. Some people have questioned whether filters are capable to eliminate ultra-fine particles (PM 0.1). It is also uncertain if these particles may be assigned the same exposure-response functions as PM2.5.

Results

Results in Figure 28 are based on EURO3 emission factors from the MEET database except for PM2.5 which was taken to be 0,004 g/km. Although reduction of externalities (especially health effects) is impressive, it does not make diesels the best environmental choice, it merely brings them in line with modern petrol cars. The PM emission factors boasted by producers should therefore be carefully checked in future measurement campaigns.

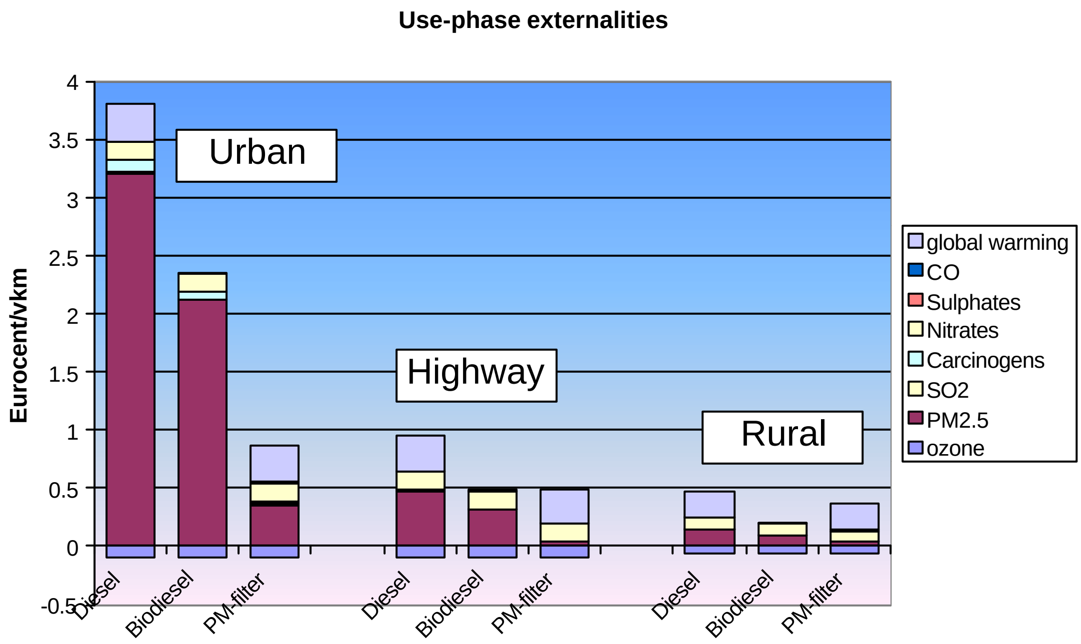


Figure 28: External costs of three types of diesel fuelled passenger cars

2.3.4.1.h. Hybrid vehicles

Emissions

Hybrid vehicles are cars that have both a classic combustion engine and an electric motor. At low speeds propulsion can be entirely electric (without emissions) while both engines work together at higher speeds. The combustion engine operates in a less dynamic mode and supplies electricity to the electromotor. This principle is relatively simple and has been used for a long time e.g. in diesel-electric locomotives. In some models the combustion engine can also directly divert power to the transmission at high speeds, creating a truly hybrid powertrain which is more complicated.

The most important improvement (in terms of emissions and fuel consumption) comes from the nearly constant power regime under which the engine is working. The regime is unrelated to the dynamic demand for power resulting from rapidly changing traffic conditions in real life driving. Hybrid vehicles are already commercially available in Japan and California. Toyota has also started selling the Prius from its Belgian website this year.

Emissions for hybrid vehicles were taken from TEMAT (CO, NOx, VOC, and CO2). Since these are very low indeed, selecting emission factors for PM becomes crucial to the outcome of externality estimates. Because hybrid vehicles for road transport are a very recent development we could not find independent measurements. We have therefore assumed the PM emissions under all circumstances to be equal to those of a regular petrol car in a rural driving mode.

Results

The use-phase results for hybrid cars are shown in Figure 29. Their impacts are considerably smaller than those of modern petrol cars, especially in highway driving. Their performance relative to normal petrol cars in urban driving depends heavily on our assumption on PM emissions. However, taking into account their ability for electric propulsion at low speeds, their performance in urban traffic may even be better than expected.

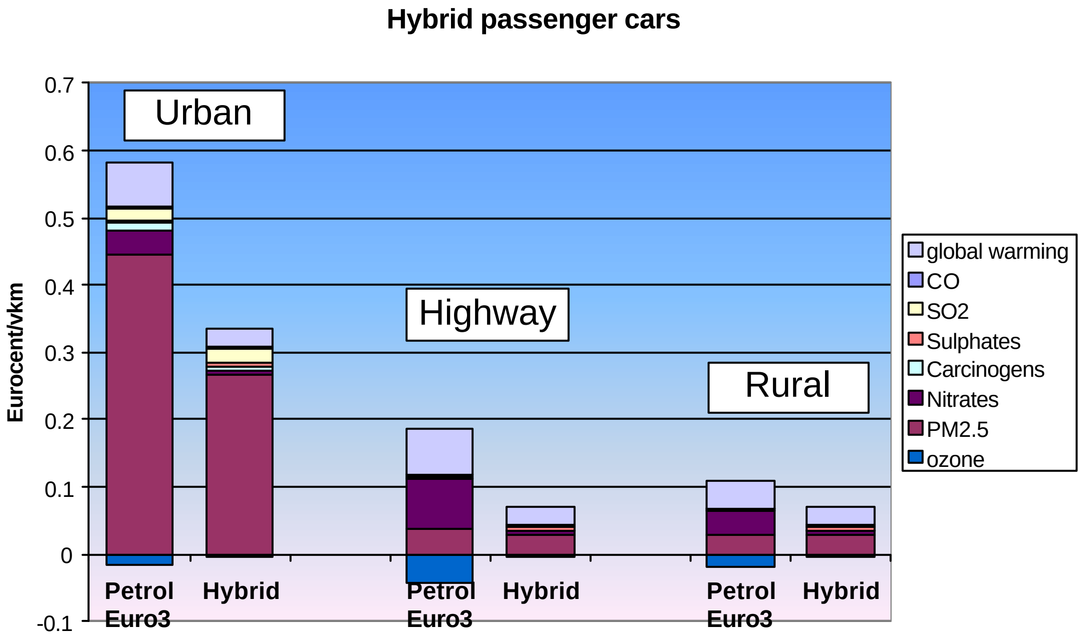


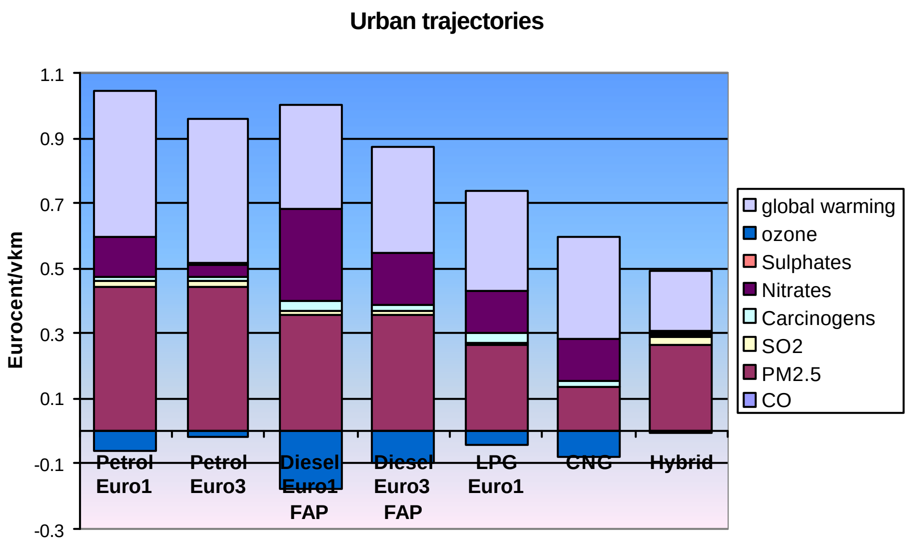
Figure 29: External costs of hybrid passenger cars compared to contemporary petrol cars

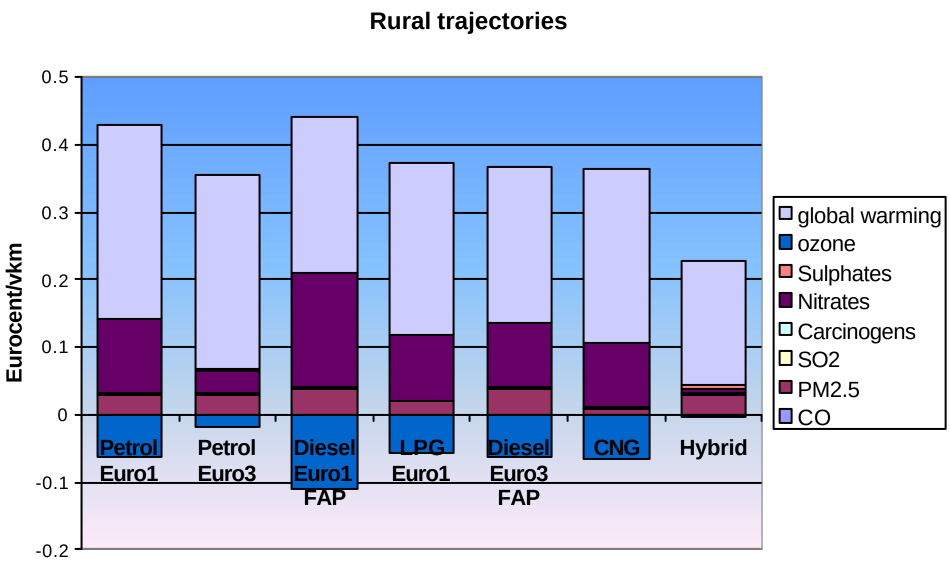
2.3.4.1.i. Comparison of alternative fuels and technologies

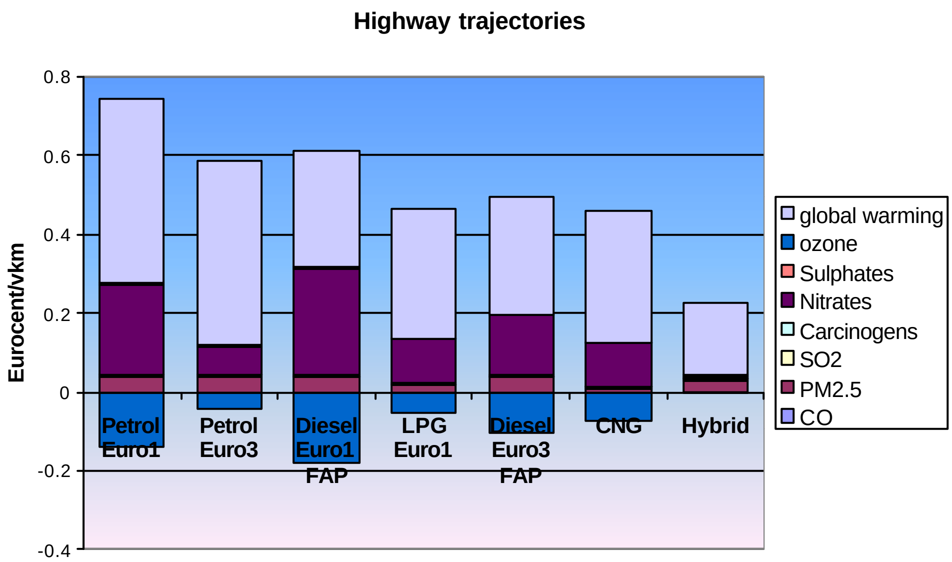
In the figures listed below, we have made a comparison of the use-phase externalities modern petrol cars and some alternatives. From all results discussed in previous paragraphs, we have emphasised existing or emerging technologies that are currently available on the market as these may be more relevant with respect to policy decisions.

