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Emissions of pipeline transport compared with those of competing modes

Environmental analysis of ethylene and propylene transport within the EU

Report

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Summary

Background of this study
The Association of Petrochemicals Producers in Europe (APPE) has the ambition to develop a Trans European olefins pipeline network fully interconnected to facilitate interchange of ethylene and propylene, the main feedstock for the Petrochemical industry. EPDC is a consortium of chemical companies working on a propylene pipeline project to connect Rotterdam with the Ruhr Area via Antwerp and Cologne.

However, the environmental and safety impacts of such an interconnection are yet unclear. An important - but not the only - element of this environmental impact is the energy use of, and hence emissions caused by, transport of ethylene and propylene by pipeline, compared to competing modes of transport.

Therefore, APPE commissioned CE Delft to execute a state-of-the-art overview study of the emissions from ethylene and propylene transport by pipeline as well as competing modes.

Aim and demarcation
The aim of this study is quite specific, namely to compare emissions per tonne kilometre caused by transport of ethylene and propylene by pipeline with those of competing modes.

The study is limited to the emissions of CO_2, NO_x, PM_{10}, SO_2, and VOC[1] per tonne-kilometre of ethylene or propylene transported. The energy use and emissions are compared for the following transport modes[2]:
– pipeline;
– rail;
– inland shipping (barges);
– short-sea shipping.

The study does NOT include:
– a quantitative assessment of the environmental impact of initial compression (required for transport by pipeline, barge and train) or cooling (required for transport by sea vessel) of the ethylene and propylene;
– VOC-emissions resulting from leakage and evaporation of ethylene and propylene;
– other environmental impacts such as noise nuisance, safety aspects, or any impacts from the construction of a new pipeline.

Therefore, this report covers only one, albeit important, element of a full environmental impact analysis of a new pipeline system.

The analysis
Original data on energy use of pipeline transport of oil products appear to be very scarce. All studies found refer to the same study by Mittal (1978). This study therefore was our first information source.

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[1] The comparison of modes on VOC-emissions is not (yet) complete due to limited information on VOC-emissions from evaporation and flaring.

[2] For of safety reasons, road only has a very small market share in ethylene and propylene transport. Therefore road transport is not included in this comparison.

The client provided a second information source, namely concrete information on the number and energy use of pumps and valve stations in a number of cases.

Finally, CE Delft developed a third original source of information, namely a relatively simple theoretical energy use calculation model. The energy use of pipelines turns out to depend on pipe diameter and roughness, density and viscosity of the fluid, and finally flow rate through the pipe.
The data of the different sources seem to match well and fall in the range of 0.11 - 0.18 MJ/tonne kilometre of primary energy. Therefore we chose 0.14 as a middle value and 0.11 and 0.18 for best and worst case analysis respectively.

The energy use of other modes, as well as the emission factors per unit of electric or diesel-driven energy, was based on figures from a previous CE Delft study[3]. Where appropriate, figures were adapted to account for the specific characteristics of ethylene and propylene transport.

In a final step, three specific cases were defined to carry out an environmental comparison, using specific load and detour factors.

Conclusions of the analysis

Energy use of initial compression or cooling

To be suitable for transport, ethylene or propylene needs to be under high pressure or low temperature. In most cases, the commodities come under high pressure out of the cracker, which makes that there is no extra compression required, when it transported under high pressure. However, in other cases extra compression or cooling is needed. The energy use of these initial processes are very high, and will in those cases dominate total energy costs. In the framework of this study it appeared impossible to give a general value for this energy cost, but it appears to amount to several hundreds of MJ of primary energy per tonne.

It is vital to consider this cost when comparing between modes, and a proper understanding of the environmental score per mode requires detailed knowledge of production and use of the ethylene/propylene.

Energy use and emissions of pipeline transport compared to other modes

Therefore, in the framework of this study it is only possible to compare the environmental impact of transport of *propylene* with 'compressed' modes pipeline, rail and barge, under the proviso *that the propylene is not decompressed - i.e. used in a high-pressure process - after transport*. Under this proviso, the analysis yield the following results:

- energy use and CO_2-emissions from propylene transport by pipeline are, per tonne kilometre, about 70 to 80% lower than those of the competing modes in the same case;
- NO_x-emissions from pipeline transport are 70% lower (compared with electric trains) up to even 98% lower (compared with inland shipping);
- PM_{10}-emissions are 70 to 90% lower;
- SO_2-emissions from pipeline transport are comparable to those of diesel trains and barges but considerably lower than of electric trains;
- for a sound comparison of the VOC-emissions, we need more reliable data on the effects of leakage and evaporation.

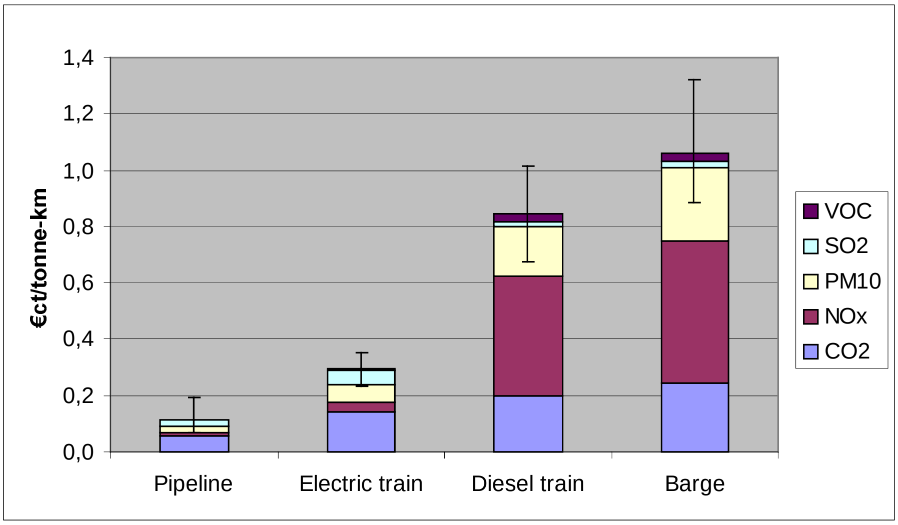
3 To shift or not to shift, that is the question - the environmental performance passenger and freight transport modes in the policy-making context, CE Delft, 2003.

For *ethylene* transport **-** only by pipeline and sea vessel **-** it is not possible to make a quantitative comparison given the differences in energy use for compression (pipelines) and cooling (sea vessel). However, when the ethylene is to be used in high-pressure processes, we would generally expect pipeline transport to require much less energy and produce much lower emissions than transport by sea vessel.

Adding up to one environmental parameter
Several techniques are in place to add different environmental impacts. In this report we use financial valuation of the impacts. All emissions are valued using so-called *shadow prices* that are based either on the damage caused by pollutants or on the social costs to prevent pollution.

For reasons mentioned above, we only present an overview of financially valued emissions of transport of propylene by pipeline, rail, and barge.

Figure 1 Financially valued environmental impact of transport of propylene by four inland transport modes, excluding VOC-emissions from evaporation, *under the proviso that the propylene is not de-compressed - i.e. used in a high-pressure process - after transport*



The conclusion from this figure is that, under the proviso mentioned and excluding evaporative VOC-emissions, in most cases the financially valued environmental impact from **propylene** transport by pipelines is lower than that of the other **inland modes**. Electric trains can in some cases come close. Diesel trains and barges generally score much worse. The impact of sea vessels is not shown in the graph for reasons mentioned, but is also likely to be much higher than that of pipelines and electric trains.

As already said, comparison of sea vessels with other modes is difficult. Nevertheless, it is highly likely that the financially valued environmental impact of **ethylene and propylene transport by sea vessel** generally exceeds that of pipelines, as:

- analysis shows that even if the financial valuation of NO_X, SO_2, PM_{10} and VOC-emissions *at sea* is set at zero, the valuation of only CO_2 generally exceeds that of all emissions from pipeline transport;

- in case ethylene or propylene are cooled prior to transport by sea vessel, the energy requirement for the whole transport chain is likely to increase relative to other modes.

Recommendations
This study certainly provides insight into an important environmental impact of pipeline transport, namely the emissions arising from the use of energy. However, the assessment is too narrow to justify recommendations as to whether the interconnection of the pipeline networks would benefit the environment. To be able to make such a recommendation, the following aspects would require closer investigation:
- the energy use of compression or cooling before or after transport;
- the impact of changes in transport flows and modal split resulting from the use of new pipelines;
- the impact of the construction of the new infrastructure itself;
- the impact of changes in environmental and transport characteristics;
- the changes in safety risks.
- impact of quality grades;

And finally, in order to arrive at a full social cost-benefit assessment of the planned interconnection, economic aspects, such as investment and maintenance costs and operational cost savings, should also be included in the analysis.

1 Introduction

1.1 Background

The Association of Petrochemicals Producers in Europe (APPE) has the ambition to develop a Trans European olefins pipeline network fully interconnected to facilitate interchange of ethylene and propylene, the main feedstock for the Petrochemical industry. EPDC is a consortium of chemical companies working on a propylene pipeline project to connect Rotterdam with the Ruhr Area via Antwerp and Cologne.

However, the environmental and safety impacts of such an interconnection are yet unclear. An important - but not the only - element of this environmental impact is the energy use of, and hence emissions caused by, transport of ethylene and propylene by pipeline, compared to competing modes of transport.

CE Delft has recently published a state-of-the-art report on environmental comparisons of different modes of transport, entitled ‘To shift or not to shift, that’s the question’ |CE/RIVM, 2003|. This report compares environmental data for the principal modes of freight and passenger transport, but does not include pipelines (this omission is made by most reports on this subject).

Therefore APPE commissioned CE Delft to execute a state-of-the-art overview of the emissions from ethylene and propylene transport by pipelines and by competing modes.

1.2 Aim of this report

The aim of this study is quite specific, namely to compare emissions per tonne kilometre, caused by transport of ethylene and propylene by pipeline with those of competing modes.

This report covers only one aspect of a full environmental impact analysis of the new pipeline system. It should be considered as a step towards providing the final answer to the question of how pipeline transport compares to other modes of transport in terms of environmental performance.

1.3 Demarcation

This study is limited to the energy use of the various modes and more specifically to the following types of emissions resulting from it:
- CO_2;
- NO_x;
- PM_{10};
- SO_2;
- VOC.

The energy use and emissions are compared for the following competing transport modes:
- pipeline;
- rail;
- inland shipping;
- short-sea shipping.

Because of safety reasons, road transport is not a relevant transport mode for ethylene and propylene. Therefore, road transport is not included in this comparison. However, a new pipeline may affect the transport volume of polymers by road. These effects are discussed in chapter 8.

This report gives a quantitative analysis of the average emissions per tonne-kilometre. In our analysis, we include the effects of energy use for compression and cooling during transports.

The following effects are not included in this study:
- initial compression or cooling of the ethylene or propylene, before it is transported;
- emissions of VOC due to evaporation or leakage;
- indirect emissions due to energy use for production, maintenance and dissembling of vehicles and infrastructure;
- greenhouse gases other than CO_2;
- impacts of new pipeline infrastructure on the transport market, particularly on the total transport volume and modal split;
- noise nuisance;
- safety aspects.

1.4 Methodology of comparing energy use and emissions of transport modes

The methodology used is generally the same as proposed and applied in the recent study on energy use and emissions of transport modes of CE-Delft and RIVM |CE/RIVM, 2003|.

In this study we include the primary energy use and emissions of:
- vehicles;
- refining of vehicle fuel; and
- electricity production and distribution.

The data are assessed for the year 2010, because this study has a long-term perspective. For most of the modes in this study, the energy use and emission factors will not change dramatically in the next decade. This data presented in this study are representative for the European Community.

Well to wheel approach
This study starts from a well-to-wheel approach. This means that energy use of both vehicles and electricity plants and refineries is included. The data of this study consist of:
- environmental effects of vehicles: energy use and emission factors;
- environmental effects of refineries and electricity plants: energy use and emission factors;
- logistic characteristics of transport modes (like the capacity of vehicles and load factors).

Energy use and emission factors of vehicles are based on vehicle characteristics like size, vehicle capacity and fuel type. For all modes, we give a vehicle-specific energy use. For modes with a combustion engine, we also give emission factors.

For transport modes with electric engines, the vehicles themselves cause no emissions, except copper emissions due to wear of the overhead lines. Obviously, in these cases the contribution of electricity plants and refineries should be taken into account. The corresponding emissions are calculated using the vehicle's energy consumption and the emission factors of electricity plants.

Talking about the energy use of electric modes, it is important to distinguish the *electric* energy use from the *primary* energy use. Electric energy refers to the energy that is provided by the grid. Primary energy refers to the energy that is used to produce and distribute the electrical energy. The primary energy is always much higher than the electric energy, because of the energy return of electricity generation and distribution. For a true comparison of modes one should always use the primary energy. Therefore, all energy values mentioned for electric modes are the *primary* electric energy consumption, i.e. including the energy losses of the electricity production and distribution.

For all modes with combustion engines, the emissions of refining are calculated by using the vehicle's energy consumption and the refinery's emission factors. To obtain a total emission for vehicles with combustion engines, the refinery emissions need to be added to the emissions of the combustion engines.

Energy use and emission factors of electricity plants and refineries
The emissions of electricity production are very different in the various EU countries, because of the different shares of coal, natural gas, hydropower and nuclear power. In this study we use average emission factors for electricity production for the whole of the EU.

CO_2-emission factors for nuclear power are much lower those for power that has been produced from fossil fuels[4]. However, in the opinion of a substantial part of the population, nuclear power is not an alternative, mainly because of the safety and waste aspects. The comparison of transport modes powered by electric engines with modes that are powered by combustion engines can give rather different results when nuclear power is excluded. For this reason we present the EU emission factors for electricity plants both with and without the share of nuclear power (see Annex F). In our comparisons we use the values without nuclear power.

Logistic characteristics of modes
To be able to judge the environmental effects of transport modes, data that characterise the transport modes are needed besides emission factors and energy use. In this study we consider the following logistic characteristics:
- load factors (average load per vehicle for productive rides, expressed in % of total vehicle capacity or in number of passengers or tonnes per vehicle);
- ratio of number of productive rides to non-productive rides (in %);

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[4] The CO_2-emission of a nuclear power plant itself is zero. Therefore, the CO_2-emission of nuclear power is only determined by the energy use of refining, enrichment and by the processing of waste.

- utilisation factors (this is the product of the load factor and the percentage of productive rides);
- detour factors;
- distance of transport to and/or from loading point in the case of intermodal transport.

1.5 Overview of the transport market for ethylene and propylene

Ethylene and propylene are the most important olefins. Ethylene is used for the production of plastics, e.g. polyethylene (58%), PVC (14%), and a number of other products, mostly plastics. Propylene is used for the production of polypropylene (56%) and a number of other products.

Propylene exists in three quality grades, which are, in order of increasing quality:
- refinery grade;
- chemical grade;
- polymer grade.

Propylene is transported by the following modes:
- pipeline;
- barges;
- rail transport;
- sea vessels.

Ethylene exists in just one quality grade: polymer grade. It is transported by only two modes:
- pipelines;
- sea vessels.

Information about the way ethylene and propylene are transported can be found in chapter 2 to 5.

Table 1 gives an overview of the transport volumes of the different modes for transport of ethylene and propylene in the EU above 50 kilometres.

Table 1 Estimate of the current transport volumes per mode for transport of ethylene and propylene above 50 km

	sea ship	***pipeline***	***barge***	***rail***	***total***
Transport volume in 1,000 tonnes/year					
Ethylene	1,450	7,290	0	0	8,660
Propylene	1,450	3,090	400	1,200	6,140
Total	*2,900*	*10,300*	*400*	*1,200*	*14,800*
Average distance (weighted) in km					
Ethylene	750	150	0	0	
Propylene	680	90	150	420	
Transport volume in million tonne-km					
Ethylene (million tonne-km)	1,088	1,094	0	0	2,181
	50%	*50%*	*0%*	*0%*	*100%*
Propylene (million tonne-km)	986	278	60	504	1,828
	54%	*15%*	*3%*	*28%*	*100%*
Total (million tonne-km)	2,074	1,113	60	504	3,751
	55%	*30%*	*2%*	*13%*	*100%*

Source: |APPE|

With these transport volumes, the energy use and emissions of ethylene and propylene transport can be calculated. Table 2 gives an overview of this, using data presented in this study.

We find that the total energy use of ethylene and propylene transport in the EU is about 1.7 PJ. This is about 0.01% of the total energy consumption of transport within the EU or, as a reference, 20% of the total energy consumption of all rail transport in Belgium.

Table 2 Indication of current energy use and emissions of transport of ethylene and propylene above 50 km

	sea ship	***pipeline***	***barge***	***rail***[5]	***total***
Total energy use (TJ)[6]	1,327	192	36	184	1,739
Emissions of:					
CO_2 (1000 tonnes)	107	12	3	13	134
NO_x (tonnes)	2,153	21	42	123	2,339
PM_{10} (tonnes)	54	4	2	6	67
SO_2 (tonnes)	742	56	3	36	837
VOC (tonnes)[7]	341	2	5	16	365

5 The emissions of trains depend on the energy source of the train engine: diesel or electric. We have no data on the shares for this specific type of transport. Therefore, we assume 50% electric and 50% diesel trains.

6 Not including initial compression or cooling.

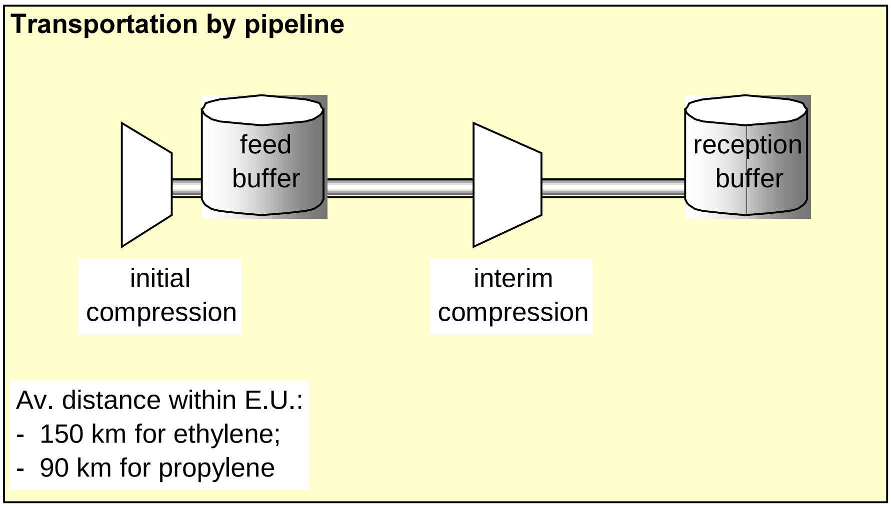
7 Not including VOC-emissions because of *evaporation* during operations and maintenance.

2 Emissions from pipeline transport

Transportation by pipeline concerns transportation between specific suppliers and consumers through a static and dedicated system of pipes at elevated pressure.

Within the European Union, five pipeline systems for long distance transportation of ethylene and/or propylene exist, each measuring several hundreds of kilometres. They link several producers and all major consumers within the region.

Figure 2 Compression steps in a pipeline system



Ethylene is transported in a super-critical state[8] under a pressure of 60 bar or higher. Propylene is transported in a liquid state under a pressure of 20 bar or higher. Sometimes, the commodity is stored in tanks prior to transportation, thus creating a buffer between the cracker and the pipeline with which fluctuations in demand can be met, see Figure 2. However, in most cases, changes in demand are buffered by the pipeline itself.

The ethylene and propylene are sent into the pipeline at overpressure to overcome pressure drops during transportation. A typical working pressure is around 100 bar for both ethylene and propylene (e.g. see |AVIV|). Depending on the specifications of the pipeline system – e.g. materials used, fluid velocity – one or several additional interim compression steps along the stretch of pipeline may be required. At the receiving end, the commodity may once again be stored in tanks, before being utilised.

[8] At a pressure of about 60 bars, gaseous ethylene comes in a super-critical state. In that state, it is still a gas, but with a much higher density than at lower pressures. However, the viscosity does not change and is still as it is below 60 bars. At lower temperatures than ambient, the super-critical phase is already reached at lower pressures than 60 bars. Propylene has no super-critical phase, but becomes liquid at about 8 bars.

The average transportation distance is 150 km for ethylene and 90 km for propylene. Detour factors for Northwest Europe range from almost 0% to approximately 100% (see also Chapter 6).

2.1 Energy consumption

Transportation by pipeline requires electricity. Electricity is consumed primarily by compression prior to transportation and by interim compression along the stretch of pipeline. Boosters for interim compression are electrically driven because of the value of the transported commodity |PLE|[9].

Other energy-consuming processes relate to the use of auxiliary equipment, e.g. valves. Energy consumption for the initial compression is, as is the case for other transport modes, not included in the calculations. Energy consumption for interim compression and auxiliary equipment is worked out in section 2.1.1.

Besides the energy consumption for compression and auxiliary equipment, there will be extra energy consumption that must also be taken into account, namely:
- detour factors;
- transport to and from loading points.

These subjects will be covered in the case studies in chapter 6.

2.1.1 Interim compression and auxiliary equipment

Energy consumption of transportation by pipeline depends on the pressure drop per unit of length along the pipeline. This in turn depends primarily on the velocity of the transported commodity, the surface texture of the pipe, the kinetic viscosity of the transported commodity and the resulting turbulence in it.

To determine energy consumption for this option, three sources of information were regarded:
- industry;
- publicly available sources;
- theory.

Information provided by the industry was compared to information from other sources and to estimates on the basis of generally accepted theory. The three sources are discussed below in separate subsections.

Information from industry
For electricity consumption for interim compression an average figure of 0.046 MJ_e/tonne·km was provided by industry. This is equivalent to 0.12 MJ/tonne·km in terms of primary energy. The calculations can be found in Annex A.

The average figure refers to a newly built pipeline, adapted to the required flow of ethylene or propylene. In existing pipeline systems energy consumption may be higher, e.g. due to the fact that current transportation practice does not quite match design specifications.

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9 This in contrast to transportation of natural gas by pipeline. In that case the transported commodity is used for driving the booster compressors.

Publicly available sources
There is little publicly available information with which to verify the information provided by the industry. Publicly available studies in the Netherlands in which values for energy consumption for pipeline transportation are quoted, all refer to the same study by NEA and Haskoning from 1993, which in turn refers to the study conducted in 1978 by Mittal |Mittal|. In this study a range of 0.11 – 0.18 MJ/tonne·km of primary energy is given. The information provided by EPDC matches this information well.

Theoretical approximate
As a second verification, electricity consumption was also estimated using generally accepted relations for the pressure drop per unit of length along the pipeline and for the work of an adiabatic pump. Applying figures from practice for pipe diameter, flow rate and pressure for ethylene and propylene pipelines gives a energy consumption of 0.050 MJe/tonne·km for ethylene and 0.046 MJe/tonne·km for propylene (see Annex A). These figures refer to a primary energy consumption of 0.13 MJ/tonne·km and 0.12 MJ/tonne·km respectively. Again this matches the data provided by industry very well.

For most modes, the theoretical energy consumption is lower than the actual energy consumption. Assuming that this is also the case for pipelines, a range of 0.11 to 0.18, as found in the literature, seems realistic from a theoretical point of view.

2.1.2 Conclusions

The values for the specific energy use derived from the different sources match very well. In this study, we use the following values for energy consumption, expressed in MJ of primary energy per tonne-km.

	Best Case	Worst Case	Average
Total energy consumption (MJ/tonne-km)	0.11	0.18	0.14

2.2 Emissions

We consider the following types of emissions:
- emissions produced by burning fuel for electricity generation (CO_2, NO_x, SO_2, VOC, and PM_{10});
- emissions of ethylene/propylene from leakage or other evaporation, during operations and maintenance (VOC only);
- emissions arising during loading or unloading of cargo and transport to and from loading points.

The following sections will cover these subjects, except the last category, which will be considered in the case studies in chapter 6.

2.2.1 Emissions produced by burning fuel for electricity generation

The most important emissions to take into account for transportation by pipeline are the - indirect - emissions related to electricity production.

In Table 3 the calculation of the magnitude of these emissions is given, for a situation typical for the EU, for ethylene and propylene, respectively. Transportation distances are 150 kilometres for ethylene and 90 kilometres for propylene. Emissions are calculated by multiplying the specific emission factors (second row from top) with the total consumption of primary energy (second column from left).

Table 3 Indirect emissions for average situation within E.U. for ethylene and propylene transportation by pipeline

	Primary energy consumption (MJ/tonne-km)	Emissions (g/tonne-km)				
		CO_2	NO_x	PM_{10}	SO_2	VOC
G/MJ primary energy*		60.0	0.11	0.019	0.291	0.011
best case	0.11	6.6	0.012	0.002	0.032	0.001
worst case	0.18	10.8	0.020	0.003	0.052	0.002
average case	0.14	8.4	0.015	0.003	0.041	0.002

* Source: CE/RIVM, 2003 p 80. The figures take into account 42% energy efficiency in electricity production and 90% in distribution, so totally 38%. The values are based on EU as a whole and exclude nuclear power since it is produces emissions and problems that are not easily comparable to those of conventional power generation.

2.2.2 Emissions of ethylene / propylene during operations and maintenance

It is possible that emissions of ethylene and propylene from pipelines occur due to sporadic maintenance activities. However there are no clear patterns with which frequency maintenance requiring evacuation of the pipeline occurs. When initiating the pipeline some flaring occurs. In the case of a 153 km pipeline from BP, 16 tonnes of ethylene was flared. This, however, constitutes a very small loss over the lifetime of the pipeline (some 60 years), and flaring makes sure that almost none of the ethylene/propylene is released to the atmosphere.

It is likely that there will be some evaporation during the normal use of the pipeline. For a natural gas pipeline in EU, gas leakage during normal operation is about 0.008%/100 km (|ExternE|), and although a different technology is used for propylene/ethylene pipelines, this leakage may be of the same order of magnitude. If equal, we can expect an extra 0.8 g VOC/tonne-km. For underground pipelines, much of such an amount may be decomposed to CO_2 before reaching the atmosphere. It would be equivalent to 2.3 g CO_2 per tonne-km, still a significant increase.

3 Short sea shipping

3.1 Description

Short-sea shipping is transport by ship over relatively short distances – from country to country rather than from one continent to another. Transportation of ethylene and propylene by ship is an important mode. Part of the ethylene and propylene consumed within the European Union is imported by ship from – mainly – Arabia, because of limited production capacity within the EU.

Transportation of ethylene and propylene may take place in vessels of any size. According to |Shell|, ethylene is transported from Saudi Arabia in large vessels because of the very low production costs of ethylene in Saudi Arabia, where it emerges almost spontaneously from oil fields. Transport by sea within the EU occurs in short-sea vessels with capacities of about 2,000-4,000 tonnes |EPDC, Shell, DOW|. Tonnage for short-sea vessels is only limited because producers are not situated at deep-sea ports |Shell|.

