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External costs of aviation

External costs of LTO emissions

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1 External costs of LTO emissions

1.1 Introduction

For the valuation of emissions other than CO_2 different methods are used. The aim of this annex is to survey the recent estimates of the valuation of environmental effects of aviation. The effects 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.

We have searched for studies that value these emissions. We have only sought for valuation of ground level effects, for being able to value the environmental effects of landings and take-offs (LTOs).

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 paragraph 1.2, with the main conclusions we draw from them. In paragraph 1.3 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 1 Conversion factors from $ to €, CPI-numbers

Year	CPI (US, 1989 = 100)	CPI (EU, 1989 = 100)	Exchange rate (1 € = .. $US)[1]
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

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

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

1.2 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 Euros 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[2]. 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.

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

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

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 2 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
Kageson	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 3 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 4 Overview of literature on the valuation of HC emissions in €1999

sources on damage costs	**aver-age**	**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 5 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
IIASA (1999c)	1.5	0-5			probably not sustainable
CE (2000)	3		3	3	based on Auto Oil standards
Kageson (1993)	1.6-3.2				depending on reduction targets

Table 6 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

1.3 Full survey 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[3].

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].

[3] 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 7.

Table 7 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[4]:
- 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 8 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 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

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[4] 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.

on WHO (1999I and Infras/IWW (2000). CE (2000)[5] 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 9 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 contingent valuation. For the other goods, it was possible to use the market prices, for timber, crops and so.

[5] 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 ICE 2000I 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.

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 10 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 11.

Table 11 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

	agglomerations	urban areas			extra-urban areas		uncertainty*
	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[6];
- 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)[7]. The values

[6] 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.

[7] 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 12 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 13 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 14 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 15 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 15 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 16 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[8] 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 17 the results are shown.

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

Table 17 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 18 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 19 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 20 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[9]

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.

[9] 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.

Neuenschwander, R., and F. Walter, 1992, External costs of transport: an overview, Ecoplan, Bern, Austria

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

Umwelt Bundesamt, 1991, Advantages of environmental protection/ Costs of environmental pollution: an overview of the research programme Costs of environmental pollution/ Advantages of environmental protection, UBA, Berlin, Germany

This set of information sheets provides an overview of different costs (of environmental pollution) and benefits (of environmental protection) that arise in Germany. Categories such as human health, biodiversity impacts, material damage were included, but the costs and benefits have not been related to units of pollution. Therefor, this study is not relevant to our research.

Dogs, E. and H. Platz, 1990, Externe Kosten des Verkehrs, PLANCO Consulting – GmbH, Essen, Germany

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

Ottinger, R.L., D.R. Wooley, N.A. Robinson, D.R. Hodas and S.E. Babb, 1990, Environmental costs of electricity, Pace University Center for Environmental and Legal Studies, Oceana Publications, New York, USA

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

Quinet, E., 1990, The social costs of land transport, OECD, Paris

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

Grupp, H., 1986, Die sozialen Kosten des Verkehrs, in: Verkehr und Technik, 1986/9, nr. 10

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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External costs of aviation

External costs of noise emission

Delft, February 2002

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Contents

1 External costs of noise emission

Effects of noise
The effects of noise from transport are increasingly studied. Within this study, we distinguish the following categories:
1. Effects on human well-being, which can be assessed via WTP/WTA studies and via property price decrease (hedonic pricing) studies.
2. Effects on human health, of which knowledge is gradually coming available.
3. Effects on indirect land use; governments put restrictions on land that is too heavily affected by noise.

The three effects can be added as higher noise levels in the long term lead to lower property values, more health costs, and more indirect land use.

Noise standards
There is an abundance of noise standards in Europe, and currently attempts are being undertaken to establish EU-wide standards. Currently national standards vary between 40 and 65 dB(A) for day-time noise (average: 52) and 40 and 55 for night-time noise (46 average). Scientist on average recommend 50-55 dB(A) as threshold value for day-time noise, and 40-45 dB(A) as threshold value for night-time noise (Infras/IWW 2000).