We can summarise the results by saying that:
- if diesels are equipped with PM filters that can reduce their emission to the level of petrol cars, they can no longer be considered more polluting that petrol cars. Not even in densely populated cities.
- LPG, CNG and Hybrid vehicles achieve somewhat lower externalities than modern petrol cars. Besides the presumably lower PM emissions they also have lower greenhouse impacts.

- On highway and rural trajectories, there is little difference between diesels with PM filters (FAP), LPG and CNG.
- Hybrid vehicles have the lowest externalities on all trajectories. Unfortunately we have no detailed information for the Life Cycle costs (including the battery) which will determine if hybrids are really an environmental success.
- Biofuels have the advantage of being CO2 neutral, which could be important for extra-urban trajectories. However there is no information on the combination of these fuels with sophisticated engines and after-treatment.







2.3.4.2. Alternative modes of transportation

2.3.4.2.a. Alternative modes of passenger transport

The most important and widely available alternative mode of transport is rail transport. In this paragraph we only discuss trains. Electric trams are briefly discussed in paragraph 2.3.4.2.c. Other forms of passenger transport: aviation, subway and ferries are alternatives that only serve a limited goal in the Belgian context and require different techniques to analyse. Therefore they were omitted from this report.

Results per passenger.km are shown in Figure 30 for fully occupied trains. Occupation however is very variable from virtually empty up to more than 100% (people standing up). The figures are therefore representative only for peak traffic (average occupation 90%; NMBS pers. comm.). Average values can simply be calculated by dividing these results by 3 or 4 (average weekly occupation 20-30%).

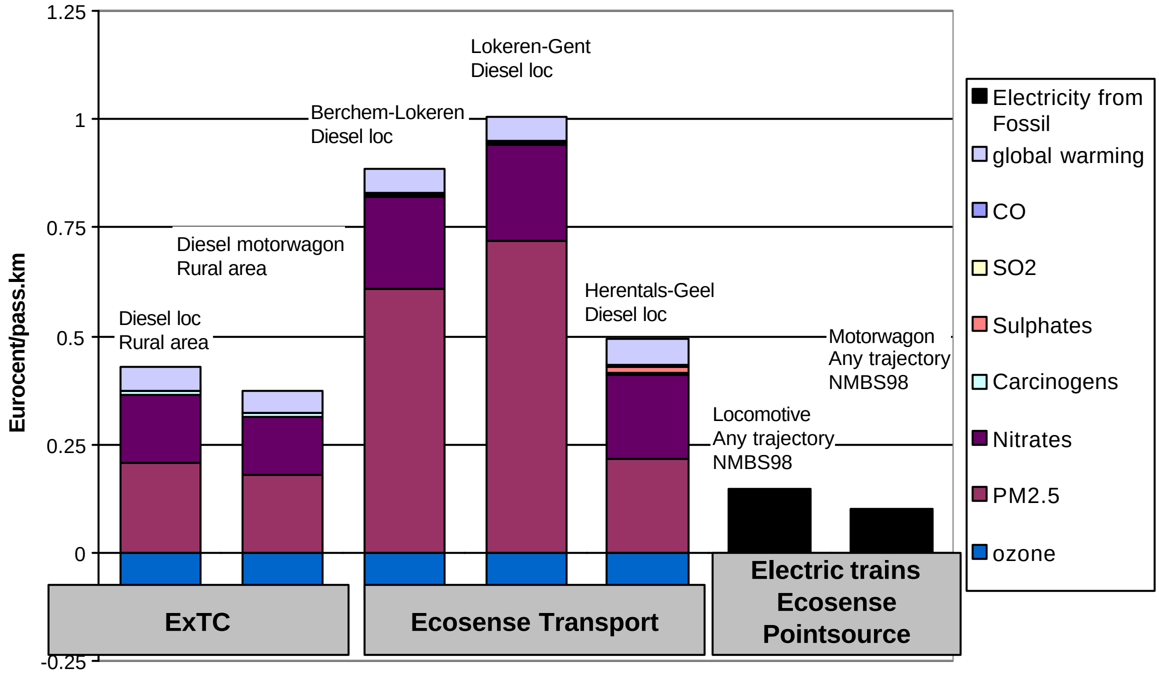


Figure 30: Indicative external costs per passenger.km for trains (all seats occupied)

The first two values were calculated with ExTC for a rural area. Diesel trains and diesel motorwagons have a widely different specific fuel consumption. The externalities of motorwagons are approximately 3 times lower on a per kilometre basis. But because the capacity of both systems differs by about the same ratio, the external cost per passenger.km in Figure 30 is similar. The next three bars display the results of runs for specific trajectories with the EcoSense Transport model. There are clear differences in exposure between the trajectories (since emissions were assumed to be identical). The value for the most rural trajectory in the Campine region is very close to the ExTC estimate based on a simplified world model. The last two bars represent the external cost due the production of electricity for electric trains. In this case the location of the trajectory is irrelevant since all emissions occur at the site of the power plant. The specific electricity consumption per km of an electric motorwagon is only half that of electric trains, but this is partly compensated by their lower capacity. Clearly electric trains perform better than diesel trains, even in rural areas and even when values for the fossil mix of power plants is used. Therefore this seems to be a robust conclusion despite the uncertainty over emissions for diesel trains (see next paragraph). In addition diesel trains have a lower occupation rate than electric trains in Belgium (20% vs 30%) because they are mainly used on rural lines that have not been electrified.

2.3.4.2.b. Alternative modes for freight transport

Rail transport

Although uncertainty about emissions is large for road vehicles, it is even larger for (diesel) trains, up to the point that it troubles our ability to distinguish between or rank transport modes. Despite the fact that the NMBS/SNCF provided us with accurate (albeit average) fuel consumption factors, the range of possible emission factors (expressed as g/kg diesel) is very wide (Figure 31). For the rest of this analysis we have used ETH96 emission factors which are very close to the (unweighted!) average given in MEET.

Results per ton/km are shown in Figure 32. Differences in exposure are clearly visible between different trajectories. Exposure is very low for emissions from electricity plants that emit from high stacks and are located at less densely populated areas. Electric freight trains seem to have a clear environmental advantage over diesel in the use-phase.

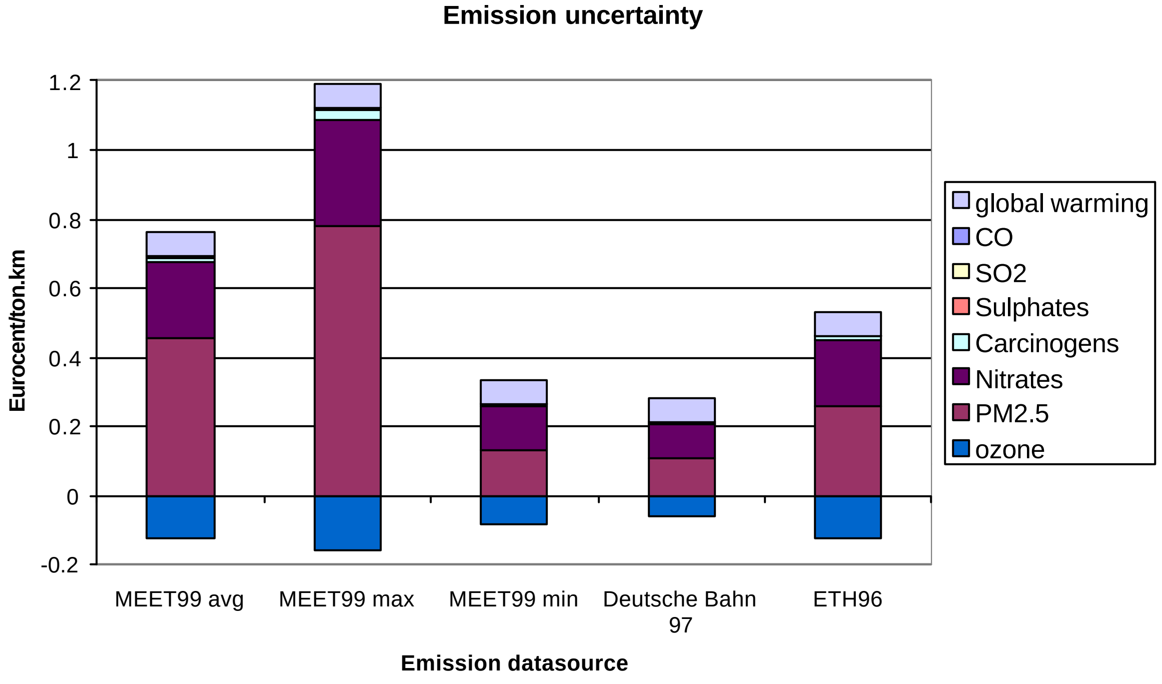


Figure 31: External costs for freight trains based on 6 different emission factors.