Ethylene and propylene is transported by vessels in supercritical state or liquid state, respectively. The commodity is transported either:
- under pressures of about 6 bar and cooled to -47°C in the case of propylene or –104°C in the case of ethylene or,
- compressed at ambient temperature to a minimum pressure of 15 - 20 bar, for propylene only (|DOW|).

Ethylene is not transported by ship in fully compressed form.

The commodity is either stored in tanks prior to and after transportation or pumped out of / into a pipeline system.

Transportation takes place both in dedicated vessels and in commercial and independent chemical tankers. Most of the vessels that can carry ethylene can also carry a number of other substances, including propylene.

3.2 Energy consumption

The energy consumption of transportation depends on:
- the energy consumption for propulsion of the ship;
- the utilisation factor (accounting for actual load carried and unproductive rides);
- energy consumption for keeping the cargo cooled/under pressure;
- detour factors;
- transport to and from loading points.

The following sections will cover these subjects, except the detour factors and transport to and from loading points, which will be worked out and applied in the case studies in chapter 6.

3.2.1 Energy consumption of the ship

We have obtained data for 20 different ships from two shipping companies. This provides a range of values for energy consumption to compare to those of other modes. In cargo capacity, these ships are approximately in the size range of normal short-sea ethylene and propylene carrying ships, according to information from EPDC. We have calculated their energy consumption and selected the extremes in the table below. Data for all ships can be found in Annex C. Two types of tankers (class OC1 and OC2) that are used in the CE/RIVM study are included for comparison |CE/RIVM, 2003|. These two have engine powers comparable to the ethylene/propylene ships, but their capacities are either lower or higher. As expected, the results for the ethylene/propylene ships fall nicely in between.

Table 4 Ship data used

	Coral Pavona	Prins J W Friso	Gas Trust	Coral Meandra	OC1	OC2
Ship type	LPG/ethylene carrier	LPG/ethylene carrier	Fully pressurised LPG carrier	Semi-pressurised/ fully refrigerated gas carrier	Tanker	Tanker
Cargo type	Ethylene /propylene	Ethylene /propylene	Propylene	Propylene	Other liquid	Other liquid
Built year	1998	1988	1996	1996	N/A	N/A
Main engine power (kW)	4,550	3,960	?	2,960	4,000	6,000
Capacity (tonnes)	3,994 (eth)	2,353 (eth)	1,835 (prop)	2,571 (prop)	1,022	8,800
Fuel consumption of main engine (tonnes/day)	17.0	14.5	11.3	11.8	N/A	N/A
Average service speed	24 km/h	25 km/h	25 km/h	25 km/h	24 km/h	24 km/h
Calculated energy consumption (MJ/tonnes-km)	0.31	0.44	0.44	0.33	1.06	0.18

Sources: OC1/2 tankers extracted from |CE/RIVM, 2003|.
Other data from http://www.anthonyveder.nl and www.exmar.be.

The ethylene carriers are also capable of transporting propylene. Due to the higher density of propylene, about 8% (by weight) more propylene than ethylene can be transported in the same volume. This is partly offset by a small increase in energy consumption. For simplicity, we shall assume identical energy consumption per tonne-km for transport of propylene and ethylene.

For short-sea transportation of ethylene and propylene we thus have a best case of 0.31 MJ/tonne-km and a worst case of 0.44 MJ/tonne-km. The average (over all ships) is 0.36 MJ/tonne-km.

3.2.2 Utilisation factor

Energy consumption also depends on the actual load carried by the ship. It appears that ships are generally full on the outbound journey, but it is often the case that the ships return empty, in which case we must include part or all of energy consumption during the return trip in the total energy consumption of the transport of the ethylene/propylene.

We apply a range based on the following scenarios:

Best Case : Full on the outward trip, 40% full on return
Worst Case : 80% full on outward trip, empty on return

The best case corresponds to a utilisation factor of 0.70, while the worst case corresponds to 0.40. The average is 0.55.

When a ship returns empty it will take in a considerable load of ballast. Moreover, the ship will move at greater speed which also eats away most of the gain from the reduced load (|Unigas|). We assume the energy consumption per km for the "empty" ship to be 90% of that of the fully loaded ship (see annex D for a fuel-consumption curve).

Also for reduced loads, we will assume that the ships take in more ballast and increase speed a bit. In case of reduced loads, the fuel consumption is between 90% and 100% of normal, but the fuel consumption per tonne-km will nearly double in the case of a half-loaded ship. We calculate energy consumption for reduced loads by linear interpolation.

3.2.3 Energy consumption of cooling / maintaining pressure of the cargo

A powerful cooling system is necessary to keep the ethylene/propylene in a liquid state. Substantial amounts of energy are required for this, depending on the requirements of the type of cargo. For a 2,700-tonne propylene carrier this would add an extra 8% to the fuel consumption (|Lauritzen Kosan|). In the case of ethylene, which needs more cooling, it may be about 50% more than for propylene (|Unigas|), corresponding to 12% extra energy consumption. As the difference between the two is rather small compared to the total energy needs, we select 10% as an average. As no cooling is needed when no cargo is transported (|Unigas|), the energy for this purpose decreases with decreasing utilisation factor.

Fully pressurising the cargo is another option. This will probably, as is the case for trains, cost no extra energy to maintain. However, for the large volumes transported by sea it is most economical to cool it rather than pressurise it so therefore we choose to disregard pressuring.

3.2.4 Conclusion on energy consumption by sea vessels

As each of the factors considered are influenced only to a minor degree by whether the cargo is ethylene or propylene, we shall assume these to be equal. In the table below we summarise our findings.

Table 5 Summary of energy consumption of transport of ethylene/propylene by short sea vessels

	Best Case	Worst Case	Average
Vehicle energy consumption (MJ/tonne-km)	0.31	0.44	0.36
Utilisation factor (in %)	70%	40%	56%
Energy consumption of empty vessel	90%	90%	90%
Factor for maintaining cooling/pressure	10%	10%	10%
Total energy consumption (MJ/tonne-km)	**0.45**	**1.05**	**0.64**

3.3 Emissions

We consider the following types of emissions:
- emissions produced by burning fuel for transportation (CO_2, NO_x, SO_2, VOC, and PM_{10});
- emissions produced in the refining of this fuel (CO_2, NO_x, SO_2, VOC, and PM_{10});
- emissions of ethylene/propylene from leaking or other evaporation, during operations and maintenance (VOC only);
- emissions arising during loading/unloading cargo, and transport to and from loading points.

The following sections will cover these subjects, except the last category, which will be considered in the case studies in chapter 6.

3.3.1 Emissions produced by burning fuel for transportation

In calculating vehicle emissions, we use standard emission factors based on the energy consumption of the vehicles.

Table 6 Emission factors for sea vessels (fleet average 2010)

	CO_2	NO_x	PM_{10}	SO_2	***VOC***
g/MJ	75	1.59	0.04	0.53	0.18

Source: CE/RIVM, 2003 p 79

3.3.2 Emissions produced in the refining of fuel

In calculating emissions from refining, we use these emission factors based on type of fuel used.

Table 7 Emission factors from refining of fuel oil

	CO_2	NO_x	PM_{10}	SO_2	***VOC***
g/MJ fuel	5.4	0.032	0.001	0.029	0.077

Source: CE/RIVM, 2003 p. 80

3.3.3 Emissions of ethylene / propylene during operations and maintenance

It is to be expected that there will be some loss of ethylene/propylene during storage, transfer and/or transportation, mostly due to evaporation. Emissions during loading and unloading have been given by Shell. According to our source emissions amount to only 1 kg of ethylene or propylene in case the vessel is meant for dedicated shipping of ethylene or propylene or cargoes are compatible. In this case emission results from ‘washing’ of loading equipment with inert gas. The emission is independent of ship size.

In case the vessel has to be degassed and a vapour recovery system is available there will be also emissions of CO_2, NO_x and other pollutants due to flaring or utilisation of the vapour as a fuel. However, it is often the case that a vapour recovery system is not available (|Lauritzen Kosan|); in case the vessel has to be degassed and vapour is not recovered, the emission of ethylene or propylene will amount to approximately 1,000 kg per 1,000 m^3 cargo space.

For a ship with 3,000 tonnes ethylene going the average 715 km (|APPE|), this evaporation will in the best case (1kg/trip) lead to a VOC-emission of less than 0.001 g/tonne-km. In the worst case it will be as much as 2.9 g/tonne-km. This could certainly pose some health and safety risks. Ethylene and propylene can also contribute to the formation of tropospheric ozone, an important factor in climate change. If ethylene and propylene have a significant global warming potential the worst case could have considerable impact on the "climate score" of the transportation. While we consider it important that the magnitude of this effect be investigated, it falls out of the scope of this study.

3.3.4 Conclusion on emissions

The total emission factors are summarised in Table 8 below. Contribution from propylene/ethylene emissions will be added later.

Table 8 Overview of emission sources by type and magnitude

Emission source	CO_2	NO_x	PM_{10}	SO_2	**VOC**
Vessel g/MJ	75	1.59	0.04	0.53	0.18
Refining g/MJ	5.4	0.032	0.001	0.029	0.077
Total emission factor g/MJ	**80.4**	**1.62**	**0.04**	**0.56**	**0.26**
Leaking/evaporation g/tonne-km	-	-	-	-	0.001-2.9*

For propylene and ethylene transport by short sea shipping, we get the following emission values expressed in gram of the particular pollutant per tonne-km:

Emissions in g/tonne-km	CO_2	NO_x	PM_{10}	SO_2	**VOC**
Best case	36.3	0.7	0.02	0.3	0.12
Worst case	84.5	1.7	0.04	0.6	0.3*
Probable average	**51.8**	**1.0**	**0.03**	**0.4**	**0.17**

*For VOC-emissions, we have not included the worst case scenario for leaking of ethylene and propylene since a proper estimate is not available for all modes.

4 Inland shipping

4.1 Description

By inland shipping we understand transport by boat via inland waterways such as rivers. This transportation occurs by barges. Producers in the Rotterdam-Antwerp-Cologne triangle supply their customers by barges of up to 1,500 tonnes loading capacity |Shell|, |DOW|, |TUDelft|. Barges do not transport ethylene.

Transportation distances by barge in Northwest Europe range from 100 – 300 kilometres, with an average of 150km (|APPE|).

The propylene is, when transported by barge, not cooled (|Chemgas|) but compressed at minimum 15-20 bar (|DOW|). Barge transport of propylene takes place mostly by dedicated barges that return empty (|Chemgas|).

4.2 Energy consumption

The energy consumption of transportation depends on:
– the energy consumption for transporting the barge;
– the utilisation factor (accounting for actual load carried and unproductive rides);
– energy consumption for keeping the cargo cooled/under pressure;
– detour factors;
– transport to and from loading points.

The following sections will cover these subjects, except detour factors and transport to and from loading points, which will only be applied in the case studies in chapter 6.

4.2.1 Energy consumption of the barge

Considering that propylene barges have capacities up to 1,500 tonnes, we will base our calculation on a barge of such dimension from a previous CE report |CE/RIVM, 2003|. From that source we use the following information.

Table 9 Barge information overview

Barge type	1,000-1,500 tonnes capacity barge
Energy consumption of average loaded barge (45% utilisation)	376 MJ/km
Average capacity	1,250 tonnes
Main engine power (kW)	580
Average service speed	14 km/h

Sources: |CE/RIVM, 2003|

At this stage, we do not calculate the energy consumption of a fully loaded barge, but assume that the energy consumption of an “average loaded barge”, in MJ/vehicle-km as given above, is the same for the range of utilisation factors we use in this study.

4.2.2 Utilisation factor

Energy consumption also depends on the actual load carried by the barge. It appears that, just like for inland shipping, barges are generally full on the outbound journey, but it is often the case that the barges return empty. In that case we must include part or all of energy consumption on the return trip in the total energy consumption of transport of the ethylene/propylene.

We apply a range based on the following scenarios:

Best case	: Full on the outward trip, 20% full on return
Worst case	: 80% full on outward trip, empty on return

The best case corresponds to a utilisation factor of 60%, while the worst case corresponds to 40%. The average is 50%.

4.2.3 Energy consumption used for maintaining cooling / pressure of the cargo

We assume, as in the case for rail transport, that no energy is expended to maintain pressure.

4.2.4 Conclusion on energy consumption by barges

The energy consumption of transport of propylene by barge will be, for the different cases:

Best case	: 0.50 MJ/tonne-km
Worst case	: 0.75 MJ/tonne-km
Average case	: 0.60 MJ/tonne-km

4.3 Emissions

We consider same types of emissions as for short sea shipping:
- emissions produced by burning fuel for transportation (CO_2, NO_x, SO_2, VOC, and PM_{10});
- emissions produced in the refining of this fuel (CO_2, NO_x, SO_2, VOC, and PM_{10});
- emissions of ethylene/propylene from leaking or other evaporation, during operations and maintenance (VOC only);
- emissions arising during loading/unloading cargo, and transport to and from loading points.

The following sections will cover these subjects, except the last category, which will be considered in the case studies in chapter 6.

4.3.1 Emissions produced by burning fuel for transportation

In calculating vehicle emissions, we use standard emission factors based on the energy consumption of the vehicles.

Table 10 Emission factors for barges (fleet average 2010)

	CO_2	NO_x	PM_{10}	SO_2	VOC
g/MJ	73.3	1.14	0.06	0.05	0.07

Source: CE/RIVM, 2003 p 78

4.3.2 Emissions produced in the refining of fuel

In calculating emissions from refining, we use these emission factors based on type of fuel used.

Table 11 Emission factors from refining of fuel oil

	CO_2	NO_x	PM_{10}	SO_2	VOC
g/MJ fuel	5.4	0.032	0.001	0.029	0.077

Source: CE/RIVM, 2003 p 80

4.3.3 Emissions of ethylene / propylene during operations and maintenance

As barge transport of propylene is done by dedicated barges, there are no large losses of propylene, as can be the case for sea ships. The latter practise is furthermore forbidden, and another source confirms that it does not occur (|Gaschem)|. We will assume a loss of 1 kg of propylene per trip. For a 1,500 tonne barge with normal load travelling the average 150 km, the propylene emissions will amount to 0.005 g/tonne-km.

4.3.4 Conclusion on emission

The total emission factors expressed in *gram per MJ fuel* are summarised in Table 12 below.

Table 12 Overview of emission sources by type and magnitude

Emission source	CO_2	NO_x	PM_{10}	SO_2	VOC
Barge g/MJ	73.3	1.14	0.06	0.05	0.07
Refining g/MJ	5.4	0.032	0.001	0.029	0.077
Total emission factor g/MJ	78.7	1.172	0.061	0.079	0.147
Leaking/evaporation g/tonne-km	-	-	-	-	Max 0.005

Expressed in gram of the particular pollutant per tonne-km, we get the following emissions:

Emissions in g/tonne-km	CO_2	NO_x	PM_{10}	SO_2	**VOC***
Best case emissions	39.5	0.59	0.03	0.04	0.07
Worst case emissions	59.2	0.88	0.05	0.06	0.11
Average emissions (g/tonne-km)	47.3	0.71	0.04	0.05	0.09

* Note that emissions of propylene from leaking/evaporation are not included here, as good estimates are not available for all modes.

5 Rail transport

5.1 Description

Transportation of ethylene by rail is very costly, certainly compared to pipeline transportation (|Railion|) and does therefore not occur in the EU.

Propylene is transported by rail at ambient temperature in a liquid state to a pressure of about 20 bar (|DOW|, |Railion|). Although it is technically possible to transport propylene cooled, in practise it is not, possibly because cooling is more costly for the relatively small loads.

Given the nature of the commodities transported, specialised vehicles are required. The tank wagons used are only capable of transporting a limited range of C3 and C4 hydrocarbons. This makes it difficult to replace the ethylene/propylene cargo with something else for the return trip, and since producers and consumers are also far apart, the vehicles generally return empty (|Railion|). The cargo capacity of a typical propylene tank wagon is 39 tonnes |Railion|, |TUDelft|).

Distance covered can be up to 200 kilometres in the Rotterdam-Antwerp-Cologne triangle. In EU as a whole the average transport distance by rail is 420 km |APPE|. Total amounts transported in the triangle region seem to be rather negligible (see |TUDelft|), indicating that the bulk of transportation by rail may take place in other regions within the European Union, presumably regions with not such excellent waterways as the Rotterdam-Antwerp-Cologne triangle. Transportation by rail in the Rotterdam-Antwerp-Cologne region may refer to transportation of a limited amount of propylene to one specific consumer.

5.2 Energy consumption

The energy consumption of transportation depends on:
– the energy consumption of transporting the train;
– the utilisation factor (accounting for actual load carried and unproductive rides);
– energy consumption for keeping the cargo under pressure;
– detour factors;
– transport to and from loading points.

The following sections will cover these subjects, except detour factors and transport to and from loading points, which will only be applied in the case studies in chapter 6.

5.2.1 Energy consumption of the train

For the calculation of the vehicle energy consumption we assume that a train of 24 propylene cars of 39 tonnes propylene each (estimated by |Railion|). The total load capacity of such a train is 936 tonnes. We assume that the energy use is comparable to a standard bulk freight train, as presented in |CE, 2000|.

Table 13 Average energy consumption of bulk freight trains

Type	vehicle energy consumption	with load	energy consumption in MJ/tkm
Electric train	177 MJ/trainkm	400 tonnes	0.44
	296 MJ/trainkm	1,000 tonnes	0.30
Diesel train	188 MJ/trainkm	400 tonnes	0.47
	314 MJ/trainkm	1,000 tonnes	0.31

Source: *Milieuwinst op het spoor?* (|CE, 2000|). Energy for electric modes is expressed in MJ of *primary* energy (see section 1.4)

5.2.2 Utilisation

According to Railion propylene tank wagons are mostly full on the outward journey, but nearly always return empty. As this agrees nicely with the findings for general bulk rail transport in |CE/RIVM, 2003|, we will use the assumptions in the latter of a load factor of 80% and 51% productive rides (nearly always empty return). So, we calculate with an average utilisation of 41%, which corresponds with 384 tonnes per train.

The vehicle energy consumption can be obtained from linear extrapolation of the data in Table 13:

Electric: 174 MJ/train-km => 0.45 MJ/tkm
Diesel: 185 MJ/train-km => 0.48 MJ/tkm

5.2.3 Energy consumption for maintaining cooling / pressure of cargo

No energy is required for keeping the cargo under pressure (|Railion|).

5.2.4 Conclusion on energy consumption of trains

The energy consumption of transporting propylene by rail is, accounting for reduced utilisation and (no) energy requirements for maintaining pressure:

Electrical trains : 0.45 MJ/tonne-km
Diesel trains : 0.48 MJ/tonne-km

To account for variation in utilisation and technical performance of trains, we apply a range of error of 20%.

5.3 Emissions

We consider following types of emissions:
- emissions produced by burning fuel for transportation (CO_2, NO_x, SO_2, VOC, and PM_{10});
- emissions produced in the refining of fuel (CO_2, NO_x, SO_2, VOC, and PM_{10});
- emissions produced by burning fuel for electricity generation (CO_2, NO_x, SO_2, VOC, and PM_{10});
- emissions of propylene from leaking or other evaporation. (VOC only);
- emissions arising during loading/unloading cargo, and transport to and from loading points.

The following sections will cover these subjects, except the last category, which will be dealt with in the case studies in chapter 6.

5.3.1 Emissions produced by burning fuel for transportation

In calculating vehicle emissions, we use standard emission factors based on the energy consumption of the vehicles. Electric trains have no vehicle emissions.

Table 14 Emission factors for diesel freight trains (fleet average 2010)

	CO_2	NO_x	PM_{10}	SO_2	VOC
g/MJ electricity	73.3	1.19	0.05	0.05	0.08

Source: CE/RIVM, 2003 p 76

5.3.2 Emissions produced by burning fuel for electricity generation

For electric trains we apply the emission factors associated with the electricity production. The factors are applied to the *primary* energy consumption, not the electric energy consumption (see section 1.4).

Table 15 Emission factors for electrical freight trains

	CO_2	NO_x	PM_{10}	SO_2	VOC
g/MJ primary	60.0	0.11	0.019	0.291	0.011

Source: CE/RIVM, 2003 p 80. The figures take into account 42% energy efficiency in electricity production and 90% in distribution. The values are based on EU 2010) as a whole and exclude nuclear power since including nuclear power would make it impossible to make a fair comparison between electric modes and non-electric modes.

5.3.3 Emissions produced in the refining of fuel

In calculating emissions from refining, we use these emission factors based on diesel.

Table 16 Emission factors from refining of diesel

	CO_2	NO_x	PM_{10}	SO_2	VOC
g/MJ fuel	6.8	0.036	0.001	0.052	0.088

Source: CE/RIVM, 2003 p 80

For electric modes, the refining emissions are zero |CE/RIVM, 2003|.

5.3.4 Emissions of ethylene / propylene during operations and maintenance

Besides the emissions arising from fuel consumption, also emissions of VOC can be expected because of evaporation. The wagons used for propylene can and do occasionally transport different commodities. Before using the wagon for a different commodity, it is “washed” with inert gas before the change and the vapour is recovered with a VRS (Vapour Recovery System), which are equipped on all propylene wagons (|Railion|). The recovered vapour is either returned to storage or flared. In case it is flared it will produce some emissions of CO_2, NO_x, and VOC. We presume these amounts are negligible. For ships (see section 3.3.3) the propylene emission would be 1 kg/ship in the case comparable to the situation for trains. It is likely that it will be lower than that for trains, considering the much smaller scale.

For propylene being transported by train (with an average load) an average of 420 km (|APPE|), this evaporation will in the unlikely worst case (1kg/trip) lead to a VOC-emission of 0.08 g/tonne-km. See discussion of this contribution in Section 3.3.3.

5.3.5 Conclusion on emissions

The total emission factors are summarised in Table 18 and Table 17 below. Contribution from propylene emissions is not included.

Table 17 Overview of emission factors for electric trains g/MJ

Emission factor in g/MJ	CO_2	NO_x	PM_{10}	SO_2	VOC
Train	0	0	0	0	0
Electricity Production	60.0	0.11	0.019	0.291	0.011
Refining	0	0	0	0	0
Total emission factor	**60.0**	**0.11**	**0.019**	**0.291**	**0.011**

Table 18 Overview of emission factors for diesel trains g/MJ

Emission factor in g/MJ	CO_2	NO_x	PM_{10}	SO_2	VOC
Train	73.3	1.19	0.05	0.05	0.08
Electricity Production	0	0	0	0	0
Refining	6.8	0.036	0.001	0.052	0.088
Total emission factor	**80.1**	**1.23**	**0.05**	**0.12**	**0.17**

For propylene transport by train, we get the following emission values expressed in gram of the particular pollutant *per tonne-km* by multiplying the emission factors with the values for energy consumption (see 5.2.4).

Table 19 Total emissions from train transport of propylene (in g/tonne-km)

Emissions in g/tonne-km	CO_2	NO_x	PM_{10}	SO_2	VOC*
Electric trains (average)	27.2	0.05	0.009	0.13	0.005
Diesel trains (average)	38.6	0.59	0.025	0.05	0.081

* Note that emissions of propylene from leaking/evaporation are not included here, as good estimates are not available for all modes.

6 Comparison of modes

6.1 Basis of comparison

For a proper comparison of transportation options, we considered three specific cases for transportation of ethylene and propylene in Northwest Europe. Considering a specific case has an advantage over a theoretical approach in that it can demonstrate the actual influence of different factors determining the environmental impact related to transportation in 'real life'.

In this chapter we present three representative cases:
- from Antwerp to Cologne area;
- from UK to Antwerp;
- from UK to northern Germany (Stade).

The three cases considered give an indication of the relative environmental impact arising from the different transportation options in Northwest Europe.

6.2 Detour factors and transport to and from loading points

The differences between the cases are due to:
- the transport modes that compete on the transport relations;
- detour factors.

For each case we compare the energy use and emissions of the *transport modes* that compete on the specific transport relations.

The *detour factors* are deduced from the transport distances for the different modes. For each case, the detour factor for the mode with the shortest transport distance (the 'reference mode') is set to 0%. For the competing modes, the detour factors represent the percentage that the travel distances are longer than for the 'reference mode'.

Comparing the environmental impact of different transport options, one should consider not only the main transport route but also the *transport to and from loading points*. In the cases considered, there is almost no transport to and from loading points. Therefore, we assume that all transport modes cover the entire distance from cracker to polymerisation plant.

The energy consumption, emission factors and load factors that we use in the case studies are all based on the information presented in Chapter 2 to 5.

6.3 Impact of initial compression and cooling

6.3.1 Overview of characteristics by mode

As presented in Chapters 2 to 5, ethylene and propylene can be transported either cooled (sea vessels) or under pressure (inland modes).

When the commodity is *cooled* it is cooled to below the boiling point, when it becomes liquid: for ethylene below -104°C and for propylene below -47°C.

When transported *under pressure* the commodity is usually compressed above a critical pressure at which it becomes liquid (propylene) or super-critical and behaves like a liquid (ethylene). For ethylene and propylene theses critical pressures amount to about 50 and 15 bar, respectively.

Table 20 gives an overview of the pressure and temperature of the commodities per transport mode.

Table 20 Overview of temperature and pressure of ethylene and propylene per transport mode

	Ethylene	Propylene
Pipeline	60 - 100 bar	20-100 bar
Sea vessel	< -104° C and a few bar	< -47° C and a few bar; or 15-20 bar
Barge	not transported	15-20 bar
Rail	not transported	15-20 bar

In most cases, the commodities come under high pressure out of the cracker, which makes that there is no extra compression required, when it transported under high pressure. However, in other cases extra compression or cooling is needed.

According to our calculations, full cooling of ethylene or propylene requires primary energy in the order of 300 MJ per tonne. Compression of ethylene from 1 to 55 bar requires primary energy in the order of 1,000 MJ per tonne, while the compression of propylene from 1 to 15 bar requires primary energy in the order of 400 MJ per tonne. Additional compression to 100 bar requires some extra dozens of MJ primary energy per tonne. According to field experts, these energy requirements may be subject to very large variation.

When re-calculating these figures into figures per tonne-km, and assuming a transport distance of 500 km, it appears that initial compression and cooling requires primary energy in the order of 0.6 to 2 MJ/tonne-km. This is more than the energy required for the transport alone.

The conclusion is that the energy use of the initial compressing or cooling of the commodities can be very high compared to the energy use of the transport itself. Particularly compression from ambient pressure to the critical pressure is very energy consuming, probably more than cooling down to the boiling point.

6.3.2 Consequences for inter-modal environmental comparison

The findings in the previous section have some serious consequences for the scope of the environmental comparison of ethylene and propylene transport by mode.