1.1 Overview of studies

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

Method: two methods that have been used internationally are reviewed in this study. These two methods are:
a. the willingness to pay for different noise levels (WTP);
b. the actual health risk of noise (damage cost method).

The first method measures the willingness to pay (WTP) for the reduction of noise levels. These data on willingness to pay are given in relative terms, i.e. relation to the income per capita. This results in linear relations between the (acceptable) noise level and the per capita income. Infras/IWW reviewed 5 studies:
- Pommerehne (1986);
- Soguel (1994);
- Iten (1990);
- IRER (1993);
- Weinberger (1990).

For these studies the gradient is fairly similar: for each incremental dB(A) (on average) 0.11% of per capita income is needed to compensate. Following this approach, Infras/IWW concludes that for determining the total noise cost not the marginal cost per dB (A) is crucial, but the 'target level'. Below this target level, no costs are put on the noise, above this level the cost increases by 0.11% of per capita income per dB (A). The target level can be estimated from the 5 studies, to be 50 dB (A), i.e. below this level no noise cost is apparent. Infras/IWW have decided to take a more cautious target level, namely 55 dB (A).

Table 1 WTP per person per year per dB(A) reduced in € 1999, according to Infras/IWW (2000), for the case Germany

dB(A)	55-60	60-65	65-70	70-75	>75
WTP	47	142	236	331	425

Additionally Infras/IWW value the health effects of transport noise. Two studies have empirically examined this relationship and the following table presents the results.

Table 2 Increased risk of cardiac infarctions due to transport noise, according to 2 empirical studies

Source	Location	65 – 70 dB(A)	70 – 75 dB(A)	75 – 80 dB(A)
Babisch et al. (1993)	Caerphilly, Speedwell	+ 20%	-	-
Babisch et al. (1994)	Berlin	-	+20%	+70%
Value used in Infras/IWW		+20%	+30%	

According to Infras/IWW the values found using the first method (WTP) and the damage cost (for health) can be added.

The values that are given in Infras/IWW cannot be easily translated into marginal cost per unit noise production, because noise is an 'extremely local phenomenon' (Infras/IWW). Therefor, Infras/IWW give some decisive characteristics for determining the marginal cost. These characteristics include the time zone (day and night), the land use (rural, sub-urban and urban) and traffic conditions (relaxed, dense). This exercise is necessary for each noise source separately. Another important factor is the threshold level, which is determined to be 55 dB(A) in this study. This means that the willingness to pay for a reduction in noise at a level of 55 dB(A) is zero. This threshold level is determined from a number of studies.

For the EU-countries, Switzerland and Norway (EUR17) this exercise is done, which results in (total) noise costs of €M 36,540, of which 59% comes from the WTP-approach and 41% from the health costs. Of this, air transport contributes €M 2,513, of which 62% comes from the WTP-approach and 38% from the health costs).
The amount of LTOs in 1995 in the EUR17 was 3.6 mln (table 82); the costs per LTO are thus € 700.
The total number of passengers is 582 mln (167 mln domestic and 415 mln international, table 82). Using a load factor of 50% for domestic and 65% for international transport the amount of seat LTOs is 486 mln; the costs per seat LTO then arrive at € 5.2.

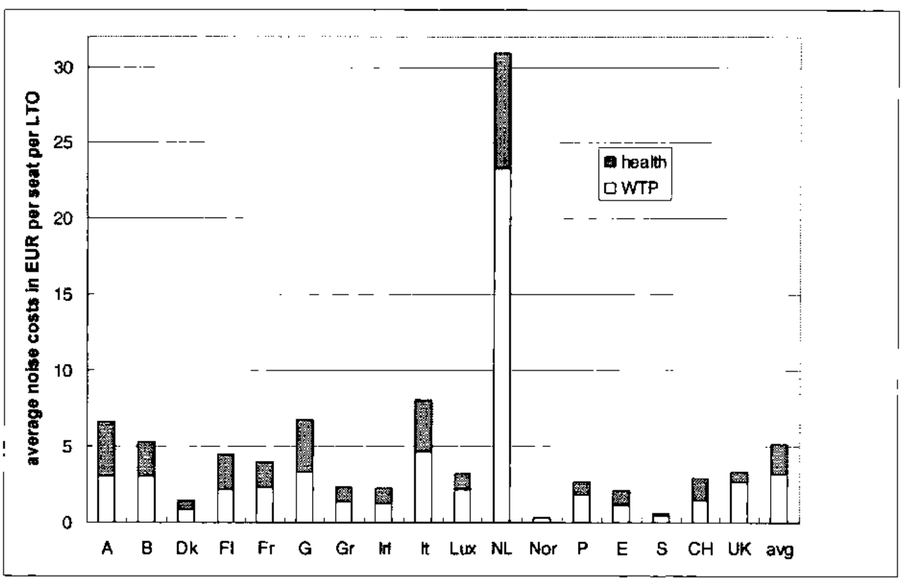
Furthermore, Infras/IWW state that the best estimate of the amount of persons exposed to different noise levels is provided in ECMT (1998).