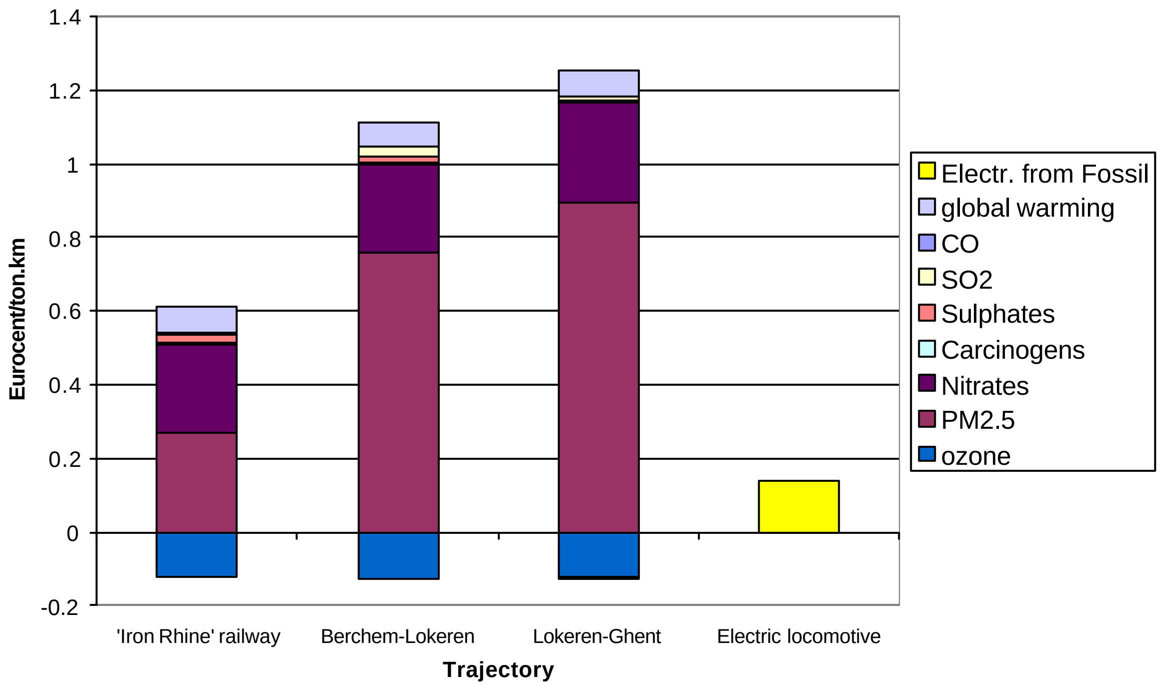


Figure 32: External costs (use-phase) per ton/km for an average Belgian freight train on different trajectories

Inland shipping

First estimates for the marginal external costs for inland shipping were given in Figure 19. These estimates were obtained with the ExTC model (a simplified world model) for a typical rural area. In this way we can compare trucks and ships unbiased by differences in local population density around the trajectories. Figure 19 compares modern inland ships with EURO2 trucks. It is clear that the inland ships have externalities that are similar to those of trucks. Only the largest ships (push tugs with 4 barges carrying 2700t each) seem to have lower externalities per ton/km than large modern trucks.

Table 24

Class	Name	load range (tonne)	Average capacity (tonne)
A	Spits	251-450	350
B	Kempenaar	451-650	550
C	Dortmund-Ems-ship	651-850	750
D	Rhine-Herne-ship	851-1050	950
E	Rhine-Herne ship	1051-1250	1150
F		1251-1800	1550
G		>1800	2250
H	Push tug	2 x 2700	5400
I	Push tug	4 x 2700	10800

In addition to the ExTC estimates, Vito has modified the European EcoSense tool to handle inland ships. An database for 10 different types of inland ships was compiled and linked to EcoSense. The types of ships are listed in Table 24. For the results presented here we have assumed the sulphur content of the fuel to be equal that of the Dutch ship fuel (0,16%). Emission factors were compiled by IVM (The Netherlands) for the European ExternE project (based on Dings et al., 1997 and others). Although Belgium did not present marginal cost estimates for inland shipping in the European report, results presented here can be compared with the results of other countries.

For this report, Vito has made additional detailed calculations for the B-class and I-class ships. I-class push tugs are commonly operated on the largest Belgian canals linking the seaports to industrial areas. B-class ships are the most commonly used ships on rivers and the Campine canals.
The results of these detailed, trajectory specific analyses are shown in Figure 33 and Figure 34. Dominant impacts are caused by emissions of PM2.5 despite the fact that most trajectories can be considered to be in relatively rural areas. Impacts of nitrates are dominant only at the most rural location of Herentals. Ozone impacts are negative for the same reasons that were discussed earlier. Assuming a fuel sulphur content of 0,16%, impacts of SO2 and sulphates are not very important, and only small reductions in externalities can be achieved from using standard diesel fuel (as in trucks). However if ships are operated on heavy fuel with S-contents of 0,8% or higher, SO2 impacts are not negligible. Global warming costs (shown here as the max ExternE value) are relatively unimportant and lower than for trucks. Location effects exist but are smaller than for trucks that drive much closer or even in densely populated areas. Local health impacts of PM and SO2 are responsible for most of the difference.

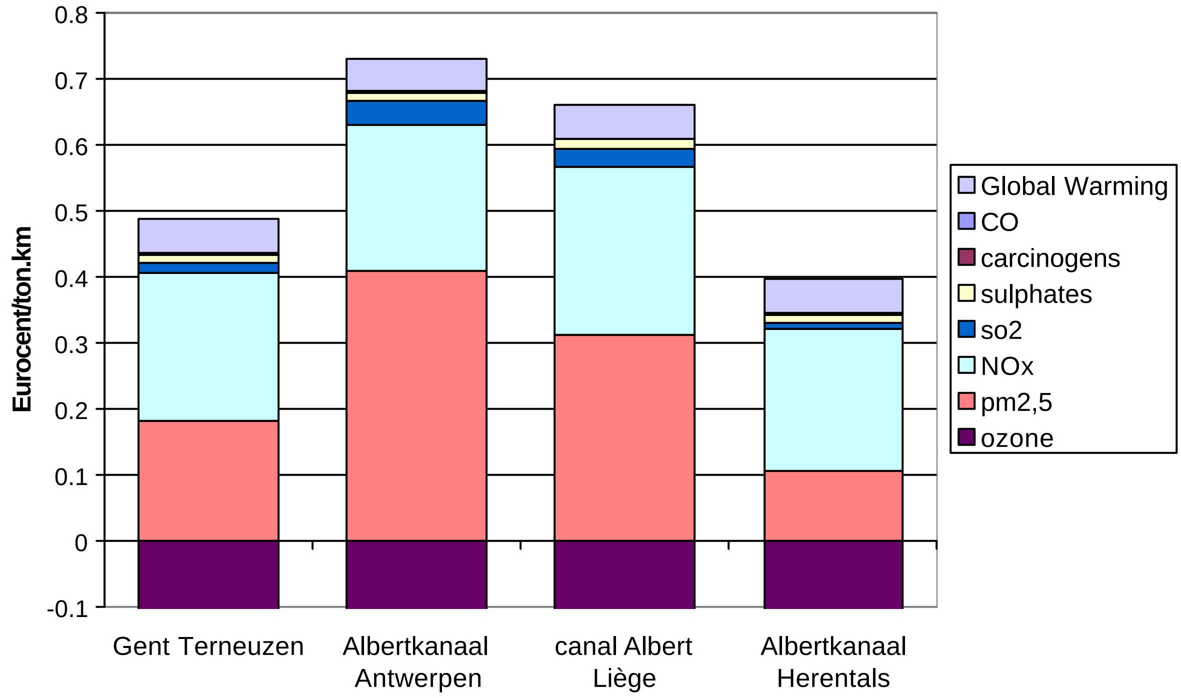


Figure 33: Marginal external costs per tonne-kilometre of small inland ships (class B)

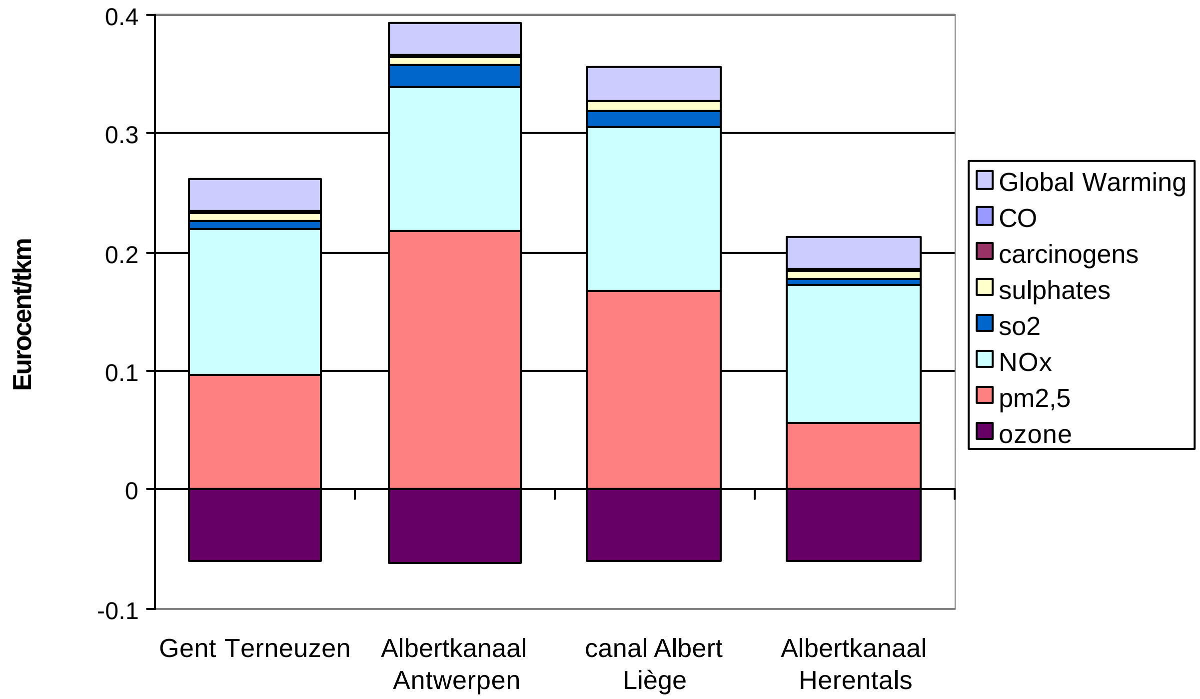


Figure 34: Marginal external costs per tonne-kilometre of large push tugs (class I)

Large ships have higher emissions, but perform better per tonne kilometre than small ships. Only the largest class (I) produces less air pollution than new trucks. Energetically, ships seem to be better.