First, there is a difference between inland modes on the one hand and sea vessels on the other hand. Ethylene or propylene transported by sea vessels, is often cooled (not compressed to the critical pressure). The energy needed for cooling can be very different from the energy use needed for compression. When comparing transport of cooled commodities (by sea vessels) with compressed commodities (all other modes of transport), it is important to know what *extra* compression, cooling or heating steps are needed in the whole chain from the beginning of the transport chain (cracker or storage) to the end of the transport chain (storage or user process, e.g. polymerisation plant). However, the pressure and temperature of the ethylene or propylene when it comes out of the cracker or when it goes into polymerisation varies a lot. Therefore it is impossible give general numbers for the initial cooling or compression.

In many cases, commodities transported by sea vessels under low temperature are compressed to the critical pressure after the sea transport, e.g. to store it or to put into a pipeline. In those cases, the commodity needs to be compressed anyway, though not before but after the sea transport. The cooling and heating that is needed for shipping is in that case an extra (energy consuming) process step, resulting in a much *higher* energy use for transport by sea shipping.

It is also possible that ethylene or propylene may occur in a transport chain where no compression is necessary – e.g. if used in processes that do not require (high) pressure and provided the ethylene or propylene is also produced at the cracker at low pressure. In such a case, the energy requirements for transport in a cooled state may be *lower* than for compressed transport.

The chemical processes where propylene and ethylene are used generally take place at high pressure. Therefore, disregarding energy required for initial cooling and compression is likely to lead to a serious underestimation of the total environmental impact of 'cooled' modes, notably sea vessels.

Second, in propylene transport there are differences between the inland modes with respect to the pressure at which products are delivered to the client. The pressure at which propylene is delivered by pipeline is regularly higher than that of other modes. If the propylene will be used after transport at a pressure higher than used in the pipeline, the energy spent on increasing the pressure of pipeline propylene will have to be spent for the other modes as well. Only under this proviso we may disregard energy required for initial compression when comparing pipeline transportation with transportation by barge and rail.

In the case studies where sea vessels are a competing mode, we present *only the impacts of transportation itself, and not the impacts of initial compression or cooling,* as we have not been able to determine the latter costs. We stress that these eventually *must* be considered for a full understanding of the transport costs associated with the different modes.

6.4 Case 1: From Antwerp to Cologne area

6.4.1 Competing transport modes

In the first case considered we compare the environmental impact related to transport of propylene from Antwerp (Belgium) to Wesseling near Cologne close to the Ruhr area of Germany.

Currently, 100,000 tonnes of propylene and ethylene are transported between these two destinations, mostly by barges. We will also consider transport by trains in this case study, distinguishing between diesel and electric trains.

6.4.2 Detour factor

Transportation distances for the different modes are given in Table 21. Based on these, we calculate the detour factors relative to the shortest transportation distance (train). Pipeline distances were provided by APPE. We estimated other distances with the help of MS Encarta. Results are shown also in Table 21.

Table 21 Detour factors for Case study 1

	Barge	Pipeline	Train
Transportation distance	210	205	210
Detour factor	2%	0%	2%

Thus, pipeline transport is the reference mode (0% detour factor) and the detour factor for the other modes is approximately 2%.

6.4.3 Energy consumption and emissions

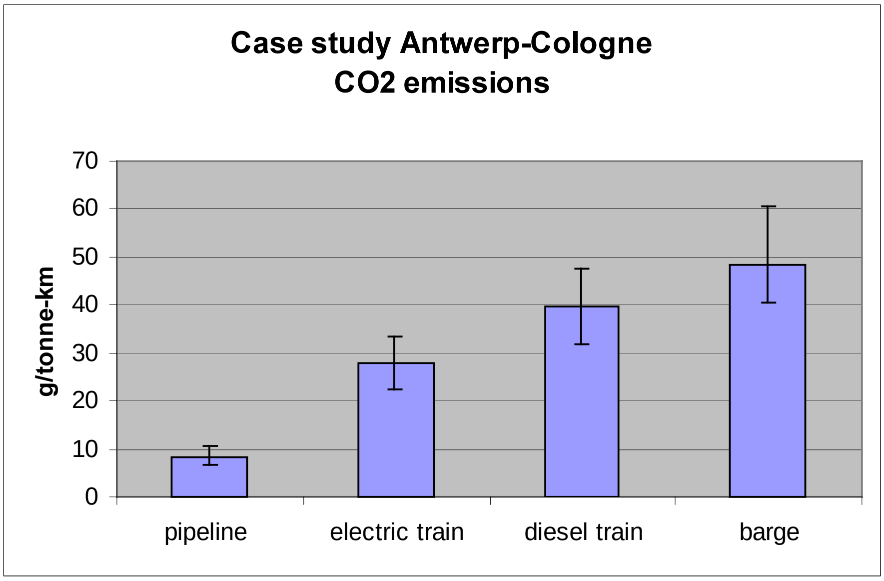
The results for energy consumption are given in Table 22. Utilisation factors are assumed to be the same as we determined earlier.

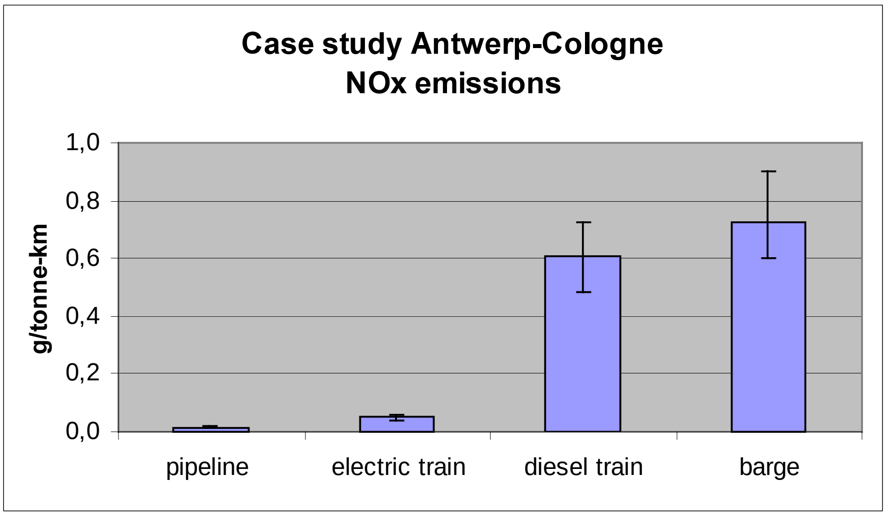
Table 22 Energy consumption of competing options for propylene transportation between Antwerp and Wesseling *under the proviso that the propylene is not de-compressed - i.e. used in a high-pressure process - after transport*

	Basic Energy Consumption MJ/tonne-km	Detour factor	Case study Energy Consumption MJ/tonne.km
Pipeline		0%	
best	0.11		0.11
worst	0.18		0.18
average	0.14		0.14
Electric train		2%	
best	0,36		0,37
worst	0,54		0,56
average	0,45		0,47
Diesel train		2%	
best	0,39		0,40
worst	0,58		0,59
average	0,48		0,49
Barge		2%	
best	0.50		0.51
worst	0.75		0.77
average	0.60		0.62

The resulting emissions are shown in the following figures. We have not added evaporative emissions of propylene or ethylene to the VOC figures, as good estimates are not available for all of the modes. When considered, this may substantially change the picture for VOC. Please see Sections 2.2.2, 3.3.3, 4.3.3, 5.3.4, and 7.3 for discussion of this contribution. Please also note that a fair comparison of the modes below require that the propylene will be used in a high-pressure process. See Section 6.3.

Figure 3 Comparison of CO_2, NO_X and PM_{10}-emissions from transport of propylene by pipeline, train or barge, *under the proviso that the propylene is not de-compressed - i.e. used in a high-pressure process - after transport*





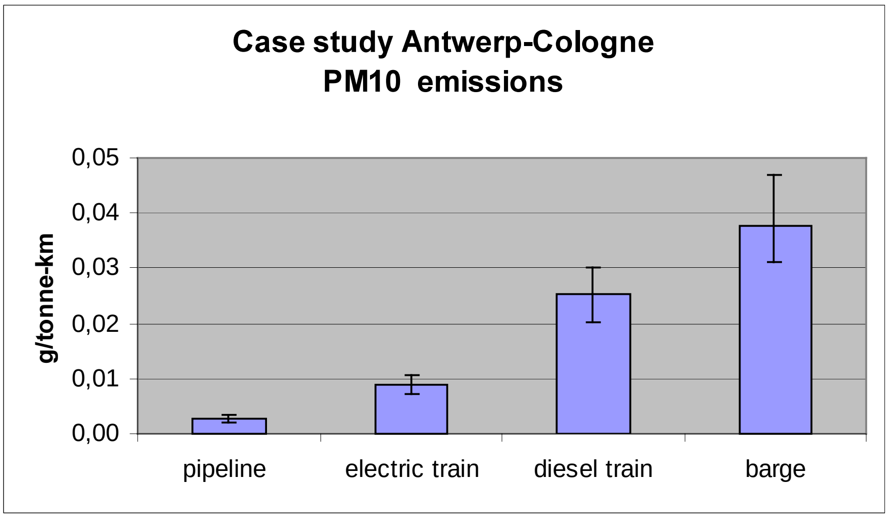
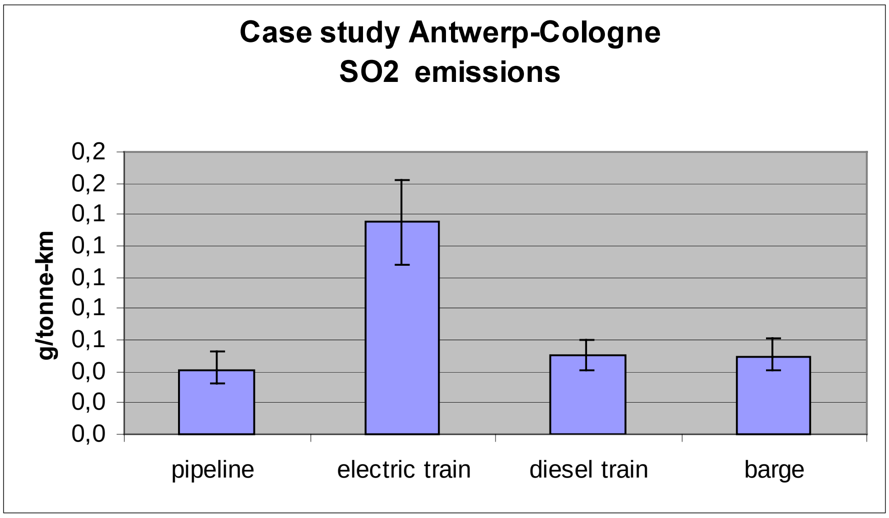
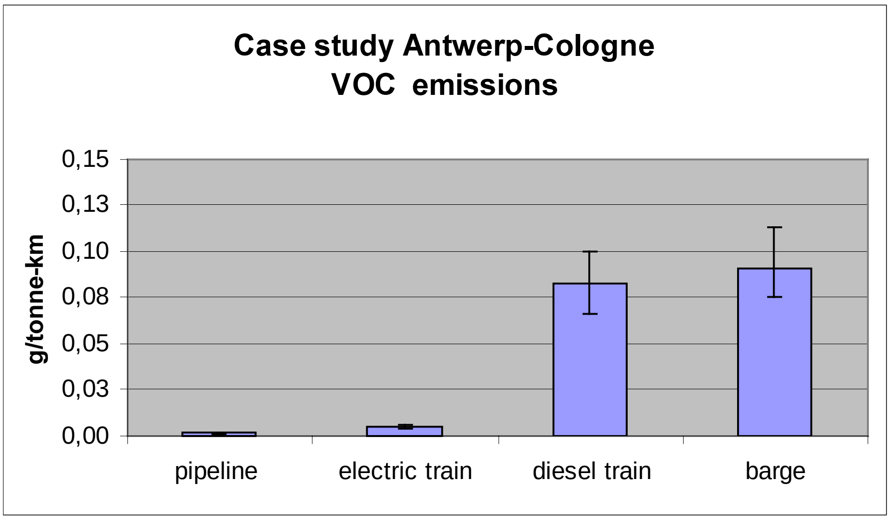


Figure 4 Overview of SO_2 and VOC-emissions from transport of propylene by pipeline, train or barge, excluding evaporative VOC-emissions, *under the proviso that the propylene is not de-compressed - i.e. used in a high-pressure process - after transport*





6.5 Case 2: From UK to Antwerp, the Inter-connector

6.5.1 Competing transport modes

In the second case considered, we compare the environmental impact related to export of ethylene from Teesside and Fife in the UK to Antwerp. In the current situation a total of 520,000 tonnes/year are transported to Antwerp from both UK production sites: 420,000 tonnes/year from Fife and 100,000 tonnes/year from Teesside. Transportation distances between Fife and Antwerp and between Teesside and Antwerp are 800 and 560 km, respectively.

One plan of APPE is to connect the ethylene pipeline network in Scotland and North England to the network in Belgium, The Netherlands and the Ruhr area/Cologne. For this 'inter-connector', a pipeline would be required with starting point in Hull and following a route from Hull via Peterborough, bending south there and then connecting to Antwerp via Rotterdam.

With transportation to Antwerp via this Inter-connector, transportation distances would be approximately 610 kilometres for the amount imported from Teesside and approximately 960 kilometres for the 400,000 tonnes imported from Fife.

6.5.2 Detour factor and load factor

To determine the energy consumption and emissions related to both transportation by short-sea tanker and transportation by pipeline, we first determined the average transportation distance for both modalities. This was done by taking the weighed average for ethylene originating from Teesside and from Fife.

The resulting average transportation distances are 750 kilometres for transportation by ship (estimated with MS Encarta) and 890 kilometres for transportation by pipeline (APPE).
Thus, short-sea shipping is the reference mode (0% detour factor) and the detour factor for transportation by pipeline is approximately 18%.

6.5.3 Energy consumption and emissions

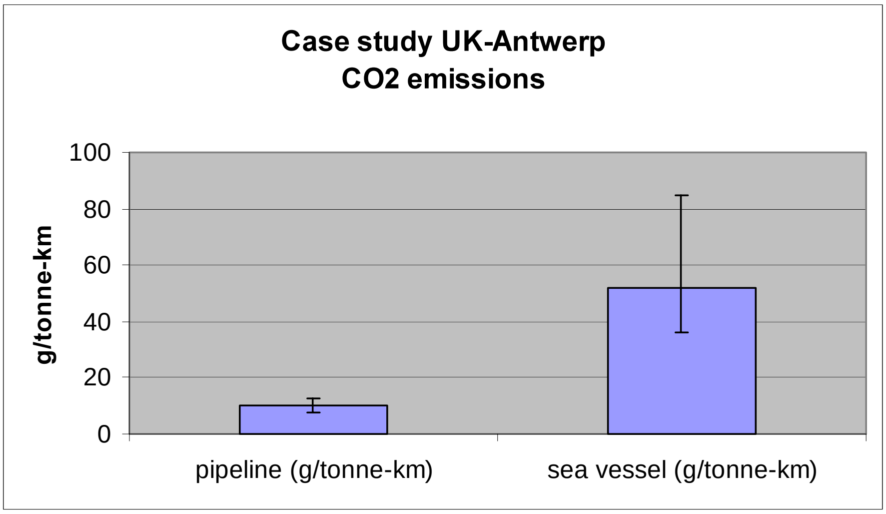
The results for energy consumption are given in Table 23.

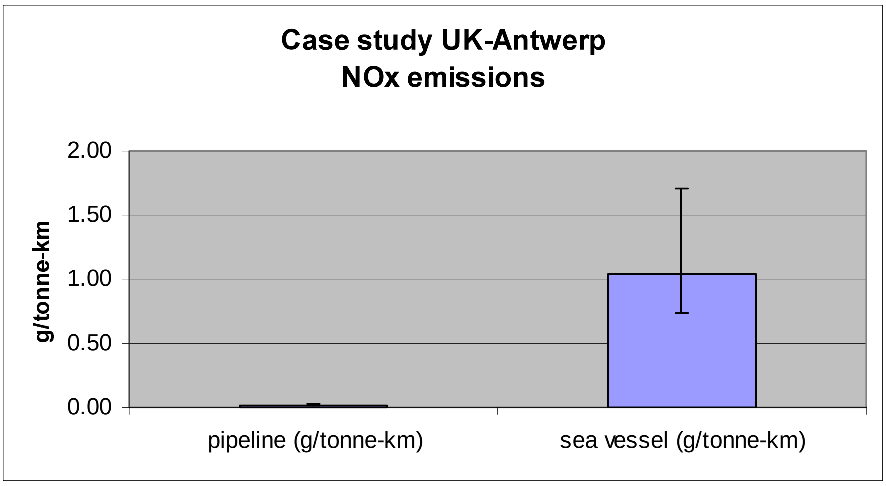
Table 23 Energy consumption of competing options for ethylene transport between UK and Antwerp. Figures apply to transport only *and exclude consumption resulting from initial compression (pipeline) or cooling (sea vessel),* and should therefore not be used for environmental comparison

	Specific energy consumption MJ/tonne-km	Detour factor	Comparison of energy consumption MJ/tonne-km
Sea vessel		0%	
- best case	0.45		0.45
- average	0.64		0.64
- worst case	1.05		1.05
Pipeline		18%	
- best case	0.11		0.13
- average	0.14		0.17
- worst case	0.18		0.22

The resulting emissions are shown in the following figures. We have not added emissions of propylene or ethylene to the VOC figures, as good estimates are not available for all of the modes. When considered, this may substantially change the picture for VOC. Please see section 2.2.2, 3.3.3, 4.3.3, and 5.3.4 for discussion of this contribution.

Figure 5 CO_2, NO_X and PM_{10}-emissions from ethylene transport between UK and Antwerp. Figures apply to transport only *and exclude emissions from initial compression (pipeline) or cooling (sea vessel),* and should therefore not be used for environmental comparison





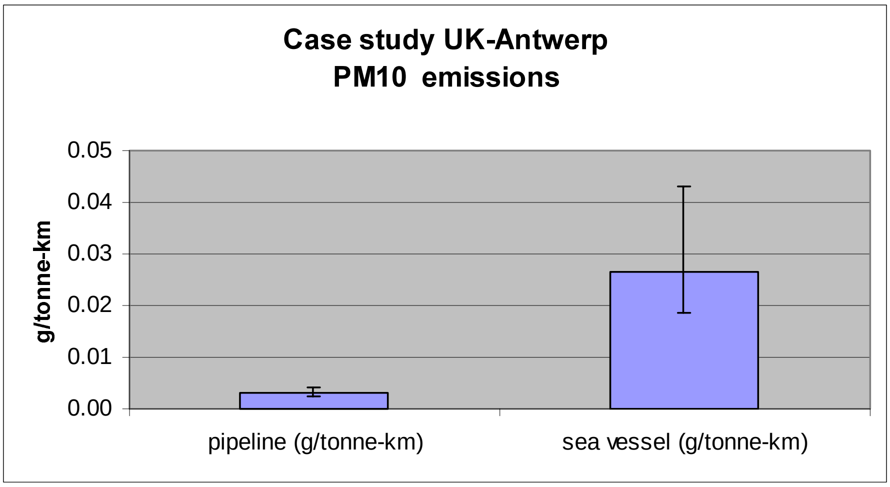
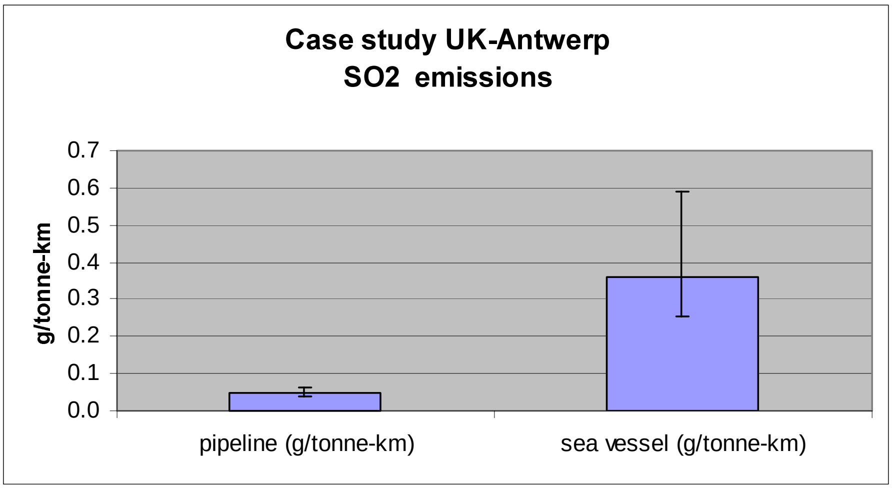
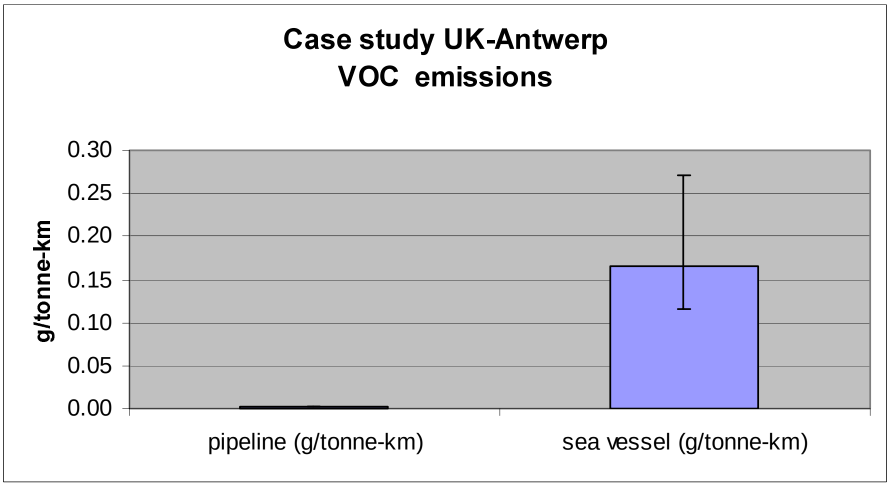


Figure 6 SO_2 and VOC-emissions from ethylene transport between UK and Antwerp. Figures apply to transport only *and exclude emissions from initial compression (pipeline) or cooling (sea vessel), and evaporative VOC-emissions,* and should therefore not be used for environmental comparison





6.6 Case 3: From UK to northern Germany (Stade)

The third case concerns the export of ethylene from the UK to northern Germany. According to industry, 300,000 tonnes on average are transported yearly by short-sea vessel from Teesside to Stade near Hamburg. According to the same source, the weight of a typical cargo is 2,500 tonnes.

Connecting Teesside by pipeline to Antwerp by means of the Inter-connector (see section 6.2) and the construction of the Chemcoast pipeline would render it possible to transport those 300,000 tonnes/year by pipeline.

Transportation by ship can take place along an almost straight line from Teesside to Hamburg, resulting in a distance of approximately 740 kilometres (estimated with MS Encarta). Transportation by pipeline, on the other hand, requires a significant detour through Belgium and Westphalia before arriving in Lower Saxony. The total pipeline length for this route adds up to 1,250 kilometres (estimated by APPE).

The methodology applied to calculate energy consumption and emissions is the same as in both previous sections and is therefore not discussed in this section.

6.6.1 Energy consumption and emissions

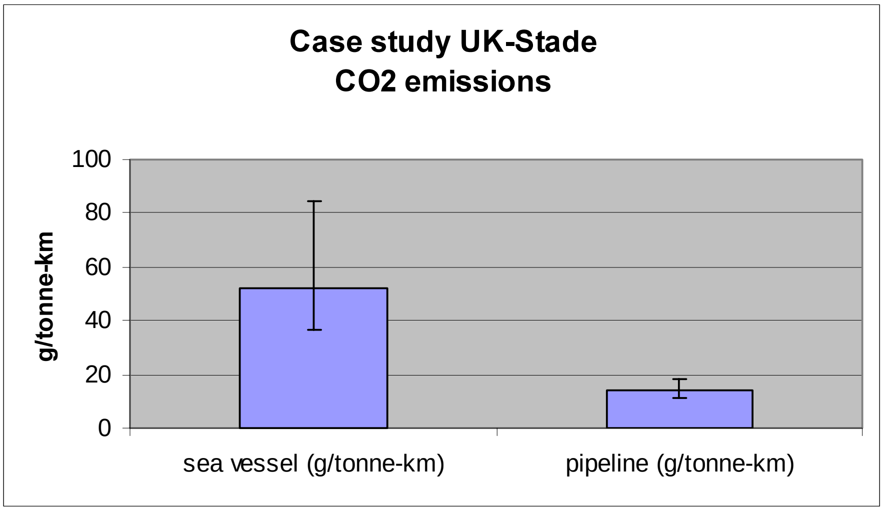
The results for energy consumption are given in Table 24.

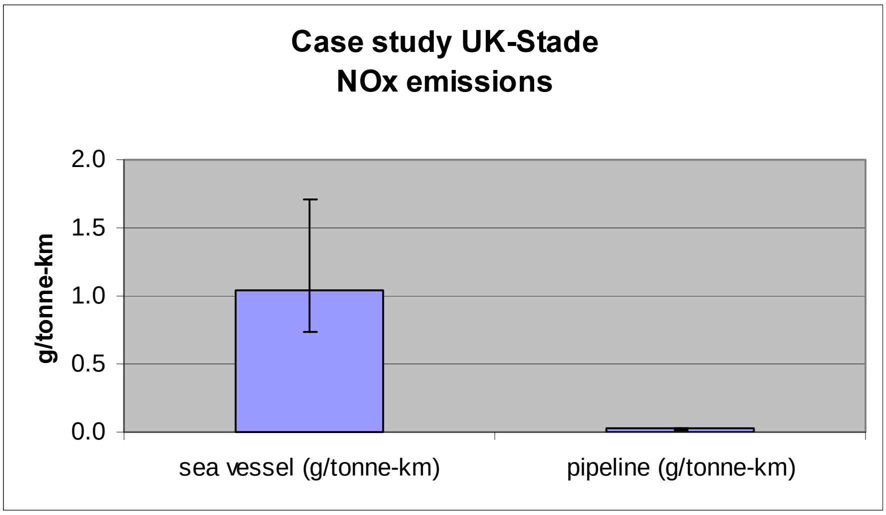
Table 24 Energy consumption of competing options for ethylene transport between UK and Stade. Figures apply to transport only *and exclude consumption resulting from initial compression (pipeline) or cooling (sea vessel),* and should therefore not be used for environmental comparison

	Specific Energy Consumption MJ/tonne-km	Detour factor	Comparison of energy consumption MJ/tonne-km
Sea vessel		0%	
- best case	0.45		0.45
- average	0.64		0.64
- worst case	1.05		1.05
Pipeline		69%	
- best case	0.11		0.19
- average	0.14		0.24
- worst case	0.18		0.30

The resulting emissions are shown in the following figures. We have not added emissions of propylene or ethylene to the VOC figures, as good estimates are not available for all of the modes. When considered, this may substantially change the picture for VOC. Please see section 2.2.2, 3.3.3, 4.3.3, and 5.3.4 for discussion of this contribution.