On the estimation of noise damage from air transport, Infras/IWW states that the marginal cost can be calculated by taking 30 – 60% of the average noise cost.

Table 3 Breakdown of annual noise costs from aviation, according to Infras/IWW (2000), in €, and recalculated to average costs per LTO and seat LTO

country	WTP €M	health costs €M	# LTOs ('000)	# seat LTOs ('000)	costs per LTO €	costs per seat LTO €
Austria	20	23	71.6	6,524	601	6.6
Belgium	29	22	110.9	9,618	460	5.3
Denmark	10	7	118.7	11,629	143	1.5
Finland	13	14	56.5	5,968	478	4.5
France	161	119	498.7	69,829	561	4.0
Germany	300	311	720.7	90,726	848	6.7
Greece	12	8	60.9	8,701	328	2.3
Ireland	9	7	64.6	7,154	248	2.2
Italy	177	131	259.4	38,144	1,187	8.1
Luxembourg	2	1	18.3	931	164	3.2
Netherlands	446	146	145.4	19,154	4,072	30.9
Norway	4	1	132.1	14,273	38	0.4
Portugal	19	9	70.6	10,395	397	2.7
Spain	83	62	407.3	69,681	356	2.1
Sweden	7	2	144.1	14,882	62	0.6
Switzerland	24	24	154.1	16,521	311	2.9
United Kingdom	249	60	559.4	91,903	552	3.4
total	1,566	947	3593	486,032	699	5.2

Figure 1 Estimates of average noise costs in the EU per seat per LTO, based on Infras/IWW (2000)



The extremely high value in the Netherlands is due to the fact that the number of people in the Netherlands that are exposed to airport noise seems to be overestimated.

Marginal costs are on average about 30-60% of this amount, according to Infras/IWW.

Pearce and Pearce *'Setting Environmental Taxes For Aircraft: a Case Study of the UK'*, CSERGE, 2000

This study derived estimates of the marginal willingness to pay (MWTP) for an aircraft 'event' (landing and take-off) for each aircraft type. They started by adopting the NSDI value of around 0.6% per dBA found by Schipper (1999). By applying this NSDI value to the average house price within the Heathrow Airport 57dB(A) daytime contour and by multiplying for the number of resident households, they were able to derive an estimate of overall MWTP for a 1dB(A) Leq reduction in the area.

Table 4 The contour areas and populations of Heathrow Airport

Leq level (dB(A)	area, km^2		% change	population (,000)		% change
	1998	1999		1998	1999	
>57	163.7	155.6	-4.9%	341.0	331.6	-2.8%
>60	94.6	87.5	-7.5%	172.5	175.5	+1.7%
>63	55.4	53.9	-2.7%	82.2	91.2	+10.9%
>66	35.2	35.4	+0.6%	38.5	39.7	+3.1%
>69	28.8	21.9	-3.9%	15.5	13.8	-11.0%
>72	13.1	12.0	-8.4%	4.4	3.9	-11.4%

For comparison: the number of people within the 57 dB(A) contour of Schiphol is about 20,000.