This analysis shows that ships and trucks have similar external costs per ton.km. Therefore ships can be considered as an alternative mode of goods transport. However before any decision is taken in actual case studies, it is important to take the detailed trajectories of both into consideration. Taking into account that trucks may pass trough more densely populated areas and that ships generate additional emissions during docking, undocking and passing locks (approach, manoeuvre, idle and depart phases-) may overthrow the balance in favour of one mode or the other.
In any case it should not be overlooked that ships may perform better than trucks with respect to noise and congestion and that differences between old and new technologies are far greater than the intermodal difference. Therefore it is important to note that although Belgium is a very important country for inland shipping, its fleet is very old (see Life Cycle costs of Inland Shipping). However it is unknown how old the diesel engines in these ships are and which emission factors should be applied. Even for ships constructed in 1990, emissions are much higher than those of new ships (NOx +10%, PM +300%, VOC +300%, CO +200). It is therefore likely that if fleet weighted average emissions could be calculated, inland ships would compare poorly to the fleet of large highway trucks that have obtained a large penetration of new technologies.
On the other hand, Life cycle cost for inland shipping are much lower (20-25% of the use-phase). In modern trucks Life cycle costs amount from 40-50% of the use-phase costs in rural and highway driving. It may therefore be concluded that if a modal shift can be realised through investment in new ships, this is environmentally beneficial.

2.3.4.2.c. Comparison of traditional and alternative transport modes for passenger transport

Use-phase results for different modes of passenger transport are presented together in Figure 35, Figure 36 and Figure 37 for rural, urban and highspeed transport respectively. These are complemented with data on the Life cycle costs in Figure 38 . The number of passengers used in the calculation in shown on top of the bars. Average occupation was assumed for cars and buses. Full capacity (all seats) was assumed for trains.
From the rural comparison it is clear that even at full capacity diesel trains are no match for modern cars or buses in the use-phase. In fact the result depends strongly on the value that is assumed for global warming. On the other hand electric trains are better than most road vehicles, with the exception of modern coaches (when fully occupied). The interpretation however is completely different when Life Cycle costs are taken into account Figure 38. It turns out then that public transport has a clear advantage over private cars. Fuel consumption (a strong point for public transport) becomes more important when emissions of the fuel cycles are taken into account. In addition the emissions generated during vehicle production are discounted over a much larger mileage than for cars.

We find much higher values for the use-phase of urban transport (i.e. Antwerp or Brussels) compared to rural areas (Figure 40). All diesel vehicles (trains, buses and cars) perform badly, although diesel trains seldom enter large cities in Belgium. Electric vehicles (trains, trams and trolleybus) have very low externalities even at low occupation. Nevertheless it should be pointed out that environmental externalities of a modern petrol car with more than 2 passengers can be lower than for public transport in off-peak situations. When Life Cycle costs are taken into account we arrive at the same conclusion as for rural transport. LCA costs are much more important for cars than for trains. Unfortunately we were unable to calculate infrastructure and vehicle production costs for trams and trolleybuses. Nevertheless it seems unlikely that they would loose their favourable position given the expected life time of the vehicle and the larger number of passengers.

Finally we have assembled a comparison for vehicles that are generally used for long-range and high-speed transport (Figure 41). Again the current diesel cars have the highest use-phase externalities. Externalities of all other vehicles are much lower, especially when only the classical pollutants are taken into account. Remarkably using a modern LPG car or moving people with a chartered coach does just as much environmental damage (through air pollution) as the high-speed trains. Nevertheless there have been technological improvements in HST design that have dramatically improved energy use (per passenger.km) over older models. A comparison that includes Life Cycle cost indicates once again that these are more important than the use-phase for cars, but hardly affect the total for electric trains. We have not extrapolated the costs of vehicle production and building of tracks to high-speed trains since we believe this requires a dedicated study.

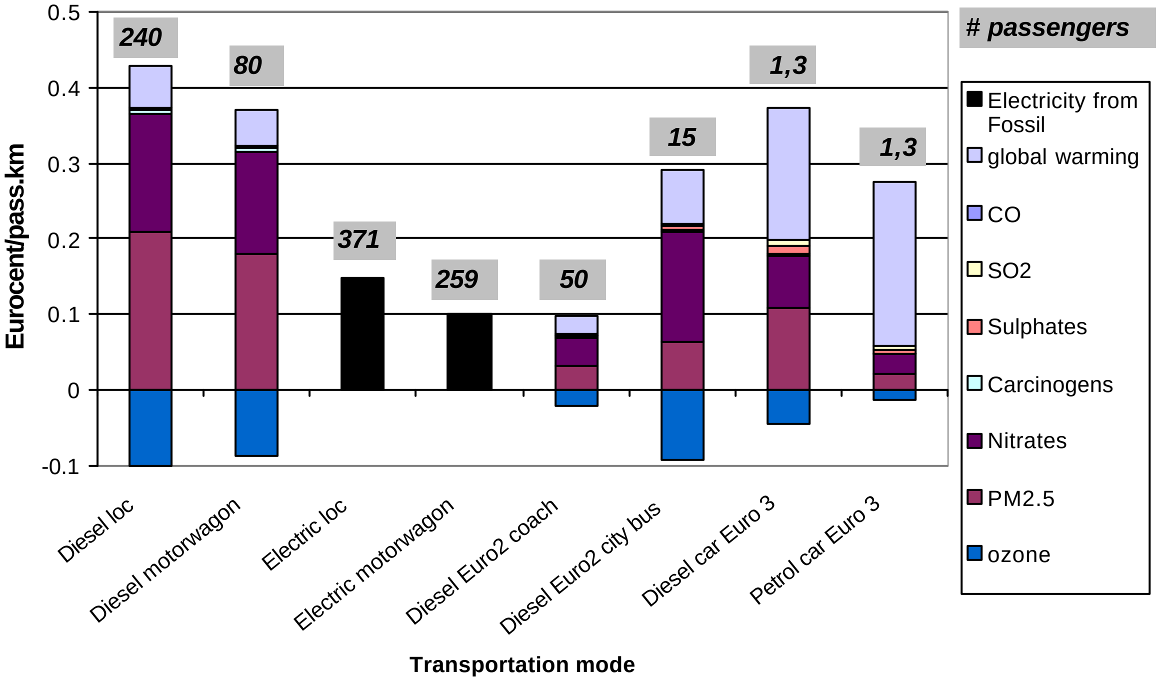


Figure 35: Comparative analysis of alternative modes for rural passenger transport

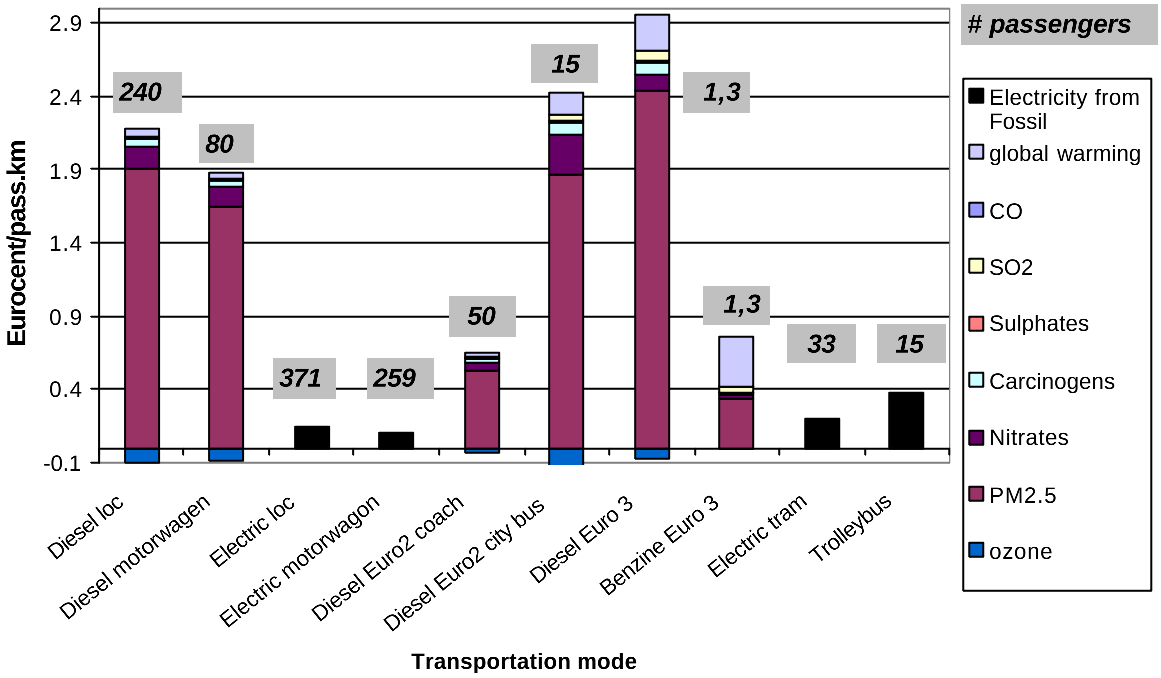


Figure 36: Comparative analysis of alternative modes for urban passenger transport