Figure 7 CO_2, NO_X and PM_{10}-emissions from ethylene transport between UK and Stade. Figures apply to transport only *and exclude emissions from initial compression (pipeline) or cooling (sea vessel),* and should therefore not be used for environmental comparison





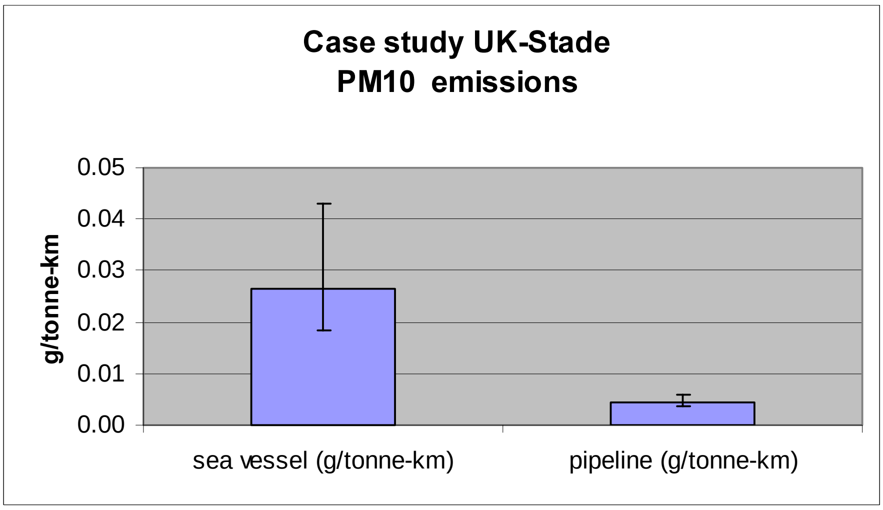
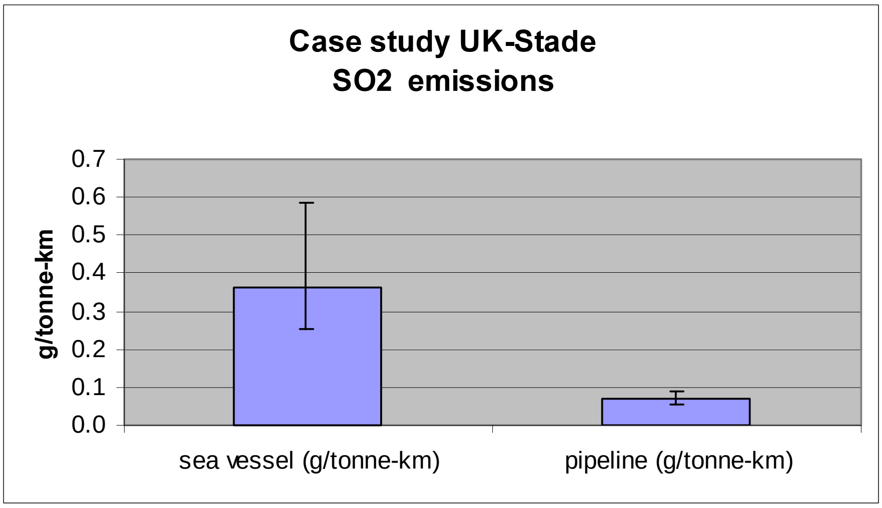
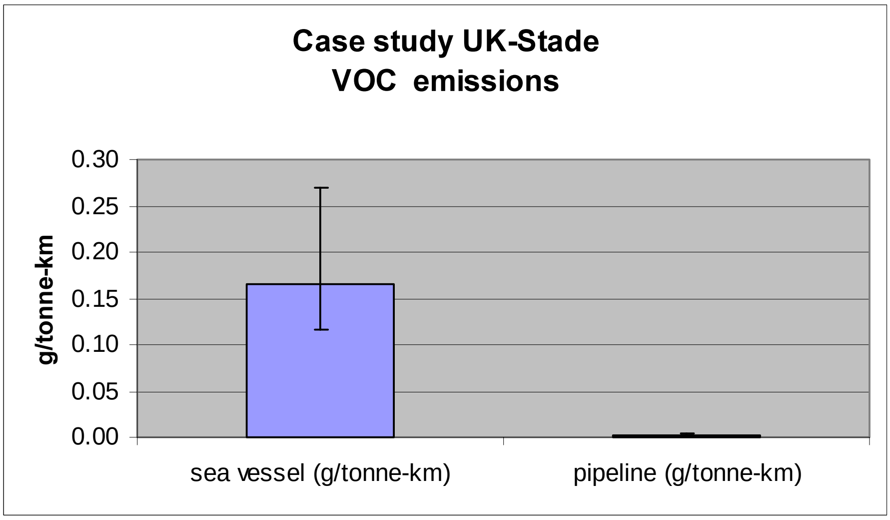


Figure 8 SO_2 and VOC-emissions from ethylene transport between UK and Stade. Figures apply to transport only *and exclude emissions from initial compression (pipeline) or cooling (sea vessel), and evaporative VOC-emissions,* and should therefore not be used for environmental comparison





7 A synthesis of case studies and emissions

7.1 Introduction

In the previous chapters, calculations were made separately for emissions of CO_2, NO_x, PM_{10}, SO_2 and VOC resulting from transport of ethylene and propylene by pipelines and other modes of transport. This resulted in five separate graphs per case study.

In this chapter we will strive for simplification of this presentation by summing the different emissions to give one single indicator. This has several advantages. For example, freight transport by electric train generally produces significantly lower emissions of CO_2 per tonne kilometre than diesel trains but at the same time higher emissions of NO_x.

There are several techniques to add different environmental impacts. In this report we use financial valuation of impacts. This has an additional advantage compared to non-financial weighting in that it allows a comparison of environmental benefits with economic costs.
Environmental impacts can be financially valued in two ways.

Cost of damage/nuisance plus avoidance/adaptation
Greenhouse gasses, pollutants and noise may damage human health, the natural environment, buildings and equipment as well as give rise to nuisance. Accidents are another possible source of social costs. Also, costs are sometimes incurred in trying to avoid or minimise the damage caused by pollution. Governments may, for example, decide to impose zoning restrictions on land that is subject to excessive noise or off-site risks. These costs can be categorised as avoidance costs of adaptation costs.

Costs of abatement and prevention measures
For some environmental effects, general (environmental) quality criteria may be laid down in the political decision-making process, i.e. across-the-board emission standards for all sectors of society. Extra emissions occurring under this kind of regime do not lead to extra environmental damage, but imply, rather, that somewhere in society additional emission abatement measures are required. Such measures to compensate for e.g. aviation emissions are once again associated with social costs.

And, finally, transaction costs, the costs of planning and monitoring the process, play a frequently forgotten but nevertheless often decisive role in the decision making process.

Annex G contains a more in-depth description of environmental valuation methodologies.

7.2 Shadow prices

Financial valuation leads to so-called shadow prices for environmental pollution. A literature overview by CE (|CE, 2002|) reveals that shadow prices assessed via the damage- and prevention cost approaches lie remarkably close together. A description of this analysis can be found in Annexes H and I.
Table 25 gives an overview of the shadow prices applied in this study.

Table 25 Applied shadow prices (€/kg)

emission	valuation (€ /kg)
CO_2	0.05
NO_x	7
PM_{10}	70
SO_2	4
VOC	3

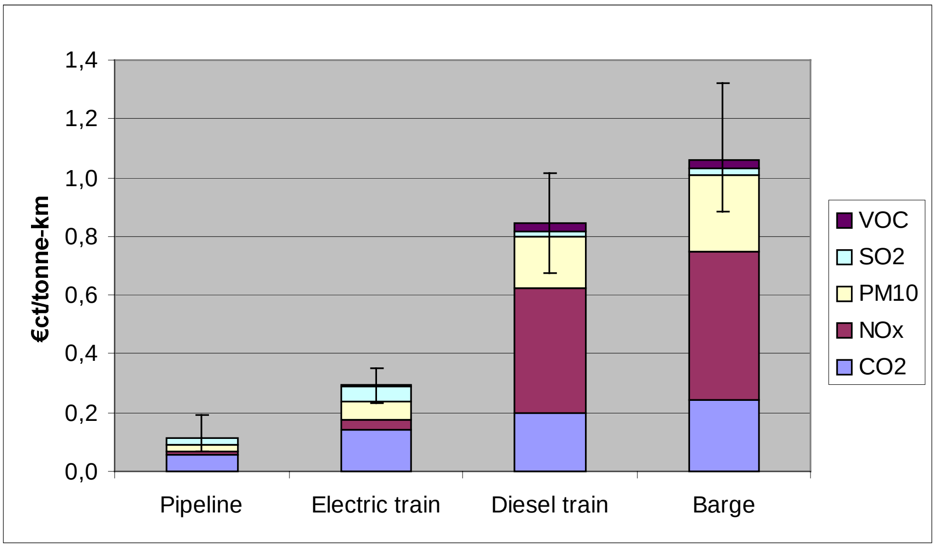
These shadow prices are valid for rural areas, which we deem most representative for our case studies.

7.3 Comparison of modes

As we have seen in Chapter 6, only the propylene transport case study Antwerp-Cologne delivered results that can be used for an absolute environmental comparison. In the ethylene case studies, in which pipelines and sea vessels are the only competitors, modes could not be compared properly because of the large influence of initial compression and cooling that could not be assessed in this study.

Therefore we only present an overview of financially valued emissions of the first case study: transport of propylene between Antwerp en Cologne.

Figure 9 Financially valued environmental impact of transport of propylene by four inland transport modes, excluding VOC-emissions from evaporation, *under the proviso that the propylene is not de-compressed - i.e. used in a high-pressure process - after transport*



Notes:

- In this study, the financially valued environmental impact of ethylene and propylene transport by *sea vessel* could not be properly compared with those of other modes, as for sea transport the products are cooled, whereas for land transport they are brought under pressure. Besides, the financial valuation of NO_X, PM_{10}, SO_2 and VOC-emissions at sea is likely to be lower than at land.
- Ethylene is not transported by rail and barge.

The conclusion from this figure is that, under the proviso mentioned and excluding evaporative VOC-emissions, in most cases the financially valued environmental impact from **propylene** transport by pipelines is lower than that of other **inland modes**. Electric trains can in some cases come close. Diesel trains and barges generally score much worse. The impact of sea vessels is not shown in the graph for reasons mentioned, but is also likely to be much higher than that of pipelines and electric trains.

A comparison of sea vessels with other modes is difficult for reasons mentioned below Figure 9.

Nevertheless, it is highly likely that the financially valued environmental impact of **ethylene and propylene transport by sea vessel** generally exceeds that of pipelines, as:

- analysis shows that even if the financial valuation of NO_X, SO_2, PM_{10} and VOC-emissions *at sea* is set at zero, the valuation of only CO_2 generally exceeds that of all emissions from pipeline transport;
- in case ethylene or propylene are cooled prior to transport by sea vessel, the energy requirement for the whole transport chain is likely to increase relative to other modes.

Degassing or other leakage of ethylene/propylene can have a dramatic influence on the shadow price of a mode – even when any possible climate effect is not included. If we add the worst-case ethylene/propylene emissions, and count it in the VOC category, the shadow price for the worst case may be very high, see Table 26.

Table 26 Extra price when including ethylene/propylene evaporation, worst case

Transport mode	indication of 'worst case' evaporative VOC-emissions (g/tonne-km)	indication of additional financial valuation (€ct/tonne-km)
Sea Vessels	2.9	0.9
Barge	0.005	0.002
Trains	0.008	0.002
Pipelines	0.8	0.2

The conclusion is that in this 'worst case' evaporation scenario, and under the proviso mentioned in Figure 9, the financially valued environmental impact of pipelines could arrive at the same order of magnitude as that of transport by electric train, but remain lower than that of other transport modes.

8 Towards a full environmental impact analysis

The analyses in chapters 6 and 7 merely constitute a - necessary - first step towards a full environmental impact analysis of a new pipeline system and should certainly not be considered as the definitive answer to the question of environmental friendliness of pipeline transport versus other modes.

In the previous chapters we compared the energy use and emissions of transport by pipeline with those of other modes. Though the average emissions of pipeline transport are relatively low, these results do not tell us whether the total impact of a new pipeline system is positive or negative. Therefore, the total environmental impact of a new pipeline should be investigated.

In this chapter we will highlight some additional analyses that should be made in order to answer these questions. In brief, the analysis should be broadened with respect to the types environmental impacts considered, the parts of the 'well to wheel' transport chain (in particular costs of initial compression and cooling), and possible volume impacts of pipelines. Besides the average emissions per tonne-kilometre, also effects like changes in the total transport volume and the impact on noise nuisance and safety should be included. In addition to that, the environmental impact of building new infrastructure can also be an important factor in the overall environmental comparison. We will discuss each of these factors in more detail.

8.1 Factors of influence

In general, to examine the total environmental impact of new transport infrastructure, all potential factors of influence, direct or indirect, should be duly accounted for, in particular (see |CE/RIVM, 2003|):
- **volume effects** (effects on total transport volume);
- **substitution effects** (modal shift, due to competitive characteristics);
- **environmental efficiency effects** (effects on environmental characteristics);
- **transport efficiency effects** (effects on logistical characteristics).

More specifically, in the case of judging the environmental impact of a new pipeline for ethylene or propylene all of the following effects should be included:

1 Changes in transport volumes and modal split:
- effects on the total transport volume of ethylene and propylene;
- effects on the total transport volume of polymers made from ethylene or propylene;
- effects on the market shares of the different modes.

2 Changes in environmental and transport characteristics:
- environmental and transport characteristics of the new pipeline system;
- effects on environmental and transport characteristics of all competing modes, including existing pipeline systems;
- impact on noise nuisance.

3 Impact of infrastructure:
- impact of building of the new infrastructure;
- changes in effects of maintenance of infrastructure.

4 Changes in safety risks and congestion:
- traffic victims;
- external risks because of the ethylene or propylene;
- changes in congestion levels.

5 Impact of initial compression and cooling:
- energy required for bringing the ethylene/propylene into a state suitable for transportation (cooling or compression into a fluid state).

6 Impact of quality grades.

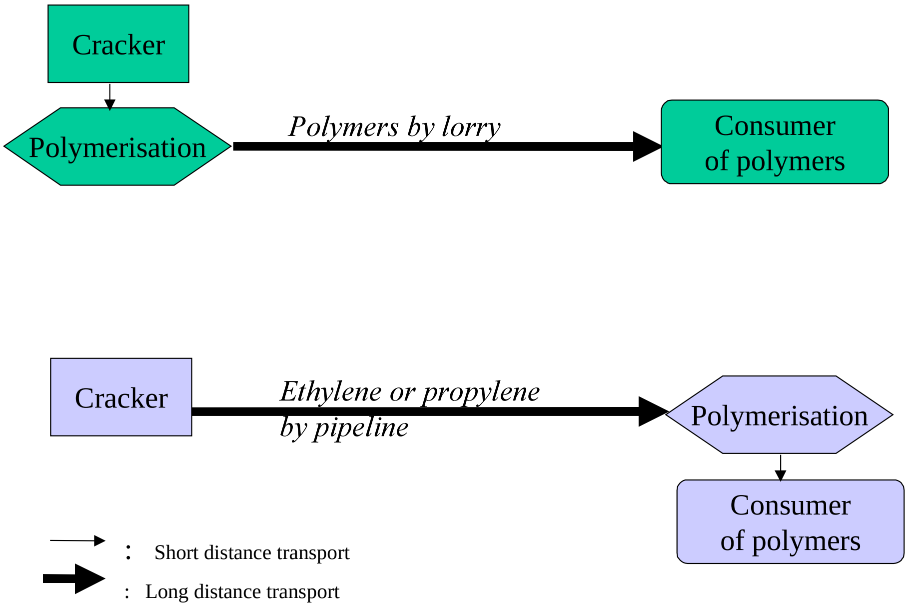
8.2 Changes in transport volumes and modal split

New pipeline infrastructure may drastically change logistics of petrochemical-related industry.

First of all the ethylene and propylene transport volumes of the different modes will change. This can affect both the modal split and the total transport volume.
Most ethylene and propylene is currently carried by short-sea shipping, inland shipping, rail or (existing) pipelines. A new pipeline could decrease the transport volumes of these other modes.
Apart from a shift from other modes, some of the transport volume of the new infrastructure can also be newly generated transport volume. A new pipeline can be an economic stimulation for the sector. Therefore, a new pipeline for ethylene or propylene might lead to an increase of the total transport volume of these petrochemicals.

It is also possible that the construction of a new pipeline will affect production locations, particularly of polymerisation plants. Currently, these plants are usually situated close to the crackers. A pipeline system could make it possible to move these plants towards locations that are closer to the industry that uses the polymers. In that case the pipeline will cause an increase of the total transport volume of ethylene and propylene. The transport volume of polymers, which are currently mainly transported by lorries would decrease. This is illustrated by Figure 10.

Figure 10 Possible changes in production locations because of a new pipeline



Changes in the locations of polymerisation plants means that new plants need to be built. This has several environmental impacts like due to the energy needed for building new plants and the possibly higher energy efficiency of these plants compared to existing plants because of technological developments.

Currently it is impossible to predict which of these effects would be dominant. Therefore, a more detailed analysis of the market for ethylene, polyethylene and their polymers is needed.

8.3 Changes in environmental and transport characteristics

The environmental impact of a new pipeline is highly dependent on the energy use and emissions per tonne-kilometre compared to those of competing modes. These data have been worked out in this report (see chapter 2-6).

With these data and the changes in transport volumes (see previous section), the changes in energy use and emissions of the modes can be calculated.

However, a new pipeline may also cause changes in the current data, like:
- utilisation factors of competing modes might improve under the pressure of stronger competition with pipeline transport;
- utilisation and energy use of existing pipelines might change once interconnected by new pipelines.

Apart from the impact on energy use and emissions, the impact on noise nuisance should be included. Particularly changes in rail transport or road transport will affect the noise nuisance levels.

8.4 Impact of new infrastructure

Building new infrastructure usually costs a lot of energy and money. For a proper examination of the environmental impact of a new pipeline, this energy consumption should be included.

Building a new pipeline might also affect natural habitats. This effect should be included. Besides building new infrastructure, also changes in maintenance of infrastructure should be investigated.

8.5 Changes in safety risks

For road transport in particular, traffic accidents constitute an important issue. Changes in the modal split may cause changes in traffic safety.

In case of transport of hazardous goods like ethylene and propylene another important issue is *external risks*. Transport of ethylene and propylene by pipelines has a different risk profile than transport of these commodities by ship or rail. Transport of *polymers* will have no significant external risks, because they are no hazardous goods.

The impact of a new pipeline on both traffic victims and external risks because of the transport of hazardous goods needs further study.

8.6 Impact of initial compression or cooling

Unlike conventional products, ethylene and propylene require a change of state for transportation. The energy requirements for this, initial compression or cooling, appear to dominate total energy costs associated with transportation, but could unfortunately not be quantified in the context of this study. These impacts must be considered carefully in each case. A broad overview of the whole transport chain is needed. At the *consumer* end, it must be known how the commodity is stored and used. For example, if the consumer uses ethylene in a process where it must be put under high pressure, transporting it in a cooled state would require a large extra energy expenditure for recompression that would be unnecessary if the ethylene was transported under high pressure. At the *producer* end, it must likewise be known in what state it is produced. One conversion from cooled to compressed or vice versa during transportation will produce a much higher energy cost for the whole transportation chain.

In some cases cooled transportation by ship may be better, such as if ethylene/propylene is produced and used in processes at very low or no pressure. When ethylene/propylene is used in high-pressure processes, pipeline transportation is most favourable.

The origin (e.g. electric or from fossil fuels) of the energy for initial compression/cooling will also be an important factor to consider when assessing a particular transport chain.

8.7 Impact of quality grades

The impact of different quality grades of propylene may also be considered, as they are used in different processes. Pipelines generally only transport the highest grade (polymer grade).

9 Conclusions

9.1 Conclusions

Pipeline and sea shipping currently most dominant modes

Transportation by pipeline accounts for respectively 50% of the total transport volume (in tonne-kilometres per year) of ethylene and 15% of the total transport volume of propylene, transported within the EU for distances above 50 km. The market share of pipelines expressed in tonnes per year is even 84% (ethylene) and 50% (propylene).

Transportation of ethylene and propylene by ship accounts for transportation of 55% of all tonne-kilometres of ethylene and 54% of all tonne-kilometres of propylene not produced and processed at integrated chemical industry sites.

The market shares of other modes are considerably smaller. Inland shipping accounts for about 2% of all tonne-kilometres of propylene in the EU. The market share of rail transport of propylene amounts to 13% of all tonne-kilometres. In the EU, ethylene is neither transported by barge nor by rail.

Because of safety reasons, road transport is not a relevant transport mode for ethylene and propylene. Therefore, road transport is not included in the comparison.

Energy use of initial compression or cooling
To be suitable for transport, ethylene or propylene needs to be under high pressure or low temperature. In most cases, the commodities come under high pressure out of the cracker, which makes that there is no extra compression required, when it transported under high pressure. However, in other cases extra compression or cooling is needed. The energy use of these initial processes are very high, and will in those cases dominate total energy costs. In the framework of this study it appeared impossible to give a general value for this energy cost, but it appears to amount to several hundreds of MJ of primary energy per tonne. Cooling is less energy intensive than compression.

It is vital to consider this cost when comparing between modes, and a proper understanding of the environmental score per mode requires detailed knowledge of production and use of the ethylene/propylene.

In the framework of this study it is only possible to compare the environmental impact of transport of propylene with 'compressed' modes pipeline, rail and barge, under the proviso *that the propylene is not de-compressed - i.e. used in a high-pressure process - after transport*.

Energy use of pipeline transport excluding initial compression
Original data on energy use of pipeline transport of oil products appears to be very scarce. All studies found refer to the same study by Mittal (1978). This study therefore was our first information source.

The client provided a second information source, namely concrete information on the number and energy use of pumps and valve stations in a number of cases.

Finally, CE Delft developed a third original source of information, namely a relatively simple theoretical model to calculate energy use. The energy use of pipelines depends on pipe diameter and roughness, density and viscosity of the fluid, and flow rate through the pipe.

The data from the different sources match well and fall in the range of 0.11 - 0.18 MJ/tonne kilometre of primary energy. Therefore, we use 0.14 as a middle value and 0.11 and 0.18 for best and worst case analysis, respectively.

Energy use and emissions of pipeline transport compared to other modes
As already said, it is only possible to make a quantitative comparison of energy use and emissions of *propylene* transport by the 'inland' modes: pipelines, rail and barges, under the proviso that the propylene is not decompressed - i.e. used in a high-pressure process - after transport. Under this proviso, the analysis yield the following results:

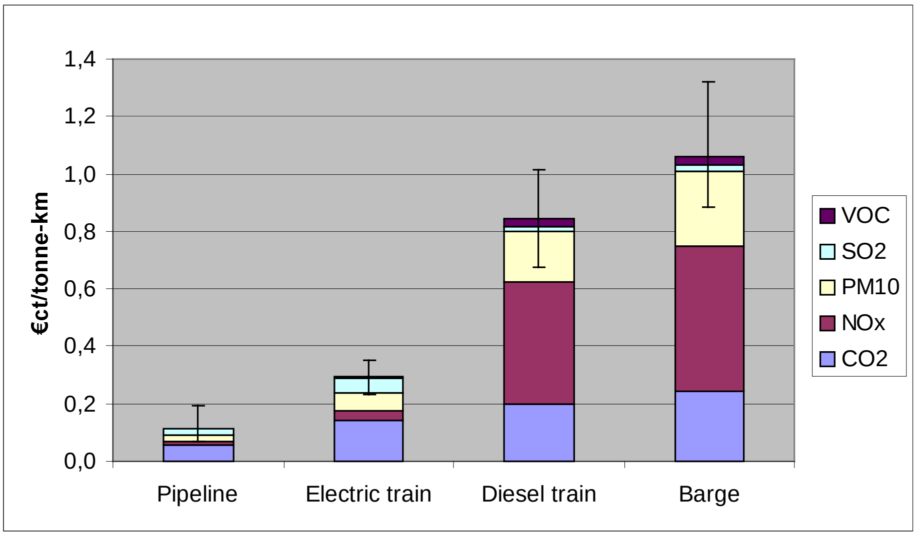
- energy use and CO_2-emissions from propylene transport by pipeline are, per tonne kilometre, about 70 to 80% lower than those of the competing modes in the same case;
- NO_x-emissions from pipeline transport are 70% lower (compared with electric trains) up to even 98% lower (compared with inland shipping);
- PM_{10}-emissions are 70 to 85% lower;
- SO_2-emissions from pipeline transport are comparable to those of diesel trains and barges but considerably lower than of electric trains;
- For a sound comparison of the VOC-emissions, we need more reliable data on the effects of leakage and evaporation.

For *ethylene* transport - only by pipeline and sea vessel - it is not possible to make a quantitative comparison given the differences in energy use for compression (pipelines) and cooling (sea vessel). However, when the ethylene is to be used in high-pressure processes, we would generally expect pipeline transport to require much less energy and produce much lower emissions than transport by sea vessel.

Adding up to one environmental parameter
Several techniques are in place to add different environmental impacts. In this report we use financial valuation of the impacts. All emissions are valued using so-called *shadow prices* that are based either on the damage caused by pollutants or on the social costs to prevent pollution.

For reasons mentioned above, we only present an overview of financially valued emissions of transport of propylene by pipeline, rail, and barge.

Figure 11 Financially valued environmental impact of transport of propylene by four inland transport modes, excluding VOC-emissions from evaporation, *under the proviso that the propylene is not de-compressed - i.e. used in a high-pressure process - after transport*



The conclusion from this figure is that, under the proviso mentioned and excluding evaporative VOC-emissions, in most cases the financially valued environmental impact from **propylene** transport by pipelines is lower than that of the other **inland modes**. Electric trains can in some cases come close. Diesel trains and barges generally score much worse. The impact of sea vessels is not shown in the graph for reasons mentioned, but is also likely to be much higher than that of pipelines and electric trains.

As already said, comparison of sea vessels with other modes is difficult. Nevertheless, it is highly likely that the financially valued environmental impact of **ethylene and propylene transport by sea vessel** generally exceeds that of pipelines, as:

– analysis shows that even if the financial valuation of NO_X, SO_2, PM_{10} and VOC-emissions *at sea* is set at zero, the valuation of only CO_2 generally exceeds that of all emissions from pipeline transport;
– in case ethylene or propylene are cooled prior to transport by sea vessel, the energy requirement for the whole transport chain is likely to increase relative to other modes.

9.2 Recommendations

This study certainly provides insight into an important environmental impact of pipeline transport, namely the emissions arising from the transport itself. However, the assessment is too narrow to justify recommendations as to whether the interconnection of the pipeline networks would benefit the environment. To be able to make such a recommendation, the following aspects would require closer investigation.