Then, they converted this figure into a daily MWTP. In order to derive estimates of MWTP for the reduction of a daily movement of each aircraft type, they multiplied the impact on Leq (16-hr) of each aircraft type (derived from noise certification data) by the daily overall MWTP figure. Table 5 shows the resulting estimated noise damage costs per aircraft event and per LTO for selected aircraft types (UK£ = € 1.6).

Table 5 Results: external costs in € per aircraft event and per LTO per seat for Heathrow Airport

type	# seats	€ per LTO	€ per seat LTO
A310	220	108	0.5
A340	320	246	0.8
B737-400	150	108	0.7
B747-400	420	538	1.3
B757	200	140	0.7
B777-300	350	172	0.5
B777	350	106	0.3
MD82	150	148	1.0

The resulting figures are rather low compared with the results of Schipper (1999) and with the synthesis at the end of this annex, certainly when the amount of people living within the 57 dB(A) contour is taken into account. The results correspond with the lowest estimates of Schipper that are based on the HP approach.

Jansen, P.G., and D. Wagner, 2000, Lärmbewertungsverfahren für den Bundesverkehrswegeplan: Verfahrensvorschlag für die Bewertung von Geräuschen im Freiraum, F+E-Vorhaben 298 55 269, Stadtplaner AK NW, Köln, Germany

Not relevant for our study, because this study only deals with the costs of preventing noise damage. The study does not go into the desirable amount of prevention, or the damage cost of noise.

Bruinsma, F.R. et al., 2000, Estimating social costs of land use by transport: efficient prices for transport ['Raming maatschappelijke kosten van ruimtegebruik door het verkeer; Efficiënte prijzen voor het verkeer'], Free University, Amsterdam

Subject: valuation of indirect land use by Schiphol Airport.
Method: opportunity cost

This study has estimated the marginal external costs of land-use through different modes of transport. Among these modes is also aviation and Bruinsma et al. (2000) have calculated the external cost of the indirect land use around Schiphol Airport and other (regional) airports in the Netherlands. As this study intends to fill up the gap of valuation of land use in the CE study 'Efficient prices for transport' CE was asked to deliver comments to a draft version in June 2000. In cases these comments were not included in the final report, we write them down here for clarity.

Land use
Around the airport there are *cordons sanitaires* to restrict damage and nuisance, which generates costs in the form of depressed local property values. The land would be more valuable if it were usable. This implies that even if there is no actual noise nuisance or any off-site accident, there are still real costs associated with noise emissions and the risk of accidents.

The study leads to the following conclusions concerning land use by Dutch airports.

Table 6 Direct and indirect land use by Dutch airports, in km^2

	direct		indirect	
	built-up area	rural area	built-up area	rural area
Schiphol	-	26.8	8.4	222.8
regional airports	--	ca 16.7	3.3	61.9
small airports	-	ca 5.5	5.7	114.8
total	-	49	17.43	399.5

Valuation
The value Bruinsma et al. (2000) put on land-use has been calculated as follows. First, they distinguish indirect land-use in urban areas and indirect land-use in rural areas. We first go into the external cost of indirect land-use in *rural areas* they calculate, after that we describe the external cost of indirect land-use in *urban areas* they describe.

If the restrictions on the land around Schiphol were to be abolished, a part of the land would be used as a built-up area. Bruinsma et al. suggest that not all land would be used for a new function, i.e. not all land would be used for building houses, offices and so on. Bruinsma et al. assume that in non-built-up areas 20% of the land will get a different function, i.e. a change from agricultural area to built-up areas.

Comment CE: our study 'External costs of aviation' focuses on marginal costs, and should thus consider marginal changes. It is highly likely that the marginal use of land that suddenly comes available for new functions would be higher than the Dutch national average of 20%. The Nyfer study (1999) finds that about 53 km^2 could be used, or about 24%.