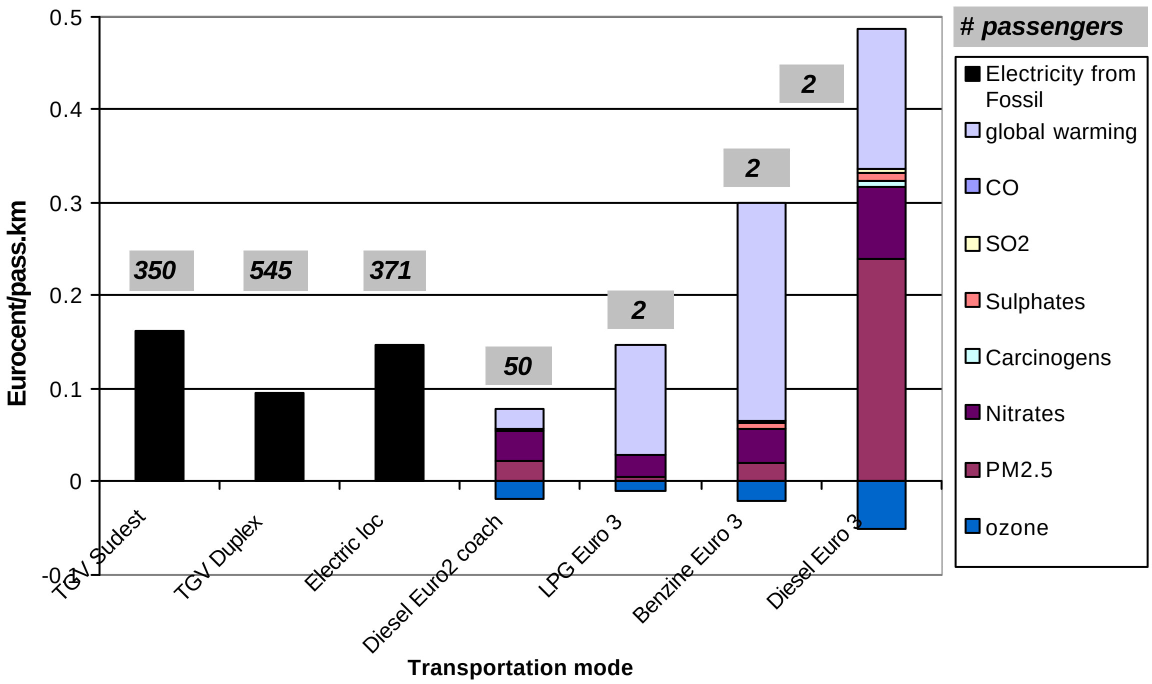


Figure 37: Comparative analysis of alternative modes for highspeed transport

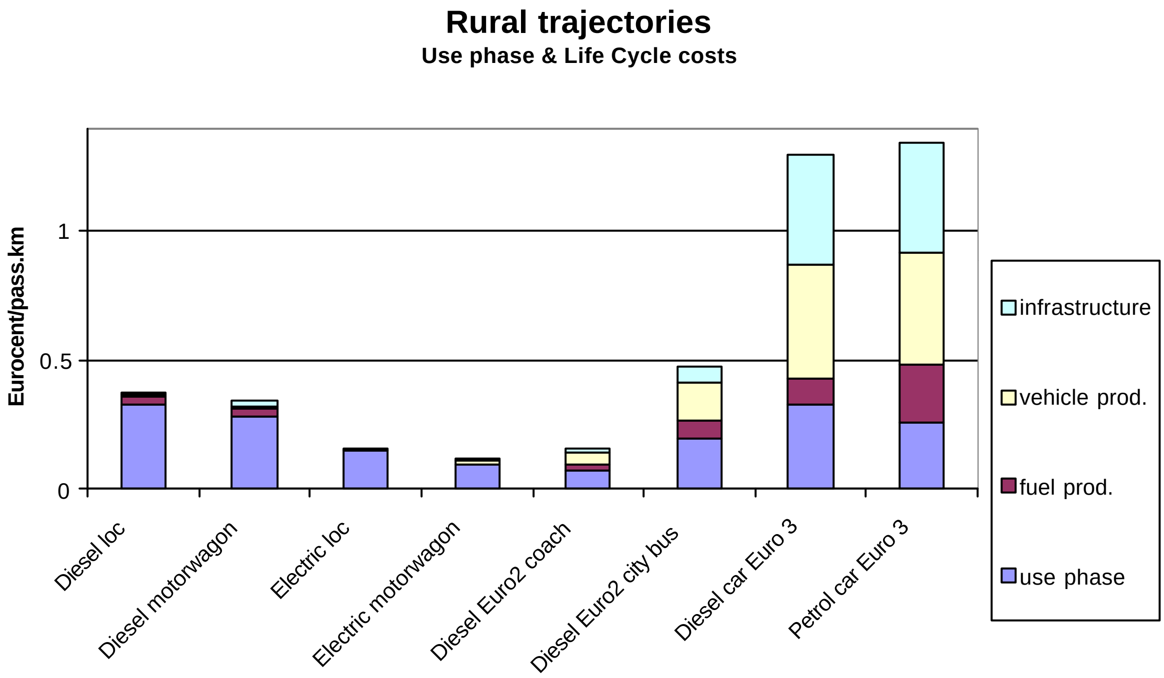


Figure 38: Comparative analysis of rural passenger transport incl. Life Cycle costs

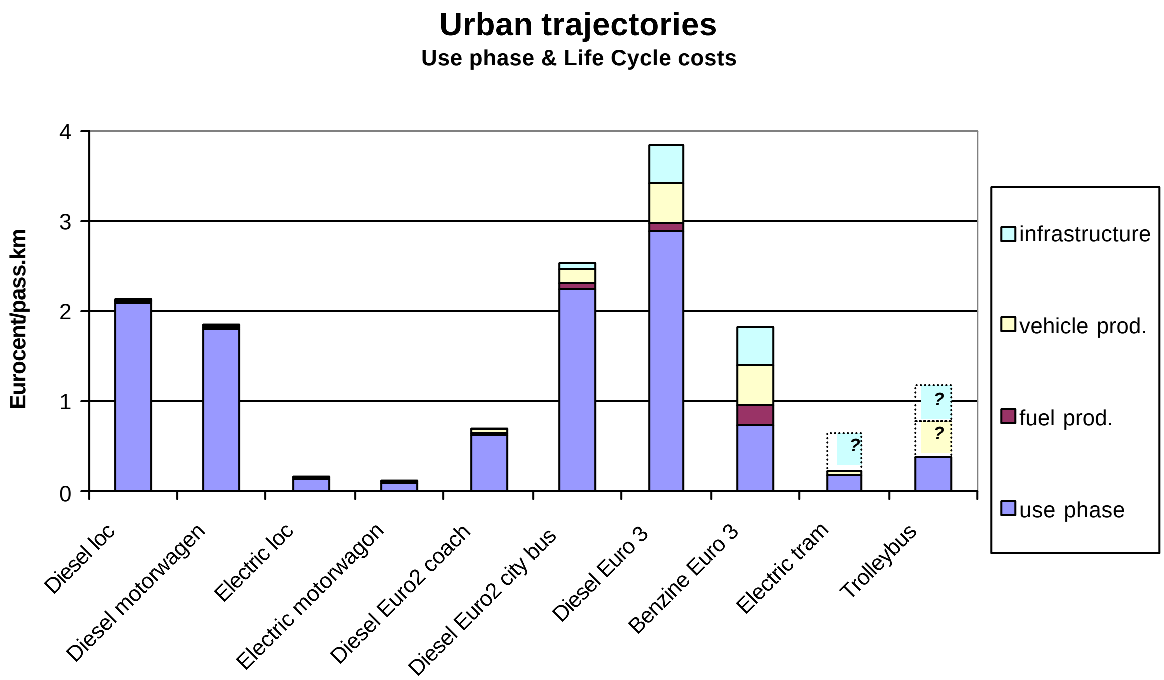


Figure 39: Comparative analysis of urban passenger transport incl. Life Cycle costs

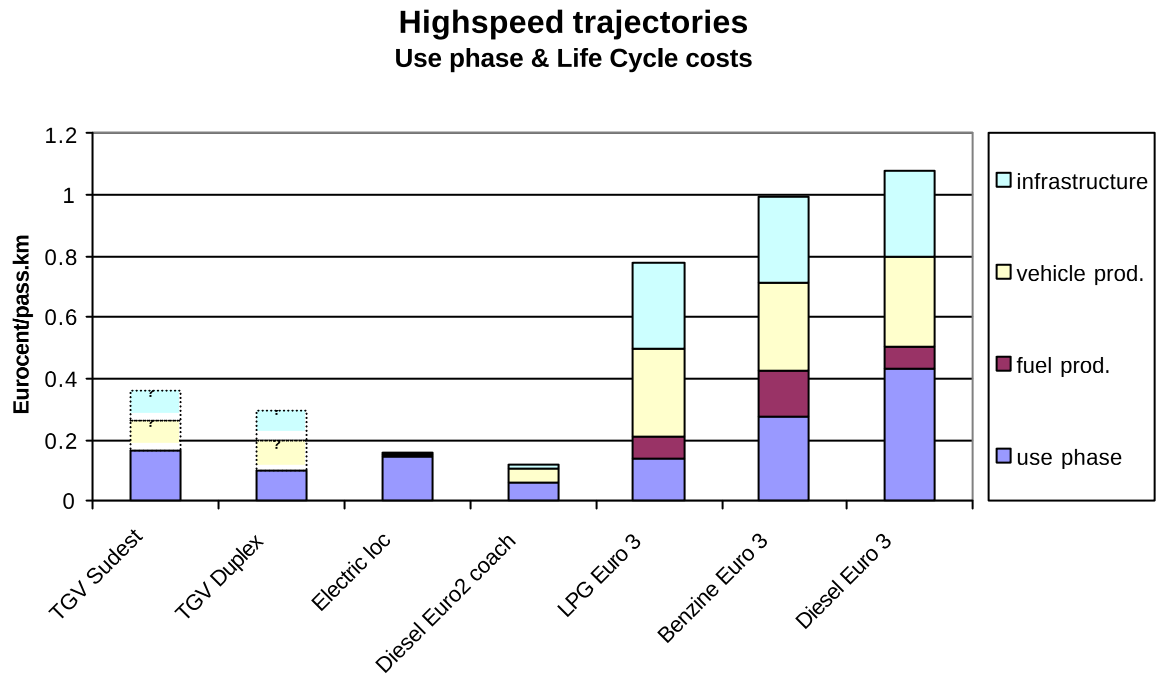


Figure 40: Comparative analysis of urban passenger transport incl. Life Cycle costs

2.3.4.2.d. Comparison of traditional and alternative transport modes for freight transport

Concluding this report, we briefly discuss the comparison of alternative modes for freight transport based on calculations for a rural trajectories in the Campine region. This was chosen as an example because exposure characteristics are similar for the canal, railway and road. Examples for another location can be found in the appendices to this report.