1 Changes in transport volumes and modal split resulting from the use of new pipelines:

- effects on the total transport volume of ethylene and propylene;
- effects on the total transport volume of polymers made from ethylene or propylene;
- effects on the market shares of the different modes.

2 Changes in environmental and transport characteristics:

- environmental and transport characteristics of the new pipeline system;
- effects on environmental and transport characteristics of all competing modes, including existing pipeline systems;
- impact on noise nuisance.

3 Impact of the construction of the new infrastructure itself:

- impact of building of the new infrastructure;
- changes in effects of maintenance of infrastructure.

4 Changes in safety risks:

- traffic victims;
- external risks because of the ethylene or propylene.

5 Impact of initial compression and cooling:

- energy required for bringing the ethylene/propylene into a state suitable for transportation.

6 Impact of quality grades.

And finally, in order to arrive at a full social cost-/benefit assessment of the planned interconnection, economic aspects, such as investment and maintenance costs and operational cost savings, should also be included in the analysis.

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Date	*Organisation*	*Person*	*Subject*
13/8	Chemgas	Mr. J. Smit Roeters	Transportation by barges
21/8	PLE	Mr. S. Bendel	Pipeline specifications and energy consumption
5/8	Shell Moerdijk	Mrs. N. Eikelenboom	Production, storage, transhipment
6/8	DOW Terneuzen	Mr. J. Alberti	Production, storage, transhipment
12/8	Unigas	Mr. Van Benten	Energy consumption of sea vessels
19/8	Lauritzen Kosan	Mr. H. Ahrenst	Energy consumption of sea vessels
21/8	APPE	Mrs. N. Schoub	Olefins movements in Europe above 50 km
6/8	Railion	Mr. Wieger Visser	Propylene transport by rail

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Emissions of pipeline transport compared with those of competing modes

Environment analysis of ethylene and propylene transport within the EU

Annexes

Report

Delft, November, 2003

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A Energy consumption of pipeline transport - calculation with industry information

We have obtained information on two different pipeline systems. The calculations are performed in parallel for each pipeline in section A.1 and A.2 below.

A.1 Pipeline system 1

The pipeline
The pipeline in question will connect Rotterdam, Antwerp, and Cologne.

Transport volume
The transport volume to be carried by this pipeline system is listed below, and is based on the current transport volume that is assumed to be replaced by the pipeline.

Table 27 Total transport volume of ethylene and propylene

tonnes	2,200,000
average distance (km)	272
total transport volume (tonne-km)	598,400,000

Energy consumption for transportation
The energy consumption of the pipeline system is calculated based on the number of pump and valve stations, and their use of electric energy. The *electric* energy use is converted to *primary* energy use assuming an energy return of 38% in the production and distribution of electrical energy. The results are shown below.

Table 28 Pipeline energy use

Pipeline energy use		***Electric* energy use**			***Primary* energy use**
	number	per station (MWh)	total (MWh)	total (MJ)	total (MJ)
Pump stations	4	2,240	8,960	32,256,000	84,884,211
Valve stations	76	2	152	547,200	1,440,000
			9,112	32,803,200	86,324,211

Total energy consumption
The energy consumption per tonne-km is obtained by dividing the total primary energy use with the total transport volume. This yields a primary energy consumption of pipeline transport of 0.14 MJ/tonne-km.

A.2 Pipeline Fife-Teesside-Antwerp

The pipeline

The calculation is based on data for the construction of a pipeline system, called the inter-connector, which is referred to in chapter 6.4. This pipeline would connect Fife in Scotland with Teesside in Northeast UK and Antwerp in Belgium.

Volume to be transported by the pipeline

The transport volume to be carried by this pipeline system is supposed to be equal to the volume currently transported by ship. The transport volume and distances are listed below.

Table 29 Total transport volume of ethylene and propylene

	Scotland-Antwerp	Teesside-Antwerp	Total
tonnes	820,000	400,000	
average distance (km)	906	616	
total transport volume (tonne-km)	742,920,000	246,400,000	989,320,000

Energy consumption for transportation

The energy consumption of the pipeline system is calculated based on the number of pump and valve stations, and their use of electric energy. The *electric* energy use is converted to *primary* energy use assuming an energy return of 38% in the production and distribution of electrical energy. The results are shown below.

Pipeline energy use		**_Electric_ energy use**			**_Primary_ energy use**
	number	per station (MWh)	total (MWh)	total (MJ)	total (MJ)
Pump stations	5	2,240	11,200	40,320,000	106,105,263
Valve stations	36	2	72	259,200	682,105
			11,272	40,579,200	106,787,368

Total energy consumption

The energy consumption per tonne-km is obtained by dividing the total primary energy use with the total transport volume. This yields a primary energy consumption of pipeline transport of 0.11 MJ/tonne-km.

B Energy consumption of pipeline transport - theoretical approach

B.1 Introduction

As stated in main report the actual system for transportation of ethylene and propylene by pipeline consists of several steps:

- compression of the commodity to the required transportation pressure;
- intermediate storage at producer;
- transportation, interim compression with boosters;
- receiving and intermediate storage at consumer.

Storage was assumed to involve zero energy consumption. Calculation of the energy consumption of initial compression and transportation by pipeline are described in following two sections.

B.2 Initial compression to transportation pressure

Ethylene and propylene are produced by steam cracking at a pressure of approximately 2 bar. As far as can be deducted from available information concerning transportation by pipeline of ethylene and propylene both commodities are typically compressed to 100 bar (see |AVIV|, |Gasunie|, |EPDC|). At this pressure both ethylene and propylene are supercritical at room temperature, the critical pressure of ethylene and propylene at room temperature being respectively 55 bar and 15 bar.

Compression of ‘fresh’ ethylene and propylene in relation to transportation by pipeline therefore comprises of a multistage process:

- compression of the gas phase up to critical pressure;
- cooling and condensation of the compressed gas;
- additional compression of the fluid phase up to transportation pressure.

Gas compression

Energy consumption of compression of the gas phase has been calculated applying the relation for adiabatic work of a compressor.

$$W = \frac{\kappa \cdot R \cdot T_1}{(\kappa - 1) \cdot \eta} \left[1 - \left(\frac{P_2}{P_1} \right)^{\left(\frac{\kappa - 1}{\kappa} \right)} \right]$$

in which: W = work (kJ/kg)
κ = isentropic compressibility
R = gas constant (kJ/kg·°K
P = pressure (kPa)
T = initial temperature (°K)
η = isentropic efficiency

In view of energy conservation, multistage compression with intercooling was assumed. Number of compression stages was assumed to be 2 for propylene and 3 for ethylene.

Adapted values for the different parameters are given in Table 30.

Table 30 Data used for calculation of energy required for compression

	Ethylene	Propylene
κ = isentropic compressibility	1.4	1.4
R = gas constant (kJ/kg·°K	0.2964	0.198
P = pressure (kPa)		
initial	2	2
final	55	15
number of compression stages	3	2
T = initial temperature (°K)	308	308
h = isentropic efficiency	75%	75%
k = isentropic compressibility	1,237	1,145

Resulting work is 550 MJ_e/tonne of ethylene and 226 MJ_e/tonne of propylene. This corresponds to 1447 MJ and 595 MJ of primary energy use.

Additional compression of fluid phase
Energy consumption related to additional compression of the liquefied commodity up to transportation pressure was calculated applying the relation for work by an adiabatic pump and assuming a polytropic efficiency of 75%:

$$W\,(kJ/kg) = \frac{\Delta P}{\rho \times 75\%}$$

in which: ΔP = pressure drop (kPa)
ρ = density (kg/m^3)

Adapted values for the different parameters are given in Table 31.

Table 31 Data used for fluid phase compression

	Ethylene	Propylene
ΔP = pressure drop (kPa)	45	85
ρ = density (kg/m^3)	365	525

Work amounts to 16.4 MJ_e/tonne for ethylene and 21.6 MJ_e/tonne for propylene. This corresponds to 43.2 MJ and 56.8 MJ of primary energy use.

B.3 Transportation through the pipe

Energy consumption of pipeline transportation of ethylene and propylene has been estimated based upon the Generaux relation with Fanning friction factor:

$$\Delta P = \frac{4{,}07\times10^{10}\times G^{1{,}84}\times 0{,}001\cdot\mu^{0{,}16}}{\rho\times d^{4{,}84}}$$

in which: ΔP = pressure drop (kPa)
G = flow rate (kg/s)
ρ = density (kg/m^3)
μ = viscosity (mNs/m^2)
d = internal pipe diameter (mm)

Energy consumption has been calculated from the pressure drop per unit of length of piping by applying the standard relation for work for an adiabatic pump and assuming a polytropic efficiency of 75%:

$$W\,(kJ/kg) = \frac{\Delta P}{\rho\times 75\%}$$

The assumed polytropic efficiency has been adapted from |KOGA|, a desk top studying underground storage of supercritical CO_2.

For ethylene and propylene pipelines following values of the parameters in Generaux's relation have been adapted.

Table 32 Assumptions for calculation of pipeline energy consumption

	Ethylene	Propylene
Parameters		
G = flow rate (kg/s)	39	27
ρ = density (kg/m^3)	370 (at approximately 80 bar)	536 (at approximately 80 bar)
μ = viscosity (mNs/m^2)	0.12	0.15
d = internal pipe diameter (mm)	250	250
Resulting electric energy consumption (MJ_e/tonne·km)	0.05	0.05
Primary energy consumption (MJ/tonne·km)	0.13	0.12

The values adapted for a propylene pipeline refer to the QRA EPDC propylene pipeline between Antwerp and the Ruhr area, currently under construction (see |AVIV|). The values adapted for an ethylene pipeline were taken from a predesign study concerning the construction of a pipeline to Delfzijl |Gasunie|.

The adapted values for flow rate refer to a velocity of the supercritical liquid of approximately 1.5 m/s. The transported amounts are rather significant, approximately 1,240,000 tonnes/year for propylene and approximately 860,000 tonnes/year for ethylene.

C Short sea vessels

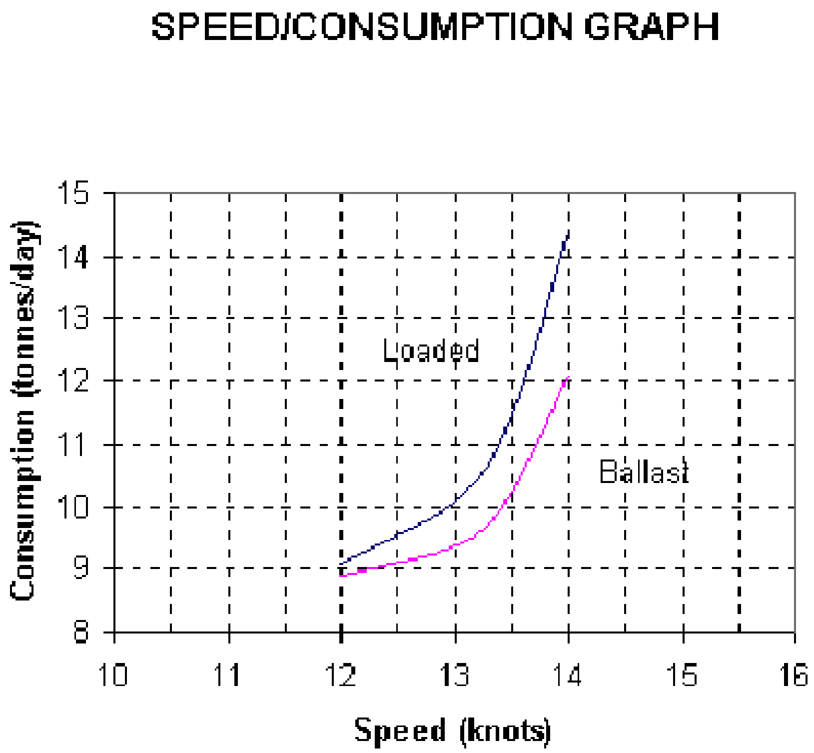
Table 33 Characteristics of short sea ships used for propylene and ethylene transport.

Ship	Year	Propylene /Ethylene	Capacity (tonnes)	Speed (knots)	Energy consumption	
					tonnes /day	MJ/ tonne-km
Anthony Veder fleet						
Coral Anthillarum	1982	P	1,716	10.5	7	0.37
Coral Acropora	1993	P	1,936	10.5	7	0.33
Victoria Lily	1992	P	1,656	12	9.1	0.44
Apollo Pacific	1988	P	1,609	13.5	9.5	0.42
Coral Obelia	1996	P	2,305	13	10	0.32
Gas Jaya	1998	P	1,627	13	9	0.41
Gas Trust	1996	P	1,835	13.5	11.3	0.44
Coral Favia	2001	P	2,057	14.5	13.4	0.43
Coral Meandra	1996	P	2,571	13.5	11.8	0.33
Coral Isis	1988	P	3,657	13	16	0.32
Pr. Johan Willem Friso	1989	E/P	2,353	13.5	4.5	0.44
Coral Rubrum	1999	E/P	3,019	14.5	14	0.31
Coral Pavona	1995	E	3,994	13	17	0.31
LauritzenKosan fleet						
Greta Kosan	1990	P	2,680	14	13	0.33
Sigas General	1982	P	2,172	13	9.7	0.33
Exmar fleet:						
Lady Elena	1998	P	1,775	13.5	9	0.36
Lady Barbara	1990	P	1,798	14.6	9.5	0.35
Lady Martine	1998	P	1,690	13	9	0.39
Polar Endurance		E/P	5,741	15	28	0.31
Polar Discovery	1989	E/P	5,841	15	28	0.31

Sources: www.anthonyveder.nl, www.lauritzenkosan.com, www.exmar.be

In calculating the energy consumption we use the following conversions:
1 knot is 1.852 km/h.
1 tonne fuel contains the equivalent of 42,700 MJ.

D Ship fuel consumption curves



Source: Greta Kosan, 2680 ton LPG carrier |Lauritzen Ship Owner A/S|

E Emissions and shadow prices

Emissions and shadowprices for the cases

Case 1 Antwerp - Cologne		Emissions (g/tonne-km)					Shadow prices (€ct/tonne-km)					
		CO_2	NOx	PM10	SO_2	VOC	CO_2	NOx	PM10	SO_2	VOC	Total
Barges	Average	48,50	0,72	0,038	0,05	0,09	0,24	0,51	0,26	0,019	0,027	1,06
	Best	40,42	0,60	0,031	0,04	0,08	0,20	0,42	0,22	0,016	0,023	0,88
	Worst	60,63	0,90	0,047	0,06	0,11	0,30	0,63	0,33	0,024	0,034	1,32
Electric train	Average	27,91	0,05	0,009	0,14	0,005	0,14	0,04	0,06	0,054	0,002	0,29
	Best	22,33	0,04	0,007	0,11	0,004	0,11	0,03	0,05	0,043	0,001	0,23
	Worst	33,49	0,06	0,01	0,16	0,006	0,17	0,04	0,07	0,065	0,002	0,35
Diesel train	Average	39,58	0,61	0,025	0,05	0,08	0,20	0,42	0,18	0,020	0,025	0,84
	Best	31,66	0,48	0,020	0,04	0,07	0,16	0,34	0,14	0,016	0,020	0,67
	Worst	47,49	0,73	0,030	0,06	0,10	0,24	0,51	0,21	0,024	0,030	1,01
Pipeline	Average	8,40	0,015	0,003	0,041	0,002	0,04	0,01	0,02	0,016	0,000	0,09
	Best	6,60	0,012	0,002	0,032	0,001	0,03	0,01	0,01	0,013	0,000	0,07
	Worst	10,80	0,020	0,003	0,052	0,002	0,05	0,01	0,02	0,021	0,001	0,11

Case 2 UK- Antwerp		Emissions (g/tonne-km)					Shadow prices (€ct/tonne-km)					
		CO_2	NOx	PM10	SO_2	VOC	CO_2	NOx	PM10	SO_2	VOC	Total
Sea Vessels	Average	51,85	1,05	0,03	0,36	0,17	0,26	0,73	0,19	0,144	0,050	1,37
	Best	36,28	0,73	0,02	0,25	0,12	0,18	0,51	0,13	0,10	0,035	0,96
	Worst	84,55	1,71	0,04	0,59	0,27	0,42	1,19	0,30	0,235	0,081	2,23
Pipeline	Average	9,94	0,02	0,003	0,05	0,00	0,05	0,01	0,02	0,02	0,001	0,10
	Best	7,81	0,01	0,002	0,04	0,00	0,04	0,01	0,02	0,02	0,000	0,08
	Worst	12,78	0,02	0,004	0,06	0,00	0,06	0,02	0,03	0,02	0,001	0,13

Case 3 UK- Stade		Emissions (g/tonne-km)					Shadow prices (€ct/tonne-km)					
		CO_2	NOx	PM10	SO_2	VOC	CO_2	NOx	PM10	SO_2	VOC	Total
Sea Vessels	Average	51,85	1,05	0,03	0,36	0,17	0,26	0,73	0,19	0,144	0,050	1,37
	Best	36,28	0,73	0,02	0,25	0,12	0,18	0,51	0,13	0,101	0,035	0,96
	Worst	84,55	1,71	0,04	0,59	0,27	0,42	1,19	0,3	0,235	0,081	2,23
Pipeline	Average	14,19	0,03	0,004	0,07	0,00	0,07	0,02	0,03	0,028	0,001	0,15
	Best	11,15	0,02	0,004	0,05	0,00	0,06	0,01	0,02	0,022	0,001	0,12
	Worst	18,24	0,03	0,006	0,09	0,00	0,09	0,02	0,04	0,035	0,001	0,19

Shadowprices

	Applied shadow prices (€/kg)				
	built up area	rural area	*low*	*average*	*high*
CO_2	0,05	0,05	*0,03*	*0,05*	*0,06*
NOx	12	7	-	-	-
PM10	300	70	-	-	-
SO_2	10	4	-	-	-
VOC	6	3	-	-	-

Energy consumption (MJ/tonne-km)		
Sea Vessels	Average	0,64
	Best	0,45
	Worst	1,05
Barge	Average	0,60
	Best	0,50
	Worst	0,75
Electric train	Average	0,45
	Best	0,36
	Worst	0,54
Diesel train	Average	0,48
	Best	0,39
	Worst	0,58
Pipeline	Average	0,14
	Best	0,11
	Worst	0,18

Emissions factors (g/MJ)					g/MJ
	CO_2	NOx	PM10	SO_2	VOC
Sea vessels	80,4	1,62	0,04	0,559	0,26
Barge	78,7	1,17	0,06	0,079	0,15
Diesel trains	80,1	1,23	0,05	0,10	0,17
Pipelines and elec.-trains	60,0	0,11	0,02	0,29	0,01

Detour Factors	Case 1	Case 2	Case 3
Sea vessels	-	0%	0%
Barge	2%	-	-
Trains	2%	-	-
Pipelines	0%	18%	69%

F Emissions of electricity production and refining

Table 34 Overview of emissions resulting from electricity production and from refining of petroleum products. Source: 'To shift or not to shift, that's the question' |CE/RIVM, 2003|

Emission factors for electricity production	*Year*	*CO_2* g/MJe	*NO_x* g/MJe	*SO_2* g/MJe	*PM_{10}* g/MJe	*VOC* g/MJe
EU average (incl. nuclear)	2000	127.4	0.33	0.74	0.04	0.02
EU average (excl. nuclear)	2000	177.7	0.45	1.04	0.05	0.03
EU average (excl. nuclear)	2010	158.0	0.29	*0.77	0.05	**0.03
Emissions from refining		*CO_2*	*NO_x*	*SO_2*	*PM_{10}*	*VOC*
		[kg/GJ	[g/GJ	[g/GJ	[g/GJ	[g/GJ
Fuel oil		5.4	33.4	34.1	1.2	79.9
Diesel		6.9	37.3	53.6	1.1	87.4

* This value was not from the source above, but was estimated by assuming that the SO_2-emission factor in 2010 in EU would reduce as much as is expected in the Netherlands. According to same source, this reduction is 26%.

** VOC-emission factors in 2010 are assumed to be the same as in 2000, which is our best estimate.

G Financial valuation of environmental impacts

G.1 Damage and prevention costs

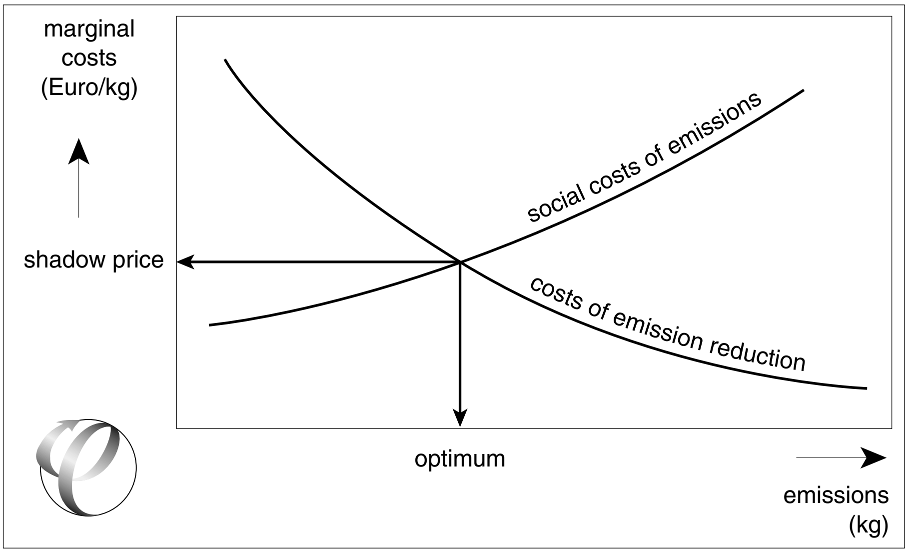
The social costs of emissions can be divided into two categories:

– **Costs of damage/nuisance plus avoidance/adaptation.** Emissions of greenhouse gases, pollutants and noise may damage human health, the natural environment, buildings and equipment as well as give rise to nuisance. Accidents are another possible source of social costs (off-site risks). Finally, costs are sometimes incurred in trying to avoid or minimise the damage caused by pollution. Governments may, for example, decide to impose zoning restrictions on land that is subject to excessive noise or off-site risks. These costs can be categorised as avoidance costs of adaptation costs.
– **Costs of abatement and prevention measures.** For some environmental effects, general (environmental) quality criteria may be laid down in the political decision-making process, i.e. across-the-board emission standards for all sectors of society. Extra emissions occurring under this kind of regime do *not* lead to extra environmental damage, but imply, rather, that somewhere in society additional emission abatement measures are required. Such measures to compensate for emissions are once again associated with social costs.

Transaction costs, the costs of planning and monitoring the process, play a frequently forgotten but nevertheless often decisive role in the decision-making process.

Figure 12 shows first, as a function of total emissions, the social cost of one extra unit of emission – the cost of health damage due to toxic emissions, for example. The second curve represents the cost of one additional unit of emission *reduction*, which also comes with a price tag. However, the costs associated with emission reduction are not paid by society as a whole, but by the sector where the scope for effective action lies. This action may take the form of technological measures, operational measures or volume measures. The further emissions are reduced, the greater the costs of additional reduction, assuming that the cheapest measures are implemented first. If little emission abatement action has already been taken, an extra unit emission can be reduced at relatively low cost. If a wide range of measures are already in place, however, and technological options have been exhausted, there comes a time when even profitable activities will have to be stopped in order to achieve a little extra emission reduction.

Figure 12 Costs to society (upward curve) and to 'polluters' (downward curve) of one extra unit of emission



From the figure the following conclusions can be drawn:

1 Theoretically there is a social optimum, at a certain emission level, represented by the intersection of the two curves. If sectors reduce their emissions by more than this optimum, they will be implementing abatement measures that cost them more than the benefits accruing to society in the form of reduced nuisance, say. If emissions are reduced by *less* than the social optimum, the opposite holds. Thus, the optimum consists neither in zero emissions nor in unrestricted emissions.
2 The social optimum is associated with a 'price' per unit emission. It is unwise to implement abatement measures costing more than this price, and equally unwise to reject abatement measures that are cheaper. The optimum therefore represents a situation in which only the cheapest measures required for achieving the optimum are implemented.

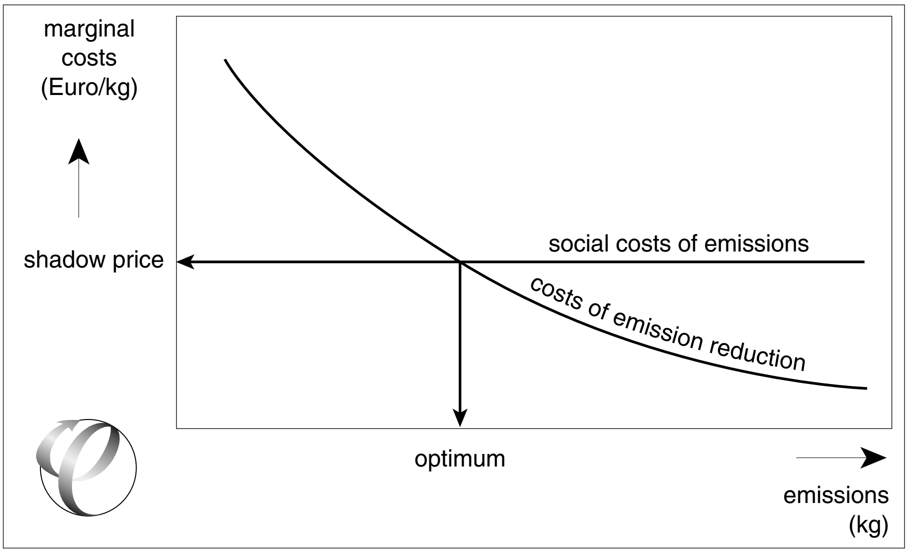
G.2 Valuation methods for environmental effects

The next question is how to assign a suitable price to the environmental effects. Different valuation methods may be applied, depending on whether or not environmental standards are in place for the specific impact concerned. These methods will be discussed in the following sections.

G.2.1 Valuation methods for damage, nuisance and avoidance

If there are no across-the-board emission reduction targets (see Section G.2.2) in place for the pollutant in question, it is the costs of damage, nuisance and avoidance (often in the form of indirect land use) that determine the social costs of emissions. Several methods are available for estimating these costs. In itself, however, this knowledge is not sufficient for calculating a shadow price, which also requires a knowledge of the curve representing the prevention costs incurred by the emitters (see Figure 12). The simplifying assumption is often made that the total costs of emission damage are proportional to the emission level or, in other words, that the so-called marginal costs remain constant. How reasonable this assumption is will depend on the external effect in question. The advantage of the assumption is that it enables valuation to be undertaken in a single step (see Figure 13).

Figure 13 Assuming constant marginal damage costs for ease of valuation



The following methods are available for calculating the social costs of damage, nuisance and avoidance[10].