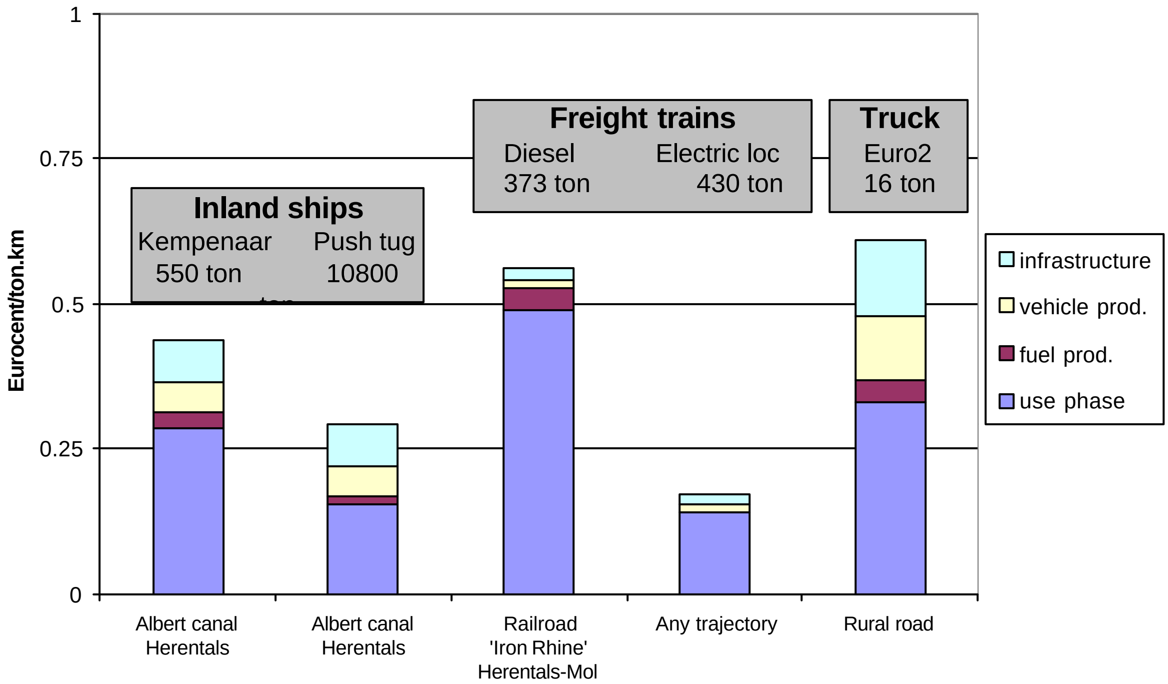


Figure 41: Comparison of different modes for freight transport

Even the most advanced diesel trucks have higher external costs (incl. Life Cycle costs) than any of the other modes. Diesel trains have higher use-phase externalities from air pollution, but lower Life Cycle costs. Inland ships have the lowest use-phase costs. Their Life Cycle costs are lower than for trucks, but higher than for trains. All things combined this leads to the conclusion that (large) inland ships and electric trains are the best option for freight transport especially when population densities are higher than in our reference case. Although we have not attempted to make any estimates for noise externalities, we want to stress that these are not negligible for trucks and trains, but close to zero for ships. Taking this into consideration, inland ships seem a good alternative to road or rail transport from an environmental point of view.

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3. THE MARGINAL EXTERNAL ACCIDENT COSTS

I. Mayeres, S. Proost, D. Vandercruyssen (CES – K.U.Leuven)

The research on the marginal external accident costs has focused on two issues. The first one is the theoretical analysis of these costs. This is treated in section 3.1. The second issue is the monetary valuation of the most important component of the marginal external accident costs: the health impacts. This is discussed in section 3.2.

3.1. The theoretical analysis

One can distinguish two categories of marginal accident costs. The first category includes the costs that are borne by the road user himself and by the other road users. The second category includes the costs that are imposed on the rest of society. The marginal external costs are those costs that are not internalised, i.e. not charged for by means of insurance premia or taxes.

3.1.1. The costs of the road users

The theoretical analysis consists of two parts. A first simple theoretical model considers the marginal external accident costs in a framework that does not consider explicitly the role of liability rules and insurance. In a next step the role of liability rules and insurance is considered in a more detailed way.

3.1.1.1. No explicit modelling of liability rules and insurance

The starting point of the theoretical analysis is a simple theoretical model, described in detail in Mayeres (1999)(see Annex 1). It is based upon existing studies in the transport literature (Newbery, 1988; Jansson, 1994; Mayeres et al., 1996; Johansson, 1997; Peirson et al.,1998) and on the literature on the valuation of health risks [see, for example, Freeman (1993) for an overview of this literature]. The model includes both the monetary and the non-monetary costs of accidents. It assumes that the individual chooses his consumption bundle such as to maximise his expected utility[4] subject to a number of budget constraints. The expected utility is a function of many factors, one of which is the accident risk. The accident risk and the material damage that occurs if an accident takes place, depend not only on exogenous factors, but also on factors controlled by the individual. Examples of exogenous factors are: the traffic flow, the level of care of the other road users or the weather conditions. The model assumes that the individual influences his accident risk and the material damage via the number of kilometres he drives and the safety measures he takes (for example, the installation of an air bag, his driving behaviour).

When the individual decides how much he drives and which safety measures he takes, he only takes into account his own costs and benefits. The marginal private costs consist of the price of these goods. The price includes the sum of the generalised price (resource costs including

[4] The expected utility is obtained by summing over all accident types the product of the probability of that accident type and the utility one can achieve if such an accident occurs. The accident types can be interpreted as, for example, accidents with material damage only, light injuries, serious injuries and fatalities.

the insurance premium, time costs, accident costs) and the taxes minus the subsidies. The marginal private benefits of an additional kilometre consist of the net impact on the expected utility: an additional kilometre increases the individual's utility for a given accident risk, but also affects his accident risk. The same holds for the safety measures, which in addition have an impact on the material damage if an accident takes place.

However, the decisions of the consumer also have an impact on the other road users. One can distinguish the following effects:

- the welfare cost associated with the change in the accident risk of the other road users: this effect is present only if the accident risk of the other road users changes when an additional road user joins the traffic flow or changes his level of care. The change in accident risk is evaluated taking into account the defensive behaviour on the part of the other road users.
- the net costs of defensive behaviour of the other road users: this effect is present if a change in the traffic flow or the level of care of a road user has an impact on the consumption of transport and safety measures by the other road users (a typical example is a cyclist who switches to car use because of the high accident risks for cyclists). It consists of the monetary cost of the defensive behaviour from which the direct impact of defensive behaviour on the expected utility is subtracted. This last correction needs to be made only if the safety measures give rise to direct utility and do not only influence the accident risk.
- the impact on the material damage of the other road users (taking into account their defensive behaviour)

3.1.1.2. The role of liability rules and insurance

The theoretical model does explicitly take into account the impact of liability rules and insurance on the behaviour of the road users. Mayeres (2000) (see Annex 2)presents an overview of the literature dealing with these issues. The paper aims to answer the following questions: Are well designed liability rules – in combination with regulation or not – sufficient to reach the socially optimal level of accident costs? Or is it necessary to complement them with other instruments, such as economic instruments (Pigouvian taxes or subsidies) or insurance regulation?

There exists an extensive literature on these issues. Comprehensive overviews are given in Boyer and Dionne (1987), Shavell (1987), Cooter and Ulen (1997) and Shavell (2000). Most of the analyses use so-called victim-aggressor models. These models make a distinction between two parties, namely the injurers and the victims, with the victims alone experiencing the accident loss. This framework is of relevance, for example, for accidents between motorised and non-motorised transport modes. The conclusions of this literature can be summarised as follows.

Risk neutral agents

In order to focus on the role of liability rules in reducing accident losses, we first assume that the agents are risk neutral. We consider the case of bilateral accidents, that is, both injurers and victims can influence the expected accident losses by their behaviour. The expected accident loss is taken to be determined by both the level of care and the activity level of the parties. Finally, to keep things simple the expected accident losses are assumed to be purely pecuniary.

In this framework certain liability rules lead to an efficient level of care by both parties. This means that the marginal social benefits of their precautionary behaviour is internalised. This is the case for all rules involving negligence (e.g., pure negligence rule, comparative negligence, strict liability with contributory negligence) if the legal standard is set at the socially optimal level.

However, there exists no liability rule that always results in optimal levels of activity for both parties. For one of the parties the marginal social costs of his activity will be partly external. This implies that the liability rules need to be accompanied by another instrument, such as Pigouvian taxes or subsidies, in order to control activity levels.

These conclusions hold under a number of assumptions: the accident losses are purely pecuniary, the legal standard includes all relevant dimensions of care, differences between parties can be assessed at low costs, the court and the parties make no errors etc. The implications of relaxing these assumptions are summarised in Mayeres (2000).

Risk averse parties

With risk averse individuals the social optimum involves not only the reduction of accident losses but also the protection of risk averse parties against risk. Risk averse agents will purchase insurance coverage. What are the implications of this for the incentives associated with liability? Will insurance mediate these incentive effects? Moreover, the problem is complicated by the existence of the moral hazard problem. This arises if the insurer cannot observe the behaviour of the insured and therefore cannot adjust the insurance premium in function of this behaviour.

In the literature these aspects have been treated within the framework of the unilateral accident model, in which only the injurer has an impact on the expected accident costs, through his level of care and activity. A crucial finding for the basic model is that it is socially undesirable to interfere with the sale of liability insurance.

Up to now it was assumed that the accident costs are purely pecuniary. However, transport accidents often result in non-pecuniary losses (fatality, injury). Within the unilateral accident model the literature shows that these non-pecuniary costs can be internalised by a combination of the strict liability rule and a fine, or by the negligence rule. No interference is desired in the liability-fine insurance market.

3.1.2. The costs for society

The additional road user also causes costs to society. These include medical costs, police costs, the net-output loss and possibly the reduction of labour productivity. Whether this category should also include a value for the pain, grief and suffering of relatives and friends depends on the form of altruism which occurs. This is discussed by Jones-Lee (1989). If altruism means that one is concerned only for the safety of other people, then one should include a value for the pain, grief and suffering of relatives and friends. However, if altruism means that one is concerned for the general welfare of others (which depends not only on their safety, but also on other factors), then it should not be included in order to avoid doublecounting.

3.2. The monetary valuation of the health effects of accidents

An important input in the calculation of the marginal external accident costs for Belgium is the monetary valuation of the health impacts of accidents. This includes not only the pure economic costs (medical costs, income losses etc.) but also a measure of the loss of enjoyment of life in the case of an injury or fatality. The first category of costs can be valued relatively easily. The second category is more difficult to value. In the literature estimates vary by a factor 4. For this reason the project includes a survey of the Flemish population to determine its value.

The project uses surveys to determine the value of a statistical life or a statistical injury. This is defined as the monetary value of the avoidance of one death or injury, irrespective of who is saved. The methodology used for the surveys is described in detail in Mayeres et al. (2001) (see Annex 4).

Several techniques can be used to derive the value of a statistical life/injury. Generally speaking, two groups of valuation techniques can be distinguished: revealed preferences (RP) and stated preferences (SP). The RP method is based on observed choices actually made by people. These choices entail an implicit or explicit trade-off between money and risk. The SP method asks the respondents in a more or less direct way what their willingness-to-pay is for a hypothetical change in accident risks. It is the latter approach that is applied in this project. There are two main reasons for this. First, there is a lack of data on real risk-money choices in the transport sector. Secondly, the RP method is plagued by a number of problems (e.g., multicollinearity, self-selection).

Various SP methods exist. The experience in other countries was surveyed. There is not yet a consensus in the literature on which is the best method. Therefore, it was decided to compare three methods on the basis of three relatively small surveys. The three methods are: the contingent valuation method (CV), a combination of CV and standard gamble (CVSG) and choice experiment (CE). In the construction of the questionnaire some co-ordination was aimed for with the team of Prof. Rietveld at the Free University Amsterdam, which is studying the same problem.