Direct damage cost estimates
This method seeks to make a direct valuation of the damage arising from a given activity, as illustrated by a few examples. A value can be assigned to air pollution damage to agriculture and forestry by valuing the ensuing crop losses. In the case of accidents, an estimate can be made of the victims' lost productive output and medical expenditure. Air pollution damage to buildings can be estimated on the basis of repair costs. From a fundamental viewpoint, this method is undoubtedly the best: if actual damage can be perfectly assessed and valued, this method is superior to the others, each of which has at least one fundamental drawback. At the same time, however, more practical considerations often make application of other methodologies unavoidable.

The first practical drawback of this method is that mostly dose-effect relationships cannot generally be established for each and every material consequence. The main reasons are lack of measurement data and statistical problems. There may even be as yet unidentified forms of damage and the method will therefore often leave many items unvalued, as 'items pending', thus providing merely a minimum estimate of lost welfare. Secondly, it is often virtually impossible to value immaterial damage. Damage to nature and biodiversity, as well as psychological damage (in the case of noise and accidents), are notoriously difficult to assess.

[10] The following discussion is based on interpretation of numerous reports, including Schipper (1999), ECMT (1998), Infras/IWW (2000), CE (1994, 1999) and MuConsult (1999).

Willingness to pay / Willingness to accept via surveys
A second approach is to use 'stated preference' (SP) surveys to establish how much people are prepared to pay to avoid damages ('willingness to pay', WTP) or the compensation they desire to accept damages ('willingness to accept', WTA). One of the strengths of this method is the fact that it covers immaterial as well as material damages. Besides several practical weaknesses (respondents providing 'strategic' answers, major influence of type of question asked), it also has two more fundamental weaknesses:

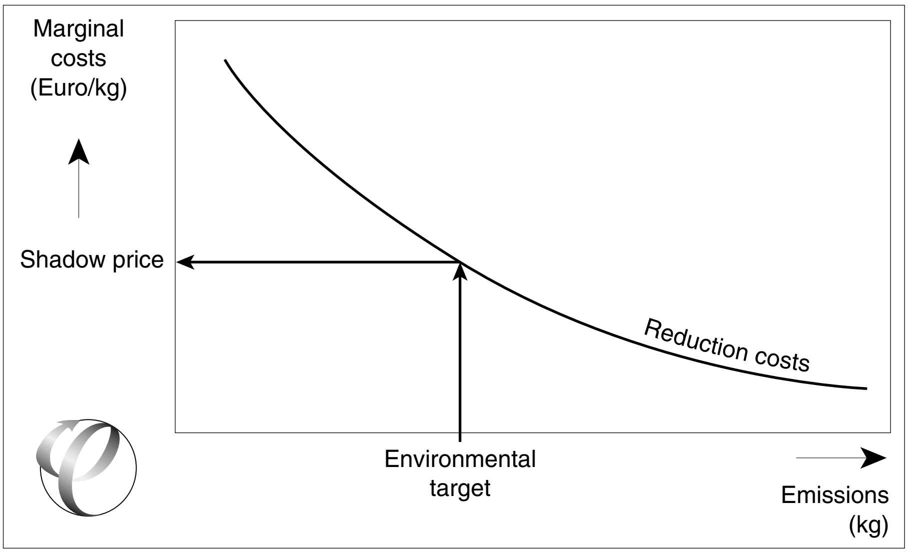
– It is extremely debatable whether respondents are capable of assigning a meaningful value to external effects, as is obvious from the example of global warming and even becoming apparent for (the health effects of) noise. While the method is useful for valuing local effects ('quality of life'), therefore, it is in principle less suitable for global and regional environmental problems.
– The method is usually applied to small groups of respondents who generally seem to be those most concerned about the problem being surveyed. However, the welfare of other people may also often be affected indirectly by the external effect. For example, while aircraft noise is of direct influence on the welfare of local residents, restrictions on land use as well as the noise itself will inhibit people outside the directly affected area from choosing an optimum housing location and raise property prices in unaffected areas.

Willingness to pay / Willingness to accept via changes in market prices
In this 'hedonic pricing' or 'revealed preference' (RP) approach a cost is assigned to external effects on the basis of their observed (revealed) impact on market prices, as when noise and air pollution cause rent and property prices to fall. This method has one fundamental drawback: its limited scope. The potential damage caused by the greenhouse effect, for example, will not be reflected in property prices. Where appropriate, though, this method is probably superior to the survey approach for WTP/WTA, since 'revealed preferences' (i.e. as reflected in market prices) appear to be a more reliable yardstick than 'stated preferences'. There remain several practical obstacles in the statistical assessment and isolation of variables, however.

G.2.2 The prevention cost method

For certain environmental impacts, across-the-board targets for environmental burden are in place for all sectors of society. In these cases, society has weighted – explicitly or implicitly – the costs and benefits of abatement measures. The price of emissions will then be formed by the marginal costs of reducing the impact to the overall target level. If one assumes that society will apply the cheapest measures first to achieve the targets, then an extra unit emission will make it necessary to apply an extra abatement measure of which the costs are equal to the shadow price. This method therefore requires greater knowledge of the shape of the reduction cost curve, i.e. the costs of the abatement measures involved (Figure 14).

Figure 14 Obtaining a shadow price from environmental targets



An important discussion that often arises when the prevention cost methodology is used is whether the across-the-board emission reduction target is ‘correct’. Some people may argue that the target is too strict (too far to the left of the graph), others that it is too lax (too far to the right). They have different perceptions of environmental damage and risks, on the one hand, and the economic damage and risks involved in setting different targets, on the other. In effect, the first category would like laxer policies and the second stricter policies.

We feel that this report – which only assesses emissions from pipeline transport, is not the appropriate place to discuss the correctness of international across-the-board emission reduction targets that have been agreed in a political process. The aim of this report is to establish the costs that arise when the pipeline transport causes one extra kg of emissions. If there are across-the-board reduction targets in place, these costs are given by the costs of reducing one kg of emissions somewhere in the economy.

In particular, we would mention a few reasons for not using so-called ‘scientific’ or ‘sustainability’ targets when governments have agreed on official targets:

1 Using targets that differ from those politically implemented would lead to inconsistencies in government policy. It is doubtful whether a sectoral study is the appropriate platform for questioning policies at a higher level, such as across-the-board emission standards for all sectors of society.
2 Setting a price tag on emissions on the basis of a target different from that holding for the rest of society would lead to inefficiencies. If a more ambitious target were set, it would lead to the sector at stake implementing measures that reduce emissions at higher cost than would have been incurred by other economic sectors. With a less stringent target the opposite would occur.
3 It is highly debatable whether targets can be formulated on a scientific basis alone. Science may be able to indicate the emission levels at which damage and risks become small. However, in virtually all cases, one must weigh the costs of risk reduction against the remaining risks. In some cases, such as global warming, there is also uncertainty involved.

There will never be zero risk and there is no clear-cut point at which risks become negligible, tolerable or acceptable, none of which concepts belong in the realm of the natural sciences, but rather require political or normative judgement.

4 If normative judgements are a necessary part of policy target formulation, the democratic decision-making process seems to be the most qualified arena for setting those targets. It will be clear that such issues as the asymmetric influence of certain lobby groups and lack of democratic legitimacy of parties at the international negotiating table may disturb this arena. Still, in the framework of the present study it cannot be judged *a priori* to which side of society's preferences the outcome of such negotiations will tend. Besides, as already stated, the aim of this study is not to contribute to the debate on across-the-board emission standards for all sectors of society, as these are already in place.

Finally, a few practical problems associated with the prevention cost method should be mentioned which should not be overlooked. This is because the establishment of marginal prevention costs requires the shape of the reduction cost curve to be known, as an *ex ante* assessment of possible future measures.

This gives rise to the following problems:

- Costs are often overestimated because the *dynamics* of technology development are underestimated. Only measures identified at the time of establishing the cost curve are taken into account, with new solutions unforeseen.
- Costs are also often overestimated because in many studies only *technological* options to reduce emissions are considered. If behavioural (operational) changes and volume changes are included in the cost curves, the marginal prevention costs will obviously fall.
- On the other hand, costs are often underestimated because prevention cost curves assume measures to be applied in order of cost-effectiveness. In other words, they assume that a perfect market exists for emission reduction. In reality, the market for emission reduction is often far from perfect, as all kinds of regulations and agreements currently in place hamper actual reduction of emissions across all sectors.
- Costs are also often underestimated because transaction costs and comfort costs are often ignored or overlooked. An example of transaction costs is the cost of incomplete information. An example of the existence of comfort costs is the fact that many people do not choose to drive a very fuel-efficient car, although doing so would save them a considerable sum of money.

Finally, we note that damage and prevention costs may not be added to arrive at a ‘final’, 'net' result. The two approaches are complementary, stem from different valuation philosophies and have their own specific pros and cons. If the reader's aim is to obtain an impression of the actual damage arising from one extra tonne of emissions, in the context of negotiations on optimum emission reduction targets, for example, they should use the damage cost approach. If the reader is convinced that one extra tonne of emissions in one place will not lead to extra damage because this will be mitigated by emission reductions elsewhere, they should use the prevention cost approach.

G.2.3 Summary

If NO across-the-board emission reduction targets exist, the most satisfactory approach to environmental valuation is direct valuation of damage, nuisance and avoidance costs. This can be done by establishing dose-response relationships for all relevant effects and valuing each of them individually. If enough data are available to value at least some of the effects, this approach can be used to obtain a good minimum estimate of costs. Indirect valuation methods, such as stated and revealed preference methods, can be applied in cases where environmental effects have a direct and local character, but due note should be taken of their drawbacks. In cases where the environmental effects are long-term and regional or even global in character, stated and revealed preference methods do not seem satisfactory because either too much knowledge is required on the part of respondents (stated preference) or clear relationships with real-market prices are lacking (revealed preference).

If broadly agreed across-the-board emission reduction targets DO exist, the prevention cost method can be applied. This is because in this case the cost of an extra unit of emissions at one location is NOT determined by the damage due to these extra emissions, but by the marginal costs of measures to reduce the same emissions elsewhere. As the costs of measures are all that count here, the debate on whether or not targets are 'correct' (i.e. set at or near the social optimum) is not relevant in this approach. Besides, this report is not the place to discuss the correctness of across-the-board emission reduction targets that have been politically agreed. The most important advantage of this approach is its consistency with politically agreed, across-the-board environmental policies. Its greatest disadvantage is that many people consider these targets either too strict or too lax. Besides, the prevention cost method has several practical drawbacks that makes actual estimation of the costs of measures harder than it may seem here.

The major findings are summarised in Table 35.

Table 35 Principal pros and cons of different approaches to valuing environmental effects

cost category	damage/nuisance + avoidance/adaptation cost approaches			prevention / abatement cost approach
subcategory	direct damage costs (dose-response)	stated preference (SP), CVM, WTP/WTA	revealed preference (RP); hedonic pricing	
main advantage	theoretically satisfying	good at non-material damage	within its scope better than stated preference	consistent with reduction targets defined
fundamental drawback	none	lack of knowledge about effects	limited scope	reduction targets may be ‘wrong’
		limited population		
practical drawback	dose-response relationships for all effects	strategic answers	statistical analysis	dynamics of technological development
	valuation of non-material damage	importance of question type		assumption of perfect markets
				transaction and comfort costs
application recommended	when adequate damage and valuation data are available	for short-term and local effects	for short-term and local effects	for regional/global effects, with agreed reduction targets
		when non-material damages are substantial	when damages are mainly material	

Source CE interpretation of international literature.

Abbreviations:

CVM Contingent Valuation Method

WTP/A Willingness to Pay / Accept

H Financial valuation of CO_2-emissions

H.1 Methods for the valuation of CO_2

The methods for determining the price of CO_2 (per kilogram), are the prevention cost method and the damage cost method.

Below we will briefly describe these two methods and the most important determinants of the differences between these methods. This knowledge is useful when analysing the literature. After that we will judge both methods.

Prevention cost method
The prevention cost method is based on the costs that must be made to reach a predetermined goal. We distinguish two variants:
– one at which an emission reduction goal is enforced to a certain sector ('closed system');
– in a second possible variant specific sectors will be free to trade emissions according to the mechanisms of the Kyoto Protocol.

In the prevention cost method, the most important variables determining the final shadow price are:
1 The reduction goal to be achieved.
2 The degrees of freedom in trade: is trade possible between Annex 1 countries or even world-wide?
3 The degrees of freedom in the use of 'flexible mechanisms' like emission trade, the Clean Development Mechanism and Joint Implementation.

Damage cost method
Besides the prevention cost method, the literature also pays much attention to the damage cost method. In this method it is tried to establish the regional consequences of climate change, mainly higher water levels and shifts in climatic zones. These changes in the ecosystem damage the economy.
The differences in literature sources that use this approach are mainly dependent on differences in dose-response relationships. Also discount rates play a large role, as damages will most occur in the future. Recalculating damages to net present values implies use of an interest rate reflecting societal preferences of time. This is illustrated in Table 36.

Table 36 Sensitivity of damage costs estimates of CO_2 for interests rates (IPCC 1996)

discount rate	**CO_2 shadow price (in € 1999 per tonne)**	
	low	high
2%	14	33
5%	1.4	3.3

Some other studies that use the damage costs method to value the damage of CO_2-emissions Nordhaus (1991, 1993) and Fankhauser (1994). Nordhaus calculates in his studies costs of about € 2.7/tonne CO_2, Fankhauser arrives at € 7 - 9/tonne CO_2.

In this study we will only use the prevention cost method to establish a CO_2 price, and we will base our estimates on the Kyoto shadow price, for reasons that have been explained in the main text.

Conversion rates used

In many cases we didn't copy the exact results from the respective sources for the following two reasons:

1 In some cases the results are given in the reduction of one tonne C and in other cases in the reduction of one tonne CO_2; we have decided to present all numbers in prices per avoided tonnes of CO_2. We have multiplied the prices of C with 12/44 where necessary, for the reduction of one tonne C equals the reduction of 44/12 tonne of CO_2.

2 In some cases the results are given in € and in some cases in US $. The basic year for the different data also varies. We've decided to convert all values to €1999. We have used the following conversion table.

Table 37 Conversion rates from $ to €

year	CPI (US, 1989 = 100)	Exchange rates (1 € = .. US$)
1990	105.4	1.40
1991	109.8	1.30
1992	113.1	1.44
1993	116.5	1.19
1994	119.5	1.25
1995	122.9	1.32
1996	126.5	1.28
1997	129.4	1.11
1998	131.4	1.19
1999	134.3	1.07

H.2 Summary of results from prevention cost method

This paragraph presents the CO_2-emission reduction costs found in the literature. A complete review follows later on in this annex.

The ranges of values we've found are presented into four variants:

1 First the variants where the different regions must reach their goal in their own region without trade between the regions.

2 Then the variants where international emission-trading is permitted between Annex I countries.

3 Next a variants where global emission trading is permitted, in other words the maximal variant of CDM.

4 We'll finish with a few examples of values where sinks are permitted, other greenhouse gasses can be reduced or explicitly not, agreement on double-bubble, etc.

H.2.1 Every region it's own

At first we'll give the ranges for the different regions distinguished in the models. Hereby we present the range in the case where the extreme values are being ignored and, between brackets, the whole range.

It further concerns the costs involved for reaching the Kyoto-goals for every region when all reductions must be made in own country.

Table 38 Every region it's own

Region	Marginal reduction cost (in € 1999 per tonne CO_2)
US	25 – 78 (17 – 105)
EU	40 – 83 (29 – 216)
Japan	29 – 177 (22 – 209)

Sources:

- for the US: 9 literature sources;
- for the EU: 8 literature sources;
- for Japan: 8 literature sources.

This table shows that in all probability the US can reach their goal in their own country in the cheapest way. This is because of the relatively energy-inefficient structure of the American economy, where with the help of energy-savings and 'good-housekeeping' a lot of win-win measures can be taken. Europe is already in a further stage of efficiency-increasing measures, which makes it more expensive to take further measures.

H.2.2 Emission trade between Annex-I countries

When we study the price per avoided tonne CO_2 when emission trading between Annex I countries is permitted, we find the following range of values:

€ 15 – 35 (10 – 49)

This range is based on the results of 10 literature sources.

In this scenario Joint Implementation is permitted, but the Clean Development Mechanism is prohibited.

H.2.3 Global emission trading

In the variant where global emission trading takes place to minimise the total costs to reach the Kyoto-goals, more cheap measurements come available resulting in a lower price.

In this situation there has been assumed that in all models the countries not belonging to Annex I will have emission rights for the forecasted emissions of that country in 2010. This results in an emission ceiling leading to a real market. This variant can be seen as a upper-limit of the opportunities of the CMG-model.

The ranges of values found are (between bracelets is the range without extreme values:

€ 6 – 8 (4.8 – 17)

There were only 4 sources of literature presenting these results.

H.2.4 Some other variants

In the literature analysed we encountered sources which have assessed some extra variants. Below in brief the characteristics with corresponding values.

Table 39 Results sensitive for assumptions

Characteristics	Development shadow price	Reference
Annex I trade + counting all the sinks	22 -> 7	Annex I trade
Annex I trade + counting halve of the sinks	22 -> 14	Annex I trade
Annex I trade + infinite high costs for reduction CO_2 gasses	22 -> 29	Annex I trade
Double-bubble	17 -> 9 (US, Jap. en Austr.) 17 -> 74 (rest OECD)	Annex I trade

Each of these variants was presented by only one source of literature.

Next to these variants model calculations have been made at which the goals of Kyoto have been extrapolated to 2020. We've presented the differences in the prices per avoided tonne CO_2 in Table 40.

Table 40 Kyoto targets also apply to 2020

Source	Prices in 2010 en 2020	Characteristic
McKibben et al. (1999)	17 -> 31	Annex I trade
McKibben et al. (1999)	6 -> 10	Global emission trade
MacCracken et al. (1999)	22 -> 36	Annex I trade

The last two tables show that:
- fully counting of sinks lowers the price of CO_2 with two thirds;
- counting halve the sinks lowers the price of CO_2 with one third;
- infinite high costs for not-CO_2 gasses raise the price of greenhouse gasses with almost one third;
- the effect in implementation of double-bubble differs greatly between the 'bubbles';
- the extrapolation of the Kyoto-goals to 2020 causes higher reduction costs, approximately 60% per tonne avoided.

H.3 Literature studied

The separate sources of literature that are found and analysed are presented below.

Capros, P., en L. Mantzos, 2000, The economic effects of EU-wide industry-level emission trading to reduce greenhouse gasses: results from PRIMES energy systems model, National Technical University of Athens.

This study describes the results of model exercises with the PRIMES-model, a partial balance model aimed at the energy markets within the European Union.

Five scenario's to reach the Kyoto-goals within the European Union are being dealt with. This study shows clearly the cost advantages of trading that can be reached.
The five scenario's are:
- every member state reaches his own goal, without trading;
- every sector within a Member State reaches its reduction as is determined for every member state;

– ever member state reaches his own goal where trading between energy producers is permitted;
– ever member state reaches his own goal where trading between energy producers and the energy intensive industries is permitted;
– the European Union reaches the goal, where trading between all sectors in all members states is permitted.

The costs to reach the goals of the EU vary greatly between the different scenario's. Table 41 shows a overview of the scenario's and the marginal reduction costs of the last tonne CO_2 needed to reach the goal.

Table 41 Estimated price per avoided tonne CO_2

Scenario	Marginal reduction cost (in $ 1995 per tonne CO_2)
(I) Every sector within the member state same target as member state	108
(ii) Every member state has a target.	46
(iii) Trade between energy producers	39
(iv) Trade between energy producers and energy intensive sectors.	37
(v) Free trade within the EU	28

This shows that the Kyoto goal of the EU can be reached at relatively low costs if a EU internal emission trading will be set up.

An important assumption in this modelling is that the transaction costs of a emission trading system are set to zero.

CPB/RIVM, 2000, De economische gevolgen van het Kyoto-protocol voor sectoren en wereldregio's (Economic consequences of the Kyoto protocol for setors and world regions), no. 00/31, Den Haag.

In this paper the investigators have performed model calculations with the model WorldScan. Goal of this paper was to map especially the economic consequences of the Kyoto protocol, focused especially on the consequences for energy exporting countries and developing countries.
WorldScan is a global general balance model, primarily to describe long term developments. The quotes about the developments in the period 2008-2012 must therefor be carefully interpreted.

The simulations are confined only to CO_2 greenhouse gas and the basic variant is given by the individual reaching of the different goals through the different countries.

The possible cost lowering mechanisms as Clean Development Mechanism, Joint Implementation and the usage of sinks can't be simulated in WorldScan.

Emission trading (between Annex I countries) can be simulated and serves as an alternative variant. In this paper there are no trading limitation simulated though, so in the alternative variant the emission reduction goal can be reached fully by trade between other Annex I countries.

The results of the (two) simulated situations are summarised as follows.

Table 42 Results of the (two) simulated situations

	Marginal reduction costs (in € 1999 per tonne CO_2)	
Region	**Without emission trade**	**With emission trade**
VS	40	15
Japan	29	15
Pacific OECD	32	15
EU	52	15
Eastern Europe	3	15
Former Soviet Union	0	15

ECN/RIVM, 1998, Optiedocument voor emissiereductie van broeikasgassen: inventarisatie in het kader van de Uitvoeringsnota Klimaatbeleid. (option document for GHG emission reduction; inventory in the framework of the Climate Change Execution Paper).

In this publication a overview is presented of the possibilities to reach the emission reduction goal of the Netherlands domestically. An analysis of the results shows that the measures that can and should be taken in the Netherlands (the so-called “basic package”) are not the cheapest measures.

A similar analysis is performed by Dings et al. (1999) and this shows that the most expensive measure in the basic package is unequal to the cheapest measure in the extra package. Nevertheless we choose to consider the most expensive measure of the basic package as the marginal costs of the last measure needed in the Netherlands to reach the Kyoto goal.

The costs are roughly € 70 per tonne CO_2. However, this price concerns only the domestic measures and can’t be used as a international price to reduce one tonne CO_2. It gives a good view of the possibilities to reach the Kyoto goals domestically.

ECN/AED/SEI, 1999, Potential and cost of Clean Development Mechanism options in the energy sector: inventory of options in non-Annex I countries to reduce GHG emissions.

This publication gives a estimation of the possibilities to reach cost savings by the CDM. The table below presents the outcomes of a simple simulation, where a perfect competition market is assumed.

This publication describes a trading system within the OECD, a system where trading between Annex I countries is permitted and a global trading system. This variant can be seen as the extreme variant of CDM.

This resulted in the following outcomes.

Table 43 Results of the simulation

Trade within	**Marginal reduction costs (in € 1999 per tonne CO_2)**
OECD	68
Annex I	21 – 35
Global	4.8 – 18

It has to be noticed that the lower prices in the range will approach the reality the closest. The lower prices will be the result if the so-called ‘no regret’ measures will count for reaching the Kyoto goals. The ‘no regret’ measures are the measure which will be economic profitable even without strict climate policy.
This separation, between profitable and not-profitable, has been made explicit in this publication.

McCracken, C.N., J.A. Edmonds, S.H. Kim and R.D. Sands, 1999, The economics of the Kyoto-protocol, in: The Energy Journal: special issue, May 1999, p. 25 – 72.

With the so-called ‘Second Generation Model” the authors estimated the marginal costs needed to reach the Kyoto goals. These marginal costs represent the costs per tonne CO_2 of the last measure needed to reach the goals. It has been done for 5 scenario’s:

1 All region comply with their Kyoto-goal, no trading.
2 Trading is permitted between Annex I countries.
3 Trading is permitted between Annex I countries and CDM is permitted.
4 Not-CO_2 greenhouse gasses are taken into account.
5 ‘Sinks” are permitted in some degree.

Below the resulting prices per avoided tonne of CO_2 for 2010. Between brackets are the values resulting form the model for the year 2020, with the assumption that the Kyoto goals in 2020 are still effective.

Table 44 Resulting prices per avoided tonne CO_2 for 2010

		Marginal reduction costs (in € 1999 per tonne CO_2)				
region	**scenario**	**(i)**	**(ii)[b]**	**(iii)[c]**	**(iv)[d]**	**(v)[e]**
Australia		36 (43)	22 (36)	8 (-)	29 (-)	7 (-)
Europe		40 (63)				
US		51 (60)				
Canada		106 (117)				
Japan		139 (130)				

a When there’s no expansion of the nuclear power capacity the marginal reduction costs in Europe can reach up to € 44.

b If Eastern Europe will behave as a monopolist on the market of tradable emission rights, the trading price for this scenario will be higher, namely € 32.

c This price was achieved by allocating non-Annex 1 countries emissions in the reference scenario and subsequently apply global trade. A fictitious market is created, in which indeed scarcity of emission reduction is achieved.

d This price is based on the assumption that the not-CO_2-gasses only can be driven back against infinite high costs; when these gasses can be driven back for free, every region can reach their Kyoto goal without costs and the resulting market price for CO_2 will be zero. The price in the second scenario is based on the assumption that the not-CO_2-gasses can be driven back against the same proportional costs as CO_2 can be driven back.

e This price is based on the assumption that all sinks count for reaching the Kyoto-targets, while further trading between Annex I countries is permitted. When only halve of the sinks are counted, the trading price to $ 14.

Table 44 shows that the different assumptions of the filling-in of the Kyoto protocol and its mechanisms have an important influence on the costs the different regions have to make.

Trade between all countries to reach Kyoto targets gives *ceteris paribus* the lowest costs for reaching the goals, namely € 8 per tonne CO_2.

McKibbin,W., M. Ross, R. Shackleton and P. Wilcoxen, 1999, Emissions trading, capital flows and the Kyoto-protocol, in: The Energy Journal: special issue, May 1999, p. 287 – 334.

This publication describes the estimation of the costs for reaching the Kyoto targets with the help of the so-called G-Cubed model. This model describes measures and adjustments in several regions and sector in an inter-temporal equilibrium model.

In this study five scenarios are calculated:
1 Only the US fulfil the Kyoto goals.
2 All Annex I countries fulfil their Kyoto goals, trade is not permitted.
3 All Annex I countries fulfil their Kyoto goals, trade is permitted between Annex I countries.
4 All Annex I countries reach their Kyoto goals, trade is permitted within two trading blocks ‘other OECD’ and “other Annex I’, while there’s no trade permitted between trading blocks.
5 Global trade is permitted where the developing countries not appearing in the Kyoto protocol get their reference emissions assigned.

The model is not capable to consider the reduction of not-CO_2-emissions as well. This approach counts for more models treated in this annex and ignores the relatively cheap reduction measures of other greenhouse gasses.

This model proclaims a strict climate policy in 2000, so the economic actors have 10 years to anticipate on the policy and take action.
We present the resulting prices per avoided tonne CO_2 in Table 45 for 2010 and 2020 (in € 1999).