In the CV questionnaire (see Annex 5) the respondents are asked to express their willingness-to-pay (WTP) for a reduction in the risk of fatal and/or non-fatal traffic accidents. The questionnaire is based on Jones-Lee et al. (1985), Beattie et al. (1998) and Jones-Lee et al. (1998). Three variants of the questionnaire were made in order to test for problems such as the embedding, scope and sequencing effects, which were identified in previous CV studies [see Beattie et al. (1998)]. These problems are related to the fact that the accident risks in transport are very small.

The CVSG questionnaire (see Annex 6) is based on Carthy et al. (1999). It proceeds in two steps. In step 1 the CV method is used to determine the WTP for a complete recovery from a non-fatal light injury. In addition the respondents are asked for their willingness-to-accept for the same injury. Step 2 uses the standard gamble method. The respondent is told that he/she has been involved in a traffic accident and will die if he is not treated. The respondent is asked to make a choice between two treatments, which have a different risk of failure (which results in death) and different outcomes if they are successful. The study of Carthy et al. indicated that the CVSG questionnaire is understood better by the respondents and that the method is plagued less by the problems of the CV method. However, Carthy et al. also pointed out the possibility of consistency problems. Therefore two versions of the questionnaire are used in this project in order to test whether these problems arise.

Finally, the CE questionnaire (see Annex 7) asks the respondents to make repeated choices between two roads that differ in terms of three characteristics: travel time, number of fatal accidents and the price of a trip. The questionnaire is based on Rizzi et al. (1999). This method is relatively new in the domain of transport safety valuation. This project will test whether it is suited for the monetary valuation of accidents.

The practical organisation of the surveys proceeded as follows. The surveys were carried out by INRA Belgium, which was chosen after a comparison of the tenders by several firms. The surveys involved three steps:
(i) a focus group in which a small group of people discussed about their attitudes towards transport safety problems
(ii) the testing of the questionnaires
(iii) the final survey (288 respondents per questionnaire; a total of 864 interviews)
The surveys were carried out in August-September 2000.

At this moment the analysis of the survey data has not yet been completed. The work will be continued in the near future. The aim is to compare the three survey techniques and to assess their strengths and weaknesses.

3.3. The development of an Excel spreadsheet

An Excel spreadsheet was developed on the basis of the theoretical model presented in Mayeres (1999)(see Annex 3). It allows to calculate the marginal external accident costs for different assumptions about the relationship between the accident risk and the traffic flow and about the valuation of the accident costs.

3.4. Publications and working reports

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4. THE MARGINAL EXTERNAL CONGESTION COSTS

B. De Borger (UFSIA)

Note on the research team:
Importantly, it turned out (potentially due to the very technical nature of the project proposal) to be extremely difficult to find qualified personnel to work on the project. Two different researchers worked briefly on the project, but most of the time no researcher could be employed on the project. The unfortunate consequence has been that (i) less output has been produced than would have been desirable and (ii) the application of the theoretical model developed for the project, in order to derive implementable policy results, has remained extremely limited.

4.1. Introduction

The standard approach for the determination of the external congestion costs in economic policy models assumes a static framework and a very simplified spatial environment. It consists of determining, for a given trajectory, the empirical relationship between the traffic flow and the average speed of that flow. This is based on the idea that an increase in traffic flow influences average speed and, therefore, the time needed to make a certain trip. Time losses due to congestion are valued negatively by the travellers. The marginal external congestion cost is then defined simply as the total value of the time losses for the other road users due to an additional vehicle. The calculation of the marginal external congestion costs requires an estimate of the impact of an additional vehicle on the average speed of a traffic flow and the valuation of the time losses.

This project aims to give a better estimate of the marginal external congestion costs by taking into account a number of complications which were ignored up to now in the existing studies for Belgium. The first extension is the introduction of dynamics. This refers to the dynamic adjustment of departure times (and therefore the time of travel) which is caused explicitly by congestion. Indeed, one observes in reality that congestion induces people to adapt their travel behaviour (leaving earlier or later, choosing another mode or route etc.). The consequences of these endogenous adjustments were not considered in previous models for Belgium. Recent theoretical work [Arnott et al. (1993), Noland and Small (1995), Noland (1997)] allows to incorporate this phenomenon and to determine its impact on congestion and external costs. The second extension is related to uncertainty. It is important to consider non-recurrent and structural or recurrent congestion simultaneously. Traffic jams are not only a structural phenomenon (recurrent congestion: demand exceeds capacity), but also partly dependent on stochastic and non-perfectly predictable elements (weather conditions, accidents...). People take into account the probability of unexpected circumstances in function of the available information. However, the variability of the unpredictable circumstances plays an important role in the behaviour of the commuters and in the determination of the observed level of congestion. A third extension concerns the possibility to reduce the uncertainty about congestion by giving specific information to the travellers. Which information has a positive impact? Is the provision of information always welfare improving? In a final stage, an explicit spatial component could be added in the framework by introducing an explicit network structure.

The remainder of this section is structured as follows. In section 4.2 we describe the methodology developed for the project. We provide the intuition underlying the model, we present the theoretical structure, and we describe how the theoretical model was operationalised numerically. Section 4.3 then briefly summarises the most important results obtained on the basis of a simple numerical application of the model. In section 4.4 some policy conclusions are suggested. The research papers written in the course of the project are provided in Annexes 8 and 9.

4.2. Methodology

4.2.1. Introduction

To analyse the various questions raised, different approaches to modelling the congestion phenomenon were considered. In an introductory analysis, the differences between bottleneck models (see, e.g., Vickrey (1969), Arnott et al. (1993)) and approaches based on speed-flow relations (see Henderson (1977), Chu (1995), Noland (1997)) were carefully analysed. It was shown and numerically illustrated that both approaches are based on very different assumptions, but that they yield the same results under specific parameter values. In general, however, they lead to different estimates of marginal external congestion costs. A summary of this research is reported in Annex 8.

A second conclusion drawn from this research was that the Henderson-Small-Noland approach was probably best suited for the purpose of the modelling exercise envisaged in this project. As a consequence we developed simple numerical models along the lines suggested by Noland (1997). For more details about the model and its application we refer to Annex 9.

4.2.2. Intuition of the modelling approach

In order to understand the intuition behind the model a simple version with one trajectory is the most appropriate. An extension to a more formal network structure is technically complex, but conceptually relatively simple.

Suppose that a group of *N* commuters has to travel on a certain trajectory by car. Each commuter has a desired arrival time at the end of the trajectory. Because of differences in the time at which work starts, differences in preferences, and variability in the distance to be travelled after the trajectory these desired arrival times can vary strongly between commuters. The desired arrival times are described by a distribution, which in our case is assumed to be normal with a mean at 8.30 a.m.

Each commuter determines his ‘optimal’ departure time in function of the desired arrival time taking into account two types of congestion. On the one hand there is recurrent congestion: the traffic flow on the trajectory determines the average speed. On the other hand there is also a probability of unexpected additional congestion; the time needed to get out of the resulting traffic jam follows a statistical distribution which is assumed to be exponential in this exercise. This reflects amongst other things that the probability of short delays is larger than that of longer incidents. Each commuter is assumed to determine his optimal departure time in order to minimize the total expected cost of the trajectory. In the end this procedure leads to a stable congestion pattern that can be used to determine various economic measures.

4.2.3. Model structure

The various time components in the model can be represented as follows. When there is free-flow traffic, the time needed for the trajectory is T_f. Recurrent congestion adds a component $T_x(t_h)$. This depends on the traffic flow which depends in its turn on the departure time t_h. The stochastic non-recurrent congestion is represented by T_r. This can be described by a probability distribution. In our example the non-recurrent congestion is assumed to follow an exponential distribution with parameter *b* (which gives mean and standard deviation). The total trajectory time is therefore $T = T_f + T_x(t_h) + T_r$.

This implies that someone who wants to arrive at t_w can never determine his departure time such that he arrives exactly on time. If he overestimates non-recurrent congestion, he is too early, if he underestimates it, he is too late. The maximum time an individual wants to arrive too early (i.e. when there is no non-recurrent congestion) is called the 'head time' and is given by T_e. This variable is determined by

$$T_e = t_w - t_h - T_f - T_x(t_h)$$

Each individual minimizes the expected cost of the trajectory when determining the desired departure time. He takes into account not only the transport costs, but also the costs of arriving too early or too late. The cost function is given by:

$$C = \boldsymbol{a}T + \boldsymbol{b}(SDE) + g(SDL) + \boldsymbol{q}D_L$$

in which *T* is the total travel time, *SDE* is the time one arrives too early, *SDL* the time one arrives too late and D_L is a dummy variable which equals one if one arrives too early. The parameters give the value that the individual associates with travel time, early arrival time and late arrival time and a discrete fine when arriving too late. The valuation of these costs can in principle differ between individuals in function of work organisation rules, preferences etc.

Taking into account the distribution of desired arrival times and the distribution of non-recurrent congestion each individual minimizes the expected cost *E*(*C*). It can be shown that the cost minimizing head time T_e is given by

$$T_e = b \ln \left[\frac{q + b(\boldsymbol{b} + g)}{b(\boldsymbol{b} - a\Delta)} \right]$$

where Δ is the change in recurrent congestion when the commuter departs a bit later. This is determined as

$$\Delta = -\frac{dT_x(t_h)}{dT_e} = -T_x^{'}(t_h).(\frac{dt_h}{dT_e})$$

Taking into account the free flow time and recurrent congestion (which depends on the traffic flow) this allows calculating the desired departure time for each individual. An iterative procedure allows determining stable congestion profiles for which each individual minimises his expected costs and takes into account changes in traffic flows over time.

4.2.4. Operationalising the model

In order to infer optimal congestion profiles and to determine the various components of congestion costs, the optimal decisions of all individuals need to be coordinated. Analytically one searches for a Nash equilibrium. To compute its characteristics in this context is not evident.

To operationalise the model for numerical application, we proceeded as follows. We start from a given distribution of desired arrival times. The period to be analysed is divided into small time intervals (e.g., 5 minutes, 15 minutes). Given the distribution of non-recurrent congestion and therefore a given degree of uncertainty, each individual determines his optimal head time T_e, conditional upon a given basic congestion pattern. This allows to determine the traffic flow in each interval. This can be used to compute the change in the expected time cost when someone departs slightly later (the discrete equivalent of Δ which is defined above). This leads to a number of adjustments in travel times of individuals. Changes in the congestion profiles then lead again to adjustments in the optimal head times and the traffic flow per interval. Iteration of this procedure continues until a stable congestion pattern is obtained.

4.3. Results

Application of the model yields the following general insights. For more details we refer to Annex 9.

1. An important part (20%-40%, depending on the circumstances) of the external congestion costs are <u>adjustment costs</u> in travel behaviour. By looking only at the role of travel costs, many previous models have calculated the marginal congestion costs incorrectly. A growing travel demand leads to higher time costs and also to important additional time adjustment costs.

2. The marginal congestion costs depend strongly on the capacity and the desired arrival time. They vary between almost zero in the off-peak period to more than 50 BEF (1.13 Euro) per km in the high peak and with relatively low capacity.

3. An increase in <u>capacity</u> reduces both the recurrent (structural) congestion and the congestion due to unforeseen circumstances. An increase in capacity changes the congestion profile (more clustering around the peak because of higher capacity), leads to a lower global congestion costs and increases the relative importance of the adjustment costs. The impact on the average head start time is small. Higher capacity reduces the spreading of the peak period.

4. A reduction in the <u>uncertainty</u> about non-recurrent congestion leads to a shift in time of the congestion profile. With higher reliability people leave later and cause a peak at a later moment. The share of the scheduling costs diminishes, the share of the costs due to late arrival increases (people leave somewhat later so that costs of arriving too early are reduced and costs of arriving too late increase). The share of transport costs increases when there is less reliability.

4.4. Policy implications

The main policy implications can be summarised as follows.

1. Previous estimates of marginal external congestion costs may have been drastically biased by ignoring people's adjustment costs in response to congestion. A very substantial fraction of congestion costs relates to having to travel at less desirable times as a consequence of congestion.

2. The role of time-differentiated pricing instruments is reinforced by the dynamic adjustment to congested facilities.

3. Another implication is that investments in techniques that reduce the uncertainty about congestion can be much more effective in reducing congestion than direct investments in capacity.

References – Section 4

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5. THE USE OF MARGINAL EXTERNAL COST INFORMATION

Information on the marginal external costs of transport is a crucial input in the evaluation of transport policies. This section gives an overview of recent CES and UFSIA publications which have benefited from the work on the marginal external costs and which were realized during the project's time span.

Journal articles

Mayeres, I. and S. Proost, 2001, Marginal tax reform, externalities and income distribution, Journal of Public Economics **79**, 343 – 363.
Proost, S. and K.Van Dender, 2001, Welfare impacts of alternative policies to address atmospheric pollution in urban road transport, Regional Science and Urban Economics, forthcoming June 2001.
Mayeres, I., 2000, The efficiency effects of transport policies in the presence of externalities and distortionary taxes, Journal of Transport Economics and Policy **34**, 233 – 260.
Mayeres, I., 2000, Een economische analyse van het mobiliteitsvraagstuk, Onze Alma Mater **54**, 281 – 292.
Mayeres, I. and S. Proost, 2000, Should diesel cars in Europe be discouraged?, Regional Science and Urban Economics, forthcoming June 2001.
Proost, S., 2000, Climate change, urban air problems quality and transport policy in the European Union, Integrated Assessment **1**, 145 – 156.
Proost, S., D.Van Regemorter, F. Lantz and V. Saint-Antonin, 2000, Limiting air pollution from transport: economic evaluation of policy options for the European Union, International Journal of Global Energy Issues **14**, 320 – 330.
Proost, S. and D. Van Regemorter, 2000, Are there cost-efficient CO_2-reduction possibilities in the transport sector? – Combining two modelling approaches, International Journal of Vehicle Design **24**, n. 2/3, 1 – 15.
Proost, S. and D. Van Regemorter, 1999, Are there cost-efficient CO_2-reduction possibilities in the transport sector? – Combining two modelling approaches, International Journal of Pollution Control **11**.
Calthrop, E. and S. Proost, 1998, Transport and environment - interaction between theory and empirical research, Environmental and Resource Economics **11**, n°. 3 – 4.
De Borger, B. and S. Proost, 1998, Mobiliteit, de juiste prijs in België, Economische Statistische Berichten.
Proost, S., 1998, De juiste prijs voor het verkeer in de stad, Onze Alma Mater **52**, 79 – 89.

Books and contributions to books

De Borger, B. and S. Proost (eds.), Reforming Transport Pricing in the European Union – A Modelling Approach, Edward Elgar, forthcoming.
O'Mahony, M., S. Proost, K. Van Dender, 2000, Optimal Urban Transport Pricing and Sustainability, in: K. Rennings, O. Hohmeyer and R.L. Ottinger (eds.), Social Costs and Sustainable Mobility – Strategies and Experiences in Europe and the United States, ZEW Economic Studies, vol. 7, Physica (Springer), Heidelberg – New York.
Proost, S., 2000, Present inefficiencies in European Transport and Environment policies, in: J. Maxwell and J.Von Hagen (eds.), Environment in Europe, Kluwer, forthcoming.

Mayeres, I., 1999, The control of transport externalities: A general equilibrium analysis, Ph.D. dissertation, Faculty of Economics and Applied Economics, K.U.Leuven.

Van Dender, K., S. Proost and S. Ochelen, 1999, An economic modelling approach for the reform of urban transport pricing, in: H. Meersman, E. Van de Voorde and W. Winkelmans (eds.), World Transport Research – Selected proceedings from the 8th World Conference on Transport Research, Antwerp, Belgium, 12-17 July 1998, Volume 4: Transport Policy, Pergamon, Elsevier Science.

6. CONCLUDING REMARKS

The previous sections show clearly that the research results are the most concrete for the air pollution costs of transport. In the other areas contributions were made to a more correct calculation of the marginal external costs. However, the research in those areas is not yet in a stage that the figures of previous studies can be revised. This difference in progress for the various external cost categories corresponds to a large extent to the state of the art in the literature. While the methodology for air pollution costs is defined relatively clearly, research is still very much in progress for some environmental costs (noise, ecological impacts), for accident costs and for congestion (for example, the dynamic adjustment of departure times, the treatment of uncertainty and the effects of the provision of information).

Nevertheless, the project has enabled the three research teams to further develop their know-how about the marginal external costs. The project has allowed the three teams to be among the top scientific groups active in this area. They play an important role in several European research consortia on the use of external costs of transport (ExternE, UNITE, MC-ICAM). The know-how created in the project will be extremely useful and crucial for the evaluation of policies which aim to reduce the social costs of transport. Interim and draft final results of the project have already been widely spread – both to the scientific world in different related disciplines, and to relevant policy actors. The new insights (e.g., relative importance of health impacts from particles, concept of external congestion costs,..) have been used for policy-oriented studies and policy preparation.

The project has allowed us to identify several new avenues for future research:

For the accident costs the role of liability rules and insurance systems in combination with economic instruments deserves further research. On the empirical side, the choice of the correct methodology for valuing a statistical life/injury is not yet fully explored. Also the relationship between accidents and their various determinants (speed limits, variation in speed, traffic rules etc.) should be explored in greater detail.

For the environmental costs, a lot of progress has been made in recent years for the assessment of impacts of air pollution. It shows that although emissions of new vehicles continue to decline significantly, our current best judgement is that total impacts remain relatively very high. Although a wealth of data is available, it remains difficult to judge to which extent current policies will guarantee a (from an air pollution point of view) sustainable transport in Belgium. These uncertainties – related to a number of parameters – also make it difficult to judge to which extent a more radical change in technologies and fuels (electric vehicles, fuel cells, biofuels,...) is required to meet long term air pollution objectives. For policy making, both a continuous update and further development of methodologies and data is required. The most important areas for improvement are:

- Keeping up to date with technological developments: both conventional technologies (PM filters, ...) and alternative technologies (hybrid vehicles, ...) change fast, and require updates of emission data and projections.
- Keeping up to date with scientific improvements and reducing uncertainties: the scientific understanding of dispersion, exposure, impacts and their valuation is changing fast, especially in areas related to particulate matter. As this is the major impact category, estimates and uncertainties of external cost data risk to be quickly outdated. New insights

may have important policy consequences, e.g., related to taxation of diesel and petrol. Overall, handling of uncertainty needs to be improved.

- Covering more impact categories: for impacts of noise, new data are becoming available that will allow for more consistent estimates. For impacts on historic buildings, ecosystems and impacts from greenhouse gases, further research is needed, including new approaches to valuation. The integration of estimates for different impacts (air pollution, noise, ...) based on different assumptions need to be further explored, as is uncertainty analysis.
- A better coverage of all transportation means: Whereas transport and emission data are widely available for passenger cars, less data are available for other road vehicles (trucks, motorcycles,...) and even less for rail traffic and inland shipping. Especially data related to current and new technologies are lacking, as are data on how to improve their environmental performance.
- Improvement of estimates of the non-use phases: Both data and tools to assess impacts from the non-use phase of transport are less developed. Main areas for improvement include data for projections to 2010, taking into account new and stricter environmental policies, and more realistic data related to fast developing new fuel cycles: the fuel cell, biofuels and electrical vehicles. On the methodological side, the integration of impacts of emissions to water and soil is an important gap in the information.
- New and indirect impacts. Both methodologies and data are lacking to evaluate the relative importance of a number of ‘new’ and indirect impacts. These include impacts of parking and traffic on the ‘quality of life’ in the ‘city’, benefits of walking and cycling on health, impacts of new infrastructure on landscape and biodiversity. Other issues are road dust (related to the discussion on particles), growing interest for some new pollutants (PAHs) or pollutants not well covered (dioxins)
- Uncertainty analysis related to policy applications: To be useful for policy and decision making, the large amount of available data need to be exploited from the perspective of specific policy questions and uncertainty analysis of these answers need to be developed. Also data to evaluate the impact of specific policy measures (e.g. the introduction of particle filters) need to be further developed. For the assessment of local transport policies, integrated models are required that link detailed traffic-air dispersion models with assessment tools.

For the congestion costs additional work is necessary on at least two issues. First, additional research should investigate the underlying determinants of the value of time losses. This was not explicitly studied in this project. Second, the recently developed dynamic models of congestion have to be integrated in welfare economic analysis of pricing and other policy measures to cope with congestion.

The estimation of the total marginal external costs requires a further integration of estimates on congestion, accidents and the environment and public health, and consistency related to their valuation. There is a need for a set of data that can be used to evaluate the cross-links between policies related to environmental protection, safety and congestion. Especially where policies may be in conflict (e.g. lighter vehicles to limit CO2 emissions versus extra weight of extra safety provisions; higher speeds and therefore lower congestion costs might increase the accident costs), such a set of indicators is required for integrated policy making.