Table 45 Resulting prices per tonne of CO_2 avoided

<table>
<tr><th colspan="2"></th><th colspan="5">Marginal reduction cost per tonne CO_2 (in € 1999)</th></tr>
<tr><th>region</th><th>Scenario</th><th>(i)</th><th>(ii)</th><th>(iii)</th><th>(iv)</th><th>(v)</th></tr>
<tr><td colspan="2">Australia</td><td>-</td><td>50 (64)</td><td rowspan="4">17 (31)</td><td rowspan="3">9 (19)</td><td rowspan="4">6 (10)</td></tr>
<tr><td colspan="2">US</td><td>22 (27)</td><td>25[a] (29)</td></tr>
<tr><td colspan="2">Japan</td><td>-</td><td>32 (45)</td></tr>
<tr><td colspan="2">rest of OECD</td><td>-</td><td>73 (88)</td><td>74 (89)</td></tr>
</table>

a The difference between this price ($ 25) and the price of 23 in case of unilateral action by the US (scenario 1) can be explained as follows: when all countries have to reduce their CO_2-emissions demand for oil and thus its price will decrease. It will be harder then to achieve the US reduction targets.

b The difference between this price ($25) and the price of $23 in case of one-sided action by the US (scenario 1) can be explained as follows: If all countries must push back.

De price that will result from global trade is about 6 €/ton CO_2 in 2010.

Manne, A.S. and R. Richels, 1999, The Kyoto-protocol: a cost-effective strategy for meeting environmental objectives? in: The Energy Journal: special issue, May 1999, p. 1 – 24.

From this article it is hard to judge the assumptions made for modelling climate policy and the resulting costs.

Next prices are mentioned as marginal reduction costs in € 1999 per tonne CO_2:

1 € 78 in case of US unilateral domestic action.
2 € 33 in case of trade between Annex 1 countries and application of CDM.
3 € 22 in case of completely global trade.

The difference between variants (ii) and (iii) the CDM potential assumed; in variant (ii) the authors assume only 15% of total CDM potential can be exploited in practice. This reflects the complexity of CDM. In case of global trade, the full potential of CDM can be exploited.

PEW Center on Global Climate Change, 1999, International emissions trading and global climate change, Arlington, USA.

This report gives an overview of advantages of international emission trade for reducing GHG-emissions. The researchers assess differences between various models used to assess the effects of GHG-emission reduction.

Table 46 below offers an overview of marginal reduction costs as calculated by various models. Reduction costs birth in case of regional and Annex 1 trade are calculated.

Table 46 Difference between models

	Marginal reduction costs per ton CO_2 (in € 1999)									
	SGM		EPPA		GTEM		G-Cubed		OECD Green	
Region	No trade	Annex I trade	No trade	Annex I trade	No trade	Annex I trade	No trade	Annex I trade	No trade	Annex I trade
VS	51	22	56	49	105	35	17	10	44	19
Japan	139	22	177	49	209	35	71	10	22	19
Western Europe	44	22	83	49	216	35	47	10	58	19
Former Soviet Union	0	22	0	49	0	35	0	10	0	19

Differences are caused by factors as previously described in this annex.

PNNL, 1997, Return to 1990: The cost of mitigating United States carbon emissions in de post-2000 period (no. PNNL-11819).

This publication gives a brief overview of possibilities to reduce costs to achieve reduction targets. For fictive US targets are used instead of the Kyoto targets. We will not discuss the results in detail, also because the results from the Second Generation model used have already been described under MacCracken et al. (1999).

I Financial valuation of non-CO_2-emissions

I.1 Introduction

The aim of this annex is to provide an overview of financial valuations for emissions other than CO_2 in the recent international literature.

The emissions that we have incorporated in this survey are the following:
- NO_x (in itself and via ozone);
- PM_{10};
- $PM_{2.5}$;
- HC, volatile hydrocarbons;
- SO_2;
- CO.

In this paragraph, we present the literature sources we have found with their results. To the extent possible, we have also presented the main assumptions and important remarks.

We first present the overview of the findings in Section I.2, with the main conclusions we draw from them. In paragraph I.3 we then present the full survey. For some literature sources we had to make some additional calculations to arrive at a unit cost, i.e. a cost per kilogram pollutant. We have presented our own calculations in separate text boxes in order to keep the description of the sources as objective as possible.

The one modification we have done for each of the sources is in the currency, because different sources use different currencies and different base years for these currencies. To provide a consistent overview we present all figures in one currency, namely in $€_{1999}$. For the conversion of the different currencies we have used the following conversion table.

Table 47 Conversion factors from $ to €, CPI-numbers

Year	CPI (US, 1989 = 100)	CPI (EU, 1989 = 100)	Exchange rate (1 € = .. $US)[11]
1990	105.4	104.1	1.40
1991	109.8	108.4	1.30
1992	113.1	112.4	1.44
1993	116.5	116.0	1.19
1994	119.5	119.1	1.25
1995	122.9	121.8	1.32
1996	126.5	124.8	1.28
1997	129.4	126.8	1.11
1998	131.4	128.2	1.19
1999	134.3	129.6	1.07

In case the original numbers in the report are denoted in another currency, we have given the relevant exchange rate.

[11] This exchange rate is the end-of-year exchange rate.

I.2 Overview of findings

Qualitative conclusions

From the literature analyses, the following conclusions can be drawn:

- the knowledge about damage costs from other than greenhouse gas emissions has been much improved the last years. Especially on the area of health effects of transport pollutants much progress has been made. Dose-response relationships have been improved, dispersion models as well, and the valuation of (years of) life (lost) is subject to much less controversy;
- the increase in knowledge on these health effects has led to increasing valuations of practically all emissions, lead to a better understanding of variations in valuations, and thus a lower spread of various results if the factors behind the variations are taken into account. For example, several studies show that in an area like the Paris inner city a gram of $PM_{2.5}$-emission leads to several Euro of health damage, and that in sparsely populated areas this is more something like 1 Euro cent. This shows that prices of emissions are very dynamic depending on the circumstances, and that with further scientific insight prices are more likely to increase further than to decrease;
- much of the health effects focus has been shifted to ultra-fine particles ($PM_{2.5}$). Extensive analysis in the framework of the ExternE programme and the WHO study of 1999 shows robust and significant dose-effect relationships. As a result, air pollution related costs from road transport, especially those of vehicles equipped with diesel engines, are dominated by the health effects of these particles;
- the most relevant health effects besides those of $PM_{2.5}$ come from nitrates and ozone;
- carbon monoxide, 1.3 butadiene, benzene, and benzo(a)pyrene, other pollutants being suspected in the past, seem not to give rise to significant health effects. Either exposure or human sensitivity is relatively low;
- it should be said, however, that possibilities to monetise values like biodiversity and the health of forests, still fall rather short compared to possibilities to value health effects;
- health *damage* costs alone already generally seem to be higher than *prevention* costs that are based on the marginal costs of achieving *politically* agreed targets like the NECs[12]. Due to this phenomenon, combined with the progress made on the valuation of health effects, the prevention cost methodology is becoming a less popular tool for emission valuation.

Quantitative conclusions per pollutant

In this paragraph we present the overview of estimates we have found. We present the results in five tables.

We first present in four tables overviews of the values found per emission (NO_X, $PM_{2.5}$, HC, and SO_2). For every emission, results from damage cost studies and prevention cost studies are distinguished. Furthermore, we try to explain ranges and we present differences between valuations for emissions emitted in urban areas and in rural areas.

In the fifth table the results are aggregated and averaged for use in this study.

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[12] Theoretically, marginal prevention costs that are necessary to achieve environmentally sustainability targets are equal to marginal damage costs in the optimum).

Damage costs

Recent (ExternE) insights come to damage cost estimates of 12 €/kg NO_X, which includes the damage of the ozone formed out of NO_X. This value is an average and varies between a presented range of 1.9–21 €/kg across the European countries in the study. The range can mainly be explained by differences in health impacts due to differences in exposed population.

The ExternE programme takes a wide range of impact categories into account:

- human health;
- crops;
- timber;
- building materials;
- ecological systems;
- non-timber benefits of forests.

Although the valuation of damage to ecological systems is uncertain, the resulting marginal damage cost per kg NO_X seems to cover most relevant impacts.

Furthermore and the valuation of mortality is quite high. The value of a statistical life, which is used throughout ExternE, is € 3.2 million. This implies that there is no distinction between a life lost, which would have otherwise been lost 1 day later or a life lost, which might otherwise have lasted for tens of years. Some people have therefor suggested to use the Value of Life Years Lost, which presents the discounted value of the expected amount of life years lost. If this valuation methodology were used, the average value presented in ExternE would be lower.

IIASA et al. (1999b) present damage costs as well, in which they distinguish estimates with the 'Value of a Statistical Life' methodology and the (lower) estimate with the 'Value of Life Years Lost' methodology. The estimate using the Value of Life Years Lost for mortality impacts is € 9, the other is 15 €/kg.

SIKA (1999) arrive at a marginal social cost of 9 €/kg NO_X as well for the Swedish case.

The last recent damage cost estimate for NO_X is provided by COWI (2000) and they make a distinction between damage in rural areas and in urban areas. They arrive at 11 €/kg NO_X in rural areas and 12 €/kg NO_X in urban areas.

Prevention cost

Recent work on the estimation of the prevention cost per kg of NO_x can be found in the studies, which were done by IIASA to calculate the costs of achieving the NECs (National Emission Ceilings). The NO_X ceiling implies a 55% reduction of NO_X-emissions in Europe in 2010, relative to 1990. Using this ceiling as a basis, IIASA arrives at a marginal social cost of reducing NO_X of 4.7 €/kg.

The reduction target is the most important factor determining the marginal cost in the prevention cost method. Ågren (1999) states that the National Emissions Ceilings, although more ambitious than the targets proposed in the so-called Gothenburg Protocol, still fall short of meeting the environmental targets as set in the Fifth Environmental Action Plan. Those targets are defined as the targets that need to be achieved in order to have no exceeding ever of the critical loads, for both human health and vulnerable biodiversity. In order to achieve those 'sustainability' targets, the prevention costs will most probably be higher than 4.7 €/kg.

Kågeson (1993) presents prevention costs for NO_x as well and he arrives at a marginal social cost of 4.8 € 4.8/kg. This marginal social cost is the result of calculating the cost of the last measure, which was needed to achieve a 50% reduction in NO_x-emissions in Europe in 2000, relative to 1985.

The level of NO_X-emissions did not change too much in Europe between 1985 and 1990, so we can conclude that the cost curves in Europe did not change too much either. Kågeson notes that the targets he used to calculate the marginal social costs needed to be seen as interim targets as well.

Total
The conclusion is that with respect to NO_X, the damage cost approach leads to higher marginal social costs than the prevention cost approach based on marginal costs to achieve politically established emission reduction targets. This suggests that reduction targets should be stricter in order to achieve maximum welfare. Therefore, we will base our final estimate of the NO_X-emission value on damage instead of prevention costs. We also differentiate for rural and urban effects.

Table 48 Overview of literature on the valuation of NO_X-emissions in €1999, including indirect damage via ozone

sources on **damage costs**	average	range	rural	urban	comment
ExternE (1999)	12	0.9-21			mainly depends on population density
ExternE transport (1999)		4-25	4-13	7-25	
IIASA (1999b)	12	9.4-15			depends on valuation of life lost
SIKA (1999)	9	7.7-10	7.4	7.7-10	Swedish case, depending on population
COWI (2000)	11		11	12	basis for estimate could not be found
IVM (1999)	4.4	0.6-32			Dutch case, only health impacts via nitrate and ozone
sources on **prevention costs**					
IIASA (1999a)	1.5-3.3				depending on scenario, targets probably not sustainable
IIASA (1999c)	4.7				
CE (2000)	5.5		5	7	based on Auto Oil standards
Kågeson	4.8				1985-2000 reduction targets

$PM_{2.5}$ / PM_{10}
Damage cost
Because the most important determining factor of PM_{10} is human health we only deal with the damage cost estimates. These damage costs crucially depend on the amount of people living in a certain area. Two sources are the most relevant for this study:
- the ExternE projects with its numerous spin-off reports;
- the WHO (1999) study used by INfras/IWW (2000) as this gives new information about the dose-response relationships.

In ExternE, a practical approximation formula has been derived: the damage cost of $PM_{2.5}$ per kg is about equal to 10 + 122*population density (in 1,000 people per km^2). One should, however, take care that transport is linked to human activity, and that therefore most transport emissions are released in areas that are more densely populated than the national average. For example in the Netherlands with its 450 inhabitants per km^2 the damage costs are higher than 10+122*0.45 = 65. For example, IVM (1999) comes, on the basis of the ExternE approach, to 130 €/kg, whereas Infras/IWW (2000) comes to 174 €/kg. In the Paris city centre, the health costs of a kg of $PM_{2.5}$ even amount to several thousand Euro.

As the relevant impact of $PM_{2.5}$-emission is human mortality and morbidity, and as scientific knowledge about the damage of PM_{10}-emission has been greatly improved, and dose-response relationships seem to be well-established, the prevention cost approach seems not suitable any more for the valuation of this emission.

Table 49 Overview of literature on the valuation of $PM_{2,5}$ or PM_{10}-emissions in €1999

source on damage costs	average	rural	urban	comment
Infras/IWW (2000)	73-194			national averages across EU, based on WHO study
ExternE transport (1999)		18-200	200-2000	depends mainly on population density, high value = Paris, low = Dutch average density
SIKA (1999)			85-915	Swedish case, high value = Stockholm centre
COWI (2000)		24	90	basis for estimate could not be identified
IVM (1999)	130	18-150	200-942	$PM_{2.5}$ from 'low source' (transport), Dutch case

* practically all transport PM-emissions fall in the range of smaller than 2.5 micron; therefore the '2.5' estimates seem to fit best the transport emission cost estimates.

VOC/HC

Damage cost

For VOC/HC there exist not too many recent estimates. ExternE leads to estimates of 4-9 €/kg. The higher estimates apply for cities like Stuttgart and Barnsley. For the Paris city centre the value explodes to 33 €/kg. SIKA (1999) presents for the Swedish case the same range of values many to take urban effects into account: € 4-9. COWI (2000) presents a value of 2.7 €/kg.

Prevention cost

IIASA (1999c) calculates the marginal social cost of a kilogramme, but this modelling is not too sophisticated, because most measures that reduce VOC/HC, also reduce NO_x. Therefor, in general all costs are allocated to either one of the pollutants. This results in almost identical prevention costs for VOC/HC as for NO_x. The value IIASA (1999c) presents is € 4.6 per kilogramme.

Total

From the different estimates it seems best to use the value of € 4 as the marginal social cost per kilogramme. The COWI estimate is lower than the other two, and also Bleijenberg et al. (1994) presented an estimate of € 5.

Table 50 Overview of literature on the valuation of HC-emissions in €1999

sources on damage costs	average	range	rural	urban	comment
ExternE transport (1999)		3.9-33	4	4-33	depends mainly on population density, high value = Paris
SIKA (1999)		3.6-8.9	3.6	4.1-8.9	Swedish case, depending on population density, 8.9 = Stockholm centre
COWI (2000)	2.7		2.7	2.7	basis for estimate not clear
sources on prevention costs					
IIASA (1999a)	1.5-3.3				depending on scenario, targets probably not sustainable
IIASA (1999c)	4.6				
CE (2000)	5.5		5	7	based on Auto Oil standards

SO_2

Damage cost

Recent (ExternE) insights come to damage cost estimates of 8.5 €/kg SO_2. This value is an average and varies widely across the European countries in the study. The presented range is € 1.5-15.5.

The resulting marginal damage cost per kilogram SO_2 seems to cover all relevant impacts. However, the damage to ecological systems is uncertain.

Other damage estimates come from IIASA (1999b), which presents € 3.5 per kilogram, and Kågeson (2000) who presents a value of € 3.3 as an absolute minimum. The recent COWI-study (2000) calculates values for rural areas (€ 5.5) and urban areas (€ 9.5).

Altogether, it seems that the ExternE-value in general is too high and from the other studies we conclude that the value from Kågeson (2000) and IIASA (1999) can be best used as the lower bound.

Prevention costs

Recent work on the estimation of the prevention cost per kg of SO_2 can again be found in the studies, which were done by IIASA to calculate the costs of achieving the NECs.

The estimate for marginal social cost of a kg of SO_2 which we could derive from IIASA (1999c) was 1.5 €/kg. This value is based upon the target set in the National Emissions Ceilings. This target boils down to a 78% reduction of SO_2-emissions in Europe in 2010, relative to 1990.

It is important to note that this value seems very low, compared to the damage cost estimates. An important factor determining the marginal cost using the prevention cost method is the target. About this target Ågren (1999) makes the following remark: the National Emissions Ceilings are more ambitious than the targets proposed in the so-called Gothenburg Protocol, but they still fall short of meeting the environmental targets, set in the Fifth Environmental Action Plan. Those targets are defined as the targets that need to be achieved in order to have no exceeding ever of the critical loads, for both human health and vulnerable biodiversity.

In order to achieve those 'sustainability' targets, the prevention costs will most probably be higher than 1.5 €/kg. Kågeson (1993) presents prevention costs for SO_2 as well and he arrived at a marginal social cost of € 1.6 per

kilogramme. This marginal social cost is the result of calculating the cost of the last measure, which was needed to achieve a 60% reduction in SO_2-emissions in Europe in 2000, relative to 1985.
However, Kågeson (1993) also calculated the marginal social cost of a reduction of 80% in 2000 relative to 1985. The value he found there was € 3.2 which is substantially higher, whereas this target still cannot be seen as a sustainable level of SO_2-emissions.

Total
When we compare the results from damage cost studies and prevention cost studies, the gap is fairly small. Both the damage cost estimates from IIASA (1999b) and SIKA (1999) can serve as a lower bound, which is € 3 per kilogramme. This value is quite similar to the highest prevention cost estimate.

Table 51 Overview of literature on the valuation of SO_2-emissions in €1999 per kg

sources on **damage costs**	average	range	rural	urban	comment
ExternE (1999)	8.5	1.3-16			variation across EU Member States
ExternE transport (1999)			6.8-8.5	10-50	mainly depends on population density
IIASA (1999b)	3.5				depends on valuation of life lost
SIKA (1999)	3.3		3.3		Swedish case, minimum estimate
COWI (2000)	7		5.5	9.5	basis for estimate could not be identified
sources on **prevention costs**					
IIASA (1999a)	1.2				variations between countries, targets probably not sustainable
IIASA (1999c)	1.5	0-5			
CE (2000)	3		3	3	based on Auto Oil standards
Kågeson (1993)	1.6-3.2				depending on reduction targets

Table 52 Overview of middle estimates from the recent European literature for the valuation of NO_X, PM_{10}, HC and SO_2, per kilogram emitted, based on damage costs

	average	urban	rural
NO_X	9	12	7
PM_{10} / $PM_{2.5}$	150	300	70
HC	4	6	3
SO_2	6	10	4

I.3 Full servey of literature

The following literature has been found on the valuation of emissions other than CO_2. For each source we shortly describe the method that is used, and the assumptions that are made. Finally the results are presented.

Infras/IWW, 2000, External costs of transport: accident, environmental and congestion costs in Western Europe, UIC, Zürich/Karlsruhe/Paris

Method: damage cost.

The impacts that are distinguished are the following:

- human health;
- materials and buildings;
- agricultural crop losses;
- forest damages[13].

Health: the method is based on WHO (1999), based on PM_{10} as the leading indicator and a value of statistical life for people affected by air pollution of € 0.9 million. The results from WHO for Austria, France and Switzerland were extrapolated by Infras/IWW by using the weighted PM_{10} and NO_X-emissions in different countries. This is done as follows.

Infras/IWW extrapolated the health impacts found by WHO (1999) (PM_{10} as leading indicator, countries Austria, France and Switzerland) to the EU Member States. As for other countries data on PM_{10} concentrations are not widely available Infras/IWW have followed an indirect approach. As NO_X-emissions in all EU Member States are well known, they defined a correlation between PM_{10}-concentrations and PM_{10} and NO_X-emissions in France, Austria and Switzerland, and use this correlation to establish PM_{10}-concentrations for the other European countries considered. A correction for non-exhaust PM_{10}-emissions was necessary in order to properly fulfil this task.

[Addition by CE: dividing the health costs by transport particulate emission estimates leads to an approximate health costs of approximately 100 € per kg of particulate emitted (urban/rural average for France, Austria and Switzerland). An important factor behind the health impact of PM_{10} emitted is population density; this amounts 107, 96 and 172, for France, Austria and Switzerland respectively. As a first order estimate, one can put a population density correction factor on the PM_{10} shadow prices, as exposure per unit of emission is approximately linearly dependent on population density]

The health costs account for an average 81% of external costs from air pollution in the countries under consideration.

Crop losses: the costs that were computed for Switzerland (Infras/Econcept/Prognos, 1996) are used to calculate the same costs for other European countries. The formula that is used is as follows:

Crop losses = α * (NO_X-emissions/country area) * agricultural production
with $\alpha = 0.0037$ [m^2/ton]

On average these costs amount to 1% of external costs from air pollution in the considered countries.

Building damages: the methodology used to calculate these costs is similar to the one used for crop losses. The costs computed in Infras/Econcept/ Prognos (1996) were scaled to other European countries using NO_X-exposure levels and building surface. The exposition levels are estimated by dividing the emissions by the country area and the building surface is estimated using population. The following formula results:

Building damage = β * (NO_x-emissions/country area) * building surface * PPP
with $\beta = 0.322$ [€/tonne].

[13] This last category is only included in the sensitivity analysis.

On average these costs account for 18% of external costs from air pollution in the considered countries.

Addition by CE: using the data on emissions as provided in the Infras/IWW report for the EU-countries, we have calculated the average cost per kilogram PM_{10} for the EU-countries. The average cost is equal to the marginal cost, because the dose-response functions are linear: at a certain location, each kilogram is assumed to have the same impact. This resulted in Table 53.

Table 53 Overview of average and marginal damage costs per kg of PM_{10}-emission

Country	**Marginal social cost (in €$_{1999}$) per kilogram of PM_{10}**
Austria	104
Belgium	143
Denmark	162
Finland	111
France	107
Germany	135
Greece	74
Ireland	109
Italy	129
Luxembourg	194
Netherlands	174
Norway	146
Portugal	73
Spain	78
Sweden	121
Switzerland	172
United Kingdom	140

From the table we see that the marginal social costs of PM_{10} in the European countries considered varies between 73 and 194 €/kg. The main variables determining this value are population density and society's purchasing power parties, mainly defined by income.

Comparing the results with those from the ExternE bottom up approach
In Infras/IWW the authors also make a comparison between the top down approach (WHO) and the ExternE bottom up approach. Infras/IWW states that there are significant differences in these two approaches; WHO leads to higher damage costs than ExternE. However, the study does not directly compare unit values per kg of PM_{10}-emission following from both methodologies.
Comparison by CE of bottom up and top down damage estimates per passenger or tonne kilometre in the Infras/IWW study leads to the conclusion that the top down values used by WHO are, on average, 2 to 3 times higher than the bottom up values as estimated following the ExternE approach. This conclusion is in line with the results of both studies as discussed in this annex.
Infras/IWW explain this difference as follows:

– the dispersion models for health costs: Whereas the top down approach, based on the WHO study (1999) uses a particulate based modelling, including as well particulates from tyres and clutches, the ExternE model

(see above) is basing their models on exhaust emissions of transport and dividing it into a regional and a local part;
- the adjustment of VSL for health costs: Whereas the WHO-study based on a VSL of 1.4 M€, ExternE bases its assumptions on a VSL of 3.2 M€. The adjustment factors are different however;
- the building damages, based on estimations of a shortage of renovation cycles or damages to cultural buildings are not considered explicitly within the ExternE model. Their approach for material damages might therefore be an underestimation.

Comparison of the health impacts with the two approaches shows that the average values based on the WHO study are similar to the results of ExternE. The uncertainty can therefore not be explained by uncertainties in the dose-response functions.

COWI, 2000, Civil aviation in Scandinavia – an environmental and economic comparison of different transport modes, Lyngby, Denmark.

Method: damage cost.

The damage cost categories that have been included are the following[14]:
- morbidity;
- premature mortality;
- reduced farming and forestry yields;
- dirty and corroded buildings.

This study has calculated the marginal external costs of emissions. Using dose-response relationships, they arrived at the following values.

Table 54 Damage costs estimates according to COWI (2000)

Pollutant	Marginal social cost (in $€_{1999}$) per kilogram	
	Rural area	Urban area
NO_X	11	12
particulates	24	90
HC	2.7	2.7
SO_2	5.5	9.5
CO	0	0

There is no further information available on the specific functional form of the dose-response relationships that were used.

CE / TNO, Early introduction of cleaner petrol and diesel fuel in the Netherlands; analysing emission reduction potentials and cost effectiveness ['Vervroegde introductie van schonere benzine en diesel in Nederland: een analyse van emissiepotentieel en kosteneffectiviteit'], Kampman, B.E., J.M.W. Dings, R. Gense, E. van de Burgwal, Delft, 2000.

Method: overview of estimates of shadow prices used.

This study in general uses shadow prices used previously in (CE 1999) and (CE 1997). The estimates for NO_X, HC and SO_2 are based on marginal prevention costs based on (CE 1994) and for NO_X and HC additionally on the

14 Damage to the global climate is also considered in this study, but we will go into that, in the section on valuation of greenhouse gases.

costs for complying with the newest EU vehicle emission and fuel standards. With respect to PM_{10}-emission a new damage cost estimate is used based on WHO (1999| and Infras/IWW (2000). CE (2000)[15] is used additionally in order to split the damage cost estimate for PM_{10} into a rural and an urban component.

The following marginal social cost estimates are used in CE (2000).

Table 55 Marginal costs estimates used in (CE 2000), based on both damage and prevention costs

Pollutant	Approach	Marginal social cost (in €$_{1999}$) per kg	
		Rural area	Urban area
NO_x	prevention	5	7
PM_{10}	damage	35 – 70	150 – 300
HC	prevention	5	7
SO_2	prevention	3	3

European Commission, DG XII, ExternE – Externalities of Energy, 1999 (**http://externe.jrc.es/overview.html)*****, Brussels, Belgium.***

Method: damage costs.

Model: for each pollutant an impact pathway is defined. This means that for each pollutant all possible impacts are taken into account, the exposure levels are identified (how many people are exposed to what concentration for example), the effects are modelled (how many people will die premature for example) and these effects are valued (what is a life lost worth for example). This approach has been followed for all different impacts as far as possible.

The methodology has thereafter been worked out for all EU-countries. The study has focused on the production of energy in different forms. This means that the values should be seen as values that arise for emissions at ground level.

The impact categories have not all been taken into account, but the larger ones have. In the eventual estimate of the damage the following cost categories arise:
- crops;
- timber;
- building materials;
- human health;
- ecological systems;
- non-timber benefits of forests.

Alternative techniques have been developed for valuation of the last three 'goods', the main ones being hedonic pricing, travel cost methods and con-

[15] This source is not included in the list of references, because it does not provide shadow prices. It does however provide information on the effects of emissions of particulates on concentration levels in rural and urban areas. Information in |CE 2000| has been used to calculate the difference in marginal social costs in rural areas as opposed to urban areas. This had led to a ratio of 4.5 which means that the marginal social cost in rural areas has been found by dividing the value for urban areas by 4.5.

tingent valuation. For the other goods, it was possible to use the market prices, for timber, crops and so.

For each of the pollutants SO_2, NO_x (including the damage through ozone formation), and PM_{10} the damage costs are identified.

On the ExternE website, the results are given for each country separately. We will here present only the ranges found across Member States and the average value found by applying a weighed average according to each member state's population.

We would like to emphasise that the damage costs, as given in ExternE are strongly dependent on the exposure levels and thus strongly fluctuates not only *between*, but also *within* countries.

Table 56 Damage costs across the EU Member States of NO_X, SO_2 and PM_{10}-emissions according to the ExternE study

Pollutant	Marginal social cost (in €$_{1999}$) per kg	
	Medium estimate	Range
NO_x	12	2.1 – 21
PM_{10}	14	2.1 – 198
$PM_{2,5}$	23	high estimate: 75
SO_2	8.5	1.1 – 16

IER, *External costs of transport in ExternE*, with contributions by IER, ETSU, IVM, ARMINES, LIEE, INERIS, IEFE, ENCO, IOM, IFP, EEE, DLR, EKONO, 1999.

In the transport section of the ExternE research several transport cases have been researched. In this overview study some of these cases are summarised in terms of MEUR per km driven. The values are shown in the able below. Consequently, they are recalculated to units per kg of emission by using emission factors as stated in the German case study (IER 1998, Transport externalities due to airborne pollution in Germany - application of the ExternE approach, Bickel, P. et al., Stuttgart, 1998), and modification factors for these emission factors mentioned in the report.
Furthermore we assume that ozone damage is for 50% caused by HC-emissions and for 50% by NO_X-emissions.

This approach leads to the results in Table 57.

Table 57 Damage estimates (vehicle use only) for diesel passenger cars in agglomerations, urban areas and extra-urban areas, given as 'best estimate' in 1995 m€/vkm, and recalculated to 1999 €/kg of pollutant

	agglom-erations	urban areas			extra-urban areas		uncer-tainty*
	Paris	Stuttgart	Amsterdam	Barnsley	Stuttgart-Mannheim (motorway)	Tiel	
primary pollutants							
$PM_{2.5}$	534.09	50.43	78.60	97.40	18.77	29.50	B
SO_2	0.93	1.12	0.71	0.80	0.60	0.32	A/B
CO	0.02	0.003	0.003	0.005	0.001	0.0004	B
Cancers	4.02	0.54	0.57	1.25	0.18	0.22	B
secondary pollutants							
Sulphates	0.59	0.82	1.30	0.63	0.68	1.10	B
Nitrates	18.18	9.14	2.70	2.82	7.24	3.80	B?
Ozone	1.29	0.96	0.90	0.93	0.78	1.20	B
damage costs per kg of pollutant							
$PM_{2.5}$	4,800	640	620	560	240	180	B
NO_X	26	17	5.7	7.4	14	4.7	B
SO_2	54	14	11	20	9.1	7.2	B
HC	36	7.8	5.5	9.3	4.3	4.2	B

* A = high confidence (a factor 2.5 to 4); B = medium confidence (a factor 4 to 6); C = low confidence (a factor 6 to 12); „?" = evidence is weak.

It can be seen that the majority of externalities is caused by $PM_{2.5}$ and nitrate.
A study by NTNU/DNV (Environmental performance of transportation -a comparative study, Magerholm Fet, A. et al., IØT-Report nr. 3/2000), is referred to ExternE damage costs functions expressed in EUR per kg of pollutant per 1,000 inhabitants per square kilometre.

$PM_{2.5}$: 10 + 122 * pop
nitrates: 2.1 + 6.4 * pop

World Health Organization, 1999, Health Costs due to road traffic-related air pollution: an impact assessment project of Austria, France and Switzerland, *prepared for the WHO ministerial conference on environment and health, London, June 1999.*

Method: damage cost.

Model: establishing dose-exposure-response relationships between emissions PM_{10} and human health effects.

This study uses a dose-response modelling exercise. The impact of emissions of PM_{10} on human health is measured for Switzerland, France and Austria. PM_{10} is not considered to be the only air pollutant, but from other studies it seems to have the strongest correlation with health impacts and it is used a indicator for urban air pollution.

The following health effects were included in the assessment:
- total mortality based on cohort studies[16];
- respiratory hospital admissions;
- cardiovascular hospital admissions;
- chronic bronchitis in adults;
- acute bronchitis in children;
- restricted activity days in adults;
- asthma attacks in children and adults.

A potentially important health effect that is not included is acute mortality.

The dose-response modelling has been done according to the following impact-pathway:

emissions → concentration → exposure → immission → health response (mortality/morbidity) → costs.

Some important remarks on the dose-response relationships are the following:
- all air pollution-related health effects are only considered for the age groups assessed by epidemiological surveys and above the lowest assessed exposure level of 7.5 $\mu g/m^3$ PM_{10};
- WTP is used for monetary valuation;
- only PM_{10} has been assessed (the annual average concentration is taken as an indicator for urban air pollution).

The monetary valuation used for (some of the important) health effects is as follows:
- € 0.9 million per prevented fatality (total mortality costs >70% in 3 countries);
- € 0.21 million per prevented case of chronic bronchitis (74% of morbidity costs);
- € 94 per restricted activity day avoided (22% of morbidity costs).

WHO states that the most recent empirical values for the willingness to pay of a risk reduction of fatal road accidents applied is € 1.4 million. WHO corrects this value to € 0.9 million to consider the lower willingness to pay of the higher average age class of air pollution related victims.

Unfortunately, the results are not recalculated into values per unit of emission. This was done by Infras and IWW (2000) as previously discussed.

SIKA, 1999, *Översyn av samhällsekonomiska kalylprinciper och kalkylvärden på transportområdet*, SIKA nr. 6, Stockholm (summary sent in a memo by *Kågeson, P., 'Calculation values used by Swedish State Agencies in the transport sector'.*

Method: damage cost.

This memo provides the English summary of values used in Swedish transport policy. The values have been calculated in SIKA (1999)[17]. The values

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[16] Increase in premature mortality is only considered for adults older than 30 years of age. Furthermore, the results from the cohort studies only detect long-term impacts, so acute mortality is not included in the analysis.

[17] The full reference of this publication is: SIKA, 1999, Översyn av samhällsekonomiska kalylprinciper och kalkylvärden på transportområdet, SIKA nr. 6, Stockholm.

are agreed upon by the state agencies for the different modes of transport (road, rail, water and air), the Swedish Environmental Protection Agency and the Swedish Institute for Transport and Communications Analysis (SIKA). They are used in cost-benefit analyses.

The values for NO_X, SO_2, VOC and PM_{10} are based upon the damage cost method. The total damage arises from local damage, as well as regional and global damage. The cost categories that have been included are the following:

- human health;
- damage to forestry and crops;
- material damage.

For the calculation of total (marginal) damage cost the two values can be added. The following table presents the ranges in regional values, local values and total values that are used in Sweden.

Table 58 Marginal damage costs for Sweden, based on SIKA (1999)

Pollutant	Marginal social cost (in €$_{1999}$) per kilogram		
	regional damage	local damage*	total
NO_x	7.4	0.3 – 2.9	7.8 – 10
PM_{10}	0	85 – 915	85 – 915
HC	3,6	0.48 – 5.3	4.1 – 8.9
SO_2	2.1	1.2 – 26	3.3 – 28

* Mainly depending on population density; figures reflect differences between North-Sweden and the Stockholm city centre.

Ågren, C., 1999, Getting more for less: an alternative assessment of the NEC Directive, Air pollution and Climate series 13, T&E 99/9, Brussels.

Method: prevention cost method.

This study presents a critical review of IIASA et al. (1999a,b). This study does not present new estimates for the marginal costs for each pollutants, but it presents (lower) estimates for the total costs needed for meeting the National Emission Ceilings (NECs) in the different EU-countries.
We will describe the main points of criticism under the heading of IIASA et al. (1999a,b).

IIASA, DNMI and RIVM, 1999a, Economic evaluation of a directive on National Emission Ceilings for certain atmospheric pollutants: part A, Cost-effectiveness analysis, Laxenburg, Austria/ Oslo, Norway/ Bilthoven, The Netherlands.

Method: prevention costs.

Model used: RAINS (Regional Air pollution INformation and Simulation), focussing on NO_x, SO_2, NH_3 and VOC. For these pollutants emission control options are identified and costs have been determined. The associated costs include investment-related and operating costs. All investments in emission reduction are annualized using a discount factor of 4%.

Not all emission control options are incorporated in the model, only the major ones for the economic activities that contribute the most. For NO_x and VOC, only the emission control options (and emissions) are given for stationary sources. The omission of control costs of mobile sources introduces an uncertainty in the results.

In the remainder of this description we focus on the emissions ceilings for 15 European countries (EU-15) and the corresponding abatement measures and costs. IIASA et al. also present figures for non-EU-countries in Europe, but these figures are not as reliable and do not show up in the summarizing tables in the report.

Different scenarios have been used, with one central scenario in which the emissions of different pollutants in the EU overall are reduced as follows, compared to the emissions in 1990:
- NO_x: -55%;
- VOC: -60%;
- SO_2: -78%.

These reductions are the results of minimising the costs to achieve environmental targets. These environmental targets arise from the acidification and ozone-exposure strategies that was also adopted in the UN/ECE Convention on Long-range Trans-boundary Air Pollution, where for all areas a target of a '60% gap closure' of excess sulphur deposition was established. However, IIASA states (p. 96) that the targets used in its report will not be sufficient to meet the environmental long-term targets (the no-damage levels) everywhere in Europe within the next one or two decades.

Three scenarios are used:
1. A base case 'central' energy scenario, which leads to a 9% increase of CO_2-emissions between 1990 and 2010.
2. A 'low CO_2' scenario' which uses the agreements as set in the Kyoto Protocol, which boils down to a cut in CO_2-emissions by 7% in 2010 relative to 1990. This leads to a large reduction in abatement costs for NO_X and VOC, and a cut of 28% in overall costs to achieve the environmental targets for NH_3, NO_X and VOC in Europe.
3. A 'low NH_3-scenario' which is based on a 10% cut in livestock all over Europe, following an expected change in the Common Agricultural Policy. This 'new' base case, which is purely hypothetical, results in lower costs for SO_2-measures. The effects on costs of measures to reduce NO_x and VOC are small.

Table 59 Derivation of *average prevention costs* from IIASA (1999a) in three scenarios (all figures relative to the reference scenario)

	central	low CO_2	low NH_3
NO_X-reduction (ktonne)	927	856	607
HC-reduction (ktonne)	1,547	1,312	1,470
NO_X + HC-reduction costs (M€)	4,508	2,567	5,538
average NO_X + HC-prevention costs in €/kg	2.2	1.5	3.3
SO_2-reduction (ktonne)	1,050	1,368	827
SO_2-reduction costs (M€)	861	994	782
average SO_2-prevention costs in €/kg	1.0	1.0	1.2

As we mentioned under the heading of Ågren (1999), the results of this IIASA-study have been criticised. The main points of criticism in this study are the following:

- the level of ambition is fairly low: although the environmental targets in the central scenario have been strengthened in comparison with the Gothenburg Protocol, the level of ambition is low compared to the first reading of the European Commission. The targets are not sufficient to achieve the objectives laid down in the Fifth Environmental Action Plan. The long-term aim is that critical loads for both human health and vulnerable biodiversity should never be exceeded;
- the costs of achieving the NECs are overestimated because of:
  - the energy scenario which serves as the input for the future emissions is not based on meeting the agreements of the Kyoto Protocol;
  - only end-of-pipe measures are included in the list of measures that can be taken to achieve the environmental targets set, whereas fuel switching and energy and transport efficiency measures have been ignored. This method thus excludes measures that might be achieved a zero cost;
  - technological improvements (including cheaper technology) is not taken into account.

Ågren (1999) presents no other average prevention cost estimates, but presents the cost consequences of and an alternative energy scenario, which brings CO_2-emissions in 2010 down with 15% relative to 1990. In this scenario, the overall costs of meeting the NEC-directive come down from the € 7.5 billion (see IIASA, 1999a) to € 2.7 billion.

IIASA and AEA Technology, 1999b, Economic evaluation of a directive on National Emission Ceilings for certain atmospheric pollutants: part B, Benefit Analysis, Laxenburg, Austria/ Culham, United Kingdom.

Method: damage cost.

Model used: ALPHA, permits analysis of the effects of sulphur/nitrogenous pollutants and ozone on public health, materials, crops, forests, ecosystems and visibility.

Not all categories are quantified in detail, and so the authors emphasize that the benefits, which are presented in the report, are a 'subtotal'. For different policy scenarios in order to achieve reductions in NO_x, SO_2, NH_3 and ozone the emission reductions and benefits are calculated.

The scenarios differ in targets set for the different pollutants.

The larger part of the benefits comes from lower mortality and morbidity. The results therefor crucially depend upon the method used to value these health impacts. Two possibilities are explored in this study, the Value of a Statistical Life (VOSL) and the Value of a Life Year lost (VOLY).

The main difference between these two approaches is the fact that in the case of VOSL each life year lost is valued at the same price, whereas the VOLY-approach uses different values for a life year lost for a young adult and a life year lost for an elder person.

The results for the different policy scenarios are almost identical when looking at the damage cost per tonne NO_X, SO_2 and NH_3 reduced. We therefor only present the average for NO_X and SO_2 below.

Table 60 Marginal damage costs of NO_X and SO_2 found in IIASA (1999b)

Pollutant	Marginal social cost (in €$_{1999}$) per kilogram	
	Low estimate (using VOLY)	High estimate (using VOSL)
NO_X	9.4	15
SO_2	3.5	6.9

Ågren (1999) points out that the following benefits have not been quantified:
- less acidification of soil and water;
- less euthrophication;
- fewer effects on biological diversity;
- less long-term risk for lowered forest productivity;
- reduced direct health effects of NO_2 and VOCs;
- less damage to historical buildings and monuments.

IIASA, 1999c, Further analysis of scenario results obtained with the RAINS model, Laxenburg, Austria.

Method: prevention costs.

Model used: RAINS (Regional Air pollution INformation and Simulation), focussing on NO_X, SO_2, NH_3 and VOC. For these pollutants emission control options are identified and costs have been determined. The associated costs include investment-related and operating costs. All investments in emission reduction are annualized using a discount factor of 4%.

This report presents for each country the marginal social costs to achieve the environmental targets on acidification and ground-level ozone as put down in the Seventh Interim Report to the European Commission. These targets are the as follows for the EU as a whole:
- NO_X: -55%;
- VOC: -60%;
- SO_2: -78%.

The marginal prevention costs can vary widely between countries (each country has its specific environmental targets) and between economic sectors. In Table 61 below we present two figures: an 'average' marginal prevention cost and a range of marginal prevention costs. In both figures the highest prevention costs across economic sectors are taken as a reference. The ranges presented are ranges of these marginal costs across countries; the 'average' figures represent the averages across these countries.

IIASA presents in table 1.7 of its report the following marginal prevention costs.

Table 61 Marginal prevention costs according to IIASA (1999c)

Pollutant	Marginal social cost (in €$_{1999}$) per kg	
	Average over all countries	Range per country over all sectors
NO_X	4.7	0 – 13
VOC	4.6	0 – 11
SO_2	1.5	0 – 5.0

IVM 1999, Monetising the benefits of environmental policy: an exploratory investigation *['Monetarisering van baten van milieubeleid: een verkennend onderzoek'] (in Dutch), Kuik, O.J., C. Dorland, and H.M.A. Jansen, Institute for Environmental Studies (IVM), Amsterdam, 1999.*

Method: damage cost.

This literature survey attempts to estimate the benefits of environmental policy for the Netherlands. In most cases the estimates are based on European studies on dose-response relations and other underlying data.

The following categories of potential effects are discerned:
- climate change;
- human health;
- material damage;
- agricultural damage;
- nature and biodiversity.

The emissions that are taken into account are PM_{10}, $PM_{2.5}$, NO_X, and CO_2. For these emissions the impact on the different categories are determined and monetised. The authors distinguish between 'high sources' and 'low sources'. Most industrial sources are considered 'high sources', whereas transport is considered a 'low source'.

Furthermore, the authors stress that the impact of a pollutant differs largely between locations. Even for a small country like the Netherlands, this results in a factor 10 difference between high and low estimates. However, in their study they only present the value for an average location in the Netherlands. For 'high sources', this average location is Amsterdam, for the 'low sources' the arithmetic average of emissions on different locations in The Netherlands is used to 'define' the average location.

In the results, the distinction between 'low' and 'high' sources has been made as follows: for low sources, i.e. mainly traffic, the particulate matter emissions are taken as particulate matter with a diameter smaller than 2.5 micron ($PM_{2.5}$). For high sources, the particulate matter consists of particles with a diameter smaller than 10 micron (PM_{10}).

The resulting marginal social costs that were found in IVM (1999) are presented below.

Table 62 Marginal damage costs found in IVM (1999)

Pollutant	Marginal social cost (in €$_{1999}$) per kilogram	
	Medium estimate	Range
NO_X (via nitrate)	2.9	0.4 – 21
NO_X (via ozone)	1.6	0.2 – 11
NO_X (total)	4.4	0.6 – 32
PM_{10} ('high source')	12	1.6 – 85
$PM_{2.5}$ ('low source')	130	18 – 942

The most important benefits from environmental protection that IVM (1999) finds are human health benefits. These benefits can be monetised following different methods. The medium estimate in the table above and the associated range are determined with a fixed monetary value for the risk of premature death, specifically k€ 150 for a 1-year reduction of lifetime from acute mortality and k€ 50 for a 1-year reduction of lifetime from chronic mortality.

The authors note that the intervals presented, reflect uncertainties in atmospheric dispersion, in numbers of exposed population and in exposure-effect relationships. The authors have also compared their estimates with a number of international studies[18] that go into the damages avoided by environmental protection and they conclude the following from their comparison:

- the medium estimates for PM_{10} and NO_X are similar with other international sources;
- the medium estimate for $PM_{2.5}$ is near the upper bound of the estimates found in the international literature; this is mainly due to the fact that in other studies the exposure-effect relationships for 'low sources' and thus for $PM_{2.5}$ are not modelled at the same level of detail as is done in Kuik et al.

ECMT, 1998, Policies for internalisation of external costs, ECMT/OECD. Paris, France.

This study draws heavily on CE (1994) and CE (1997) and therefor this study is not worked out further.

Delucchi. M.A. 1996-1998, Report series 'The annualized social cost of motor-vehicle use in the United States based on 1990-1991 data', University of California, Institute of Transportation Studies, 1996-1998:

- 1998, The annualized social cost of motor-vehicle use in the United States 1990-1991, summary of theory, data, methods, and results; Report #1 in the series, June 1998;
- 1997, The valuation of non-monetary externalities Report #9 in the series, June 1998.

IWW et al., 1998, Entwicklung eines Verfahrens zur Aufstellung umweltorientierter Fernverkehrskonzepte als Beitrag zur Bundersverkehrswegeplanung, Karlsruhe, Germany.

Method: damage costs.

This study goes into the damage caused by NO_X, VOC and diesel particulates.
For the following categories the damage has been investigated for Germany for the year 2010:

- health;
- materials and buildings;
- forests;
- crops and animals.

Finally, acute health impacts and damage to crops are valued in terms of *average* damage costs per kg of pollutant. In Table 63 the results are shown.

[18] Most of the sources they mention have been covered elsewhere in our overview of the literature.

Table 63 Estimates of average damage costs of pollutants in Germany in 2010, according to IWW et al (1998)

Pollutant	Average social costs (in €$_{1999}$) per kg		
	total	of which health	of which crops
NO_x (via ozone)	0,23	0,16	0,07
HC (via ozone)	0,30	0,20	0,1
Diesel particulates*	37 (in urban areas)	-	-

* Based on Planco, Berücksichtiging wissenschaftlichter Erkenntnisfortschritte im Umweltschutz für die Bundesverkehrswegeplanung (BVWP, Schlussbericht im Auftrag des Bundesministeriums für Verkehr, 1995.

Note: the study gives no indication on the base year used, but some figures suggest that all monetary values are denoted in DM_{1995} and the exchange rate to the ECU used in the report itself is one ECU to 1,85 DM. We use this value as well and correct for CPI developments between 1995 and 1999.

The estimates presented may serve as an underestimate for the marginal damage per kg, because:

- not all impact categories have been monetised; only acute health damage and damage to crops is included;
- the values present average instead of marginal damage costs.

CE 1997, *Optimizing the fuel mix for road transport*, Dings, J.M.W. et al., Delft, May 1997.
Serves as a basis for CE (2000); therefore see CE (2000).

IPCC, 1996, Climate change 1995: economic and social dimensions of climate change, *contribution of Working group III to the second assessment report of IPCC, UNEP/ WMO.*

Overview of different damage estimates: the following ranges are taken from IPPC (1996) in which the social costs of air pollution are mentioned to incorporate the second order benefits of CO_2-reductions.

Table 64 Estimates of marginal damage costs of pollutants in IPCC (1996)

Pollutant	Marginal social cost (in €$_{1999}$) per kilogram				
	UK	UN ECE*	Norway	US	US
source	Pearce (1994)	Pearce (1994)	Alfsen et al. (1992)	Ottinger et al. (1990)	Scheraga and Leary (1994)
NO_x	0.2	0.7	2.2 – 44	2.8	0.1 – 1.4
particulates	30	30	2.9 – 39	3.8	0.5 – 16
SO_2	0.5	0.9	0.7 – 11	6.7	0.4 – 2.6

* Damage done by a tonne of UK emissions to Western and Eastern Europe, including UK (UN ECE region).

ITS 1996, The full costs of intercity transportation, a comparison of high-speed rail, air and highway transportation in California, Levinson, D. et al., Institute of Transportation Studies, Berkely, 1996.

This study used health cost estimates from various sources from 1977 to 1990. Due to its lack of more recent estimates we do not consider this study.

IWW/Infras, 1995, External effects of transport, UIC, Karlsruhe / Zürich / Paris.

We do not go into detail for this study, because it is a similar study as the one, which has been finalised in 2000. We therefor use the update (see Infras/IWW, 2000).

Bleijenberg, A.N., Van den Berg, W.J. and G. de Wit, 1994, The social costs of traffic, literature overview, CE, Delft.

Method: literature survey.

This study provides an extensive survey of existing literature on the valuation of the external effects that occur with transport. The literature deals with WTP-studies, damage cost estimates and prevention cost estimates.

Table 65 Overview of marginal social costs estimates in (Bleijenberg et al., 1994)

Pollutant	**Marginal social cost in €$_{1999}$ per kilogram**		
	Low	Medium	High
NO_X	1.0	5.0	6.4
HC	1.9	5.0	7.3
SO_2	0.43	1.0	3.7

In these values the results from IOO (1993) have not been included because they were much lower than the values that other studies presented. This is due to the fact that IOO (1993) has not put a value on the deterioration of agricultural land, nature and forest land and leaves aside the damage to buildings.

The following studies were included in this literature survey:
- Grupp, 1986;
- Quinet, 1990;
- Dogs and Platz, 1990;
- Klaasen, 1992;
- Teufel et al., 1993;
- Kågeson, 1993;
- Neuenschwander et al., 1992;
- Maibach et al., 1992.

We have not analysed these sources separately in our study, except for the study by Kågeson (1993).

Pearce, D.W., 1994, Costing the environmental damage from energy production, mimeo, Centre for Social and Economic Research on the Global Environment (CSERGE), University College London and University East Anglia, Norwich.

This study has been included in the literature survey of IPCC (1996). We therefor do not present the results separately.

Scheraga, J.D. and N.A. Leary, 1994, Costs and side benefits of using energy taxes to mitigate global climate change, in: Proceedings of the 86th Annual Conference, National Tax Association, Washington DC, USA.

This study has been included in the literature survey of IPCC (1996). We therefor do not present the results separately.

Teufel, D., P. Bauer, G. Bekez, E. Gauch, S. Yäkel, T, Wagner, 1993, Ökologische und soziale Kosten der Umweltbelastung in der Bundesrepublik Deutschland, Umwelt un Prognose Institut, Heidelberg, Germany.

This study has been included in the literature survey by Bleijenberg et al. (1994). We therefor do not present the results separately.

Kågeson, P., 1993, Getting the prices right, European Federation for Transport and the Environment.

Method: prevention cost.

Environmental targets for SO_2 and NO_X have been established, denoted in emission reduction in 2000 relative to levels in 1985. The targets are different for the different European countries and for each country high and low targets have been set.

IIASA has constructed national abatement curves and the resulting estimates for the marginal social cost of SO_2 and NO_X have been calculated. The following table presents the results for both pollutants and the different targets.

Table 66 Marginal prevention costs according to Kågeson (1993)

Pollutant	Marginal social cost (in €$_{1999}$) per kilogram		
	Target (relative to 1985)	Medium value	Range
NO_x (including ozone)	- 50%	4.8	3.2 - 6.4
SO_2	- 60%	1.6	0.47 - 3.9
SO_2	- 80%	3.2	0.47 - 21 1.2 - 5.8[19]

Note: the value in the report are in DM$_{1985}$; to arrive at €$_{1999}$ we have used the following conversion factors: 1 DM$_{1985}$ equals 1,2 DM$_{1993}$, exchange rate in 1993 is 1 € = 2 DM and eventually we have used the CPI to come from €$_{1993}$ to €$_{1999}$.

Kågeson also mentions that the marginal social cost for NO_X is also applicable for VOC. The IIASA model is not suit to capture targets for VOC separately and construct the abatement cost curve. Therefor, Kågeson suggests to use the value found for NO_X simultaneously for VOC.

Alfsen, K.H., A. Brendemoen and S. Glomsrød, 1992, Benefits of climate policies: some tentative calculations, Discussion paper no. 69, Norwegian Central Bureau of Statistics, Oslo, Norway.

This study has been included in the literature survey of IPCC (1996). We therefor do not present the results separately.

19 Range excluding the extreme cases of Germany (€ 0,47 per kg) and Sweden (€ 21 per kg).

Klaassen, G., 1992, Marginal and average costs of reducing nitrogen oxides and sulfur dioxide emissions in Europe – A contribution to internalizing the social costs of transport, T&E, Brussels, Belgium.

This study has been included in the literature survey by Bleijenberg et al. (1994). We therefor do not present the results separately.

Maibach, M., R. Iten and S. Mauch, 1992, Internalisieren des Externen Kosten des Verkehrs, Fallbeispiel Agglomeration Zürich, INFRAS, Zürich, Switzerland.

This study has been included in the literature survey by Bleijenberg et al. (1994). We therefor do not present the results separately.

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