The difference in property values between agricultural land and land that can be built upon is estimated to be *ƒ* 50,- (€ 22) (property price per m^2 for built-up areas in the rural area) minus *ƒ* 5,- (€ 2) (the property price per m^2 for agricultural area in the rural area). This boils down to a difference property values of roughly € 20 per square metre. However, Bruinsma et al. do not take this € 20 to calculate the external cost, because they argue there is a large distributive effect due to which the economic costs of restricted land use are much lower. They argue that therefor, one should not take the price difference between agricultural land and built-up areas in the Netherlands, but instead use the difference between the property price of built-up areas *on an attractive location* and the property price of built-up areas *on a less attractive location*. Arbitrarily they choose a price difference of € 5.
Comment CE: the price difference of € 20 should be used. The distributive effect is not relevant from a national welfare point of view. Restrictions to land use around airports will indeed lead to greater demand for land in other areas. But: the higher prices that result from this do not reflect welfare gains and should thus be considered economic losses.
Consider this case. A person buys a € 2,000 computer at shop A. He would have bought this computer at shop B if it were € 100 cheaper there. Now the computer falls out of the hands of the owner of shop A so that the client cannot buy it there any more. For the client the welfare loss is only € 100, but for society welfare loss equals the full € 2,000. The same reasoning holds true for land that can not be used at one place and will therefor be used at another. The social cost is then equal to the full cost differential of (mainly) agricultural land and built-up areas.

For the indirect land-use in *urban areas*, Bruinsma et al. the methodology is roughly the same, but the figure are different. In case the noise zones of airports lies within built-up areas, a functional change is only assumed to happen for 10% of the land (p.32). The value of land in built-up areas is estimated at € 91 per m^2. Again Bruinsma at al. subtract the value of alternative land of a built-up location (€ 22) and thus arrive at a loss of € 68 per m^2.
Comment CE: again, not the distributive effect is relevant but the substitution effect which would be € 91 - € 2 = € 89 per m^2. (a 30% higher estimate).

Both comments by CE would lead to about a fivefold figure for the valuation of indirect land use outside built-up areas and a 30% higher figure for the value within built-up areas.

To arrive at the marginal social cost per vehicle kilometre the total external costs are discounted to a yearly value (using the real interest rate, 4%, as a discounting factor) and this yearly value has afterwards been allocated to the different types of aircraft.

Bruinsma et al. calculate from these assumptions, coupled to the amount of indirect land-use around the airports the following marginal social cost for indirect land-use by airports. The presentation of the marginal social cost for indirect land-use is given per person- or tonne-kilometre. The total annual costs of indirect land use cannot be directly derived from the report; close analysis suggests an annual cost of €M 8-9. Correction of these figures by CE leads to an estimate of €M 45 per annum.

Table 7 Overview of costs from indirect land use of Schiphol Airport, according to Bruinsma et al., and after modifications by CE

Type of aircraft[1]	Marginal social cost (in €ct) per person-/tonne-kilometre		
	Opportunity cost, urban area	Opportunity cost, rural area	Total opportunity cost
acc. to Bruinsma et al.			
aircraft, 150 kilometres	0.08	0.38	0.44
aircraft, 500 kilometres	0.02	0.09	0.10
aircraft, 1,500 kilometres	0.01	0.03	0.04
aircraft, 6,000 kilometres			
- passenger transport	0.00	0.00	0.00
- goods transport	0.00	0.02	0.02
after modifications by CE			
aircraft, 150 kilometres	0.10	2.40	2.5
aircraft, 500 kilometres	0.03	0.43	0.46
aircraft, 1,500 kilometres	0.01	0.13	0.14
aircraft, 6,000 kilometres			
- passenger transport	0.00	0.02	0.02
- goods transport	0.00	0.08	0.08

SEO/ Universiteit van Amsterdam, The shadow price of noise from aviation (in Dutch: De schaduwprijs van geluidhinder door vliegtuigen), not published, preliminary results presented at RLD-research days, March 23rd, 1999 (later published as Chapter 6 in: B. Baarsma, Monetary Valuation of Environmental Goods: Alternatives to Contingent Valuation, Thela Thesis no. 220, Amsterdam, 2000)

Method: non-preference method, implicit valuation through well-being evaluation measured with the use of questionnaires.

This study aims to estimate the effects on well-being from aircraft noise and to find shadow prices both for the social costs of noise nuisance and for isolation (which can be perceived as the costs of noise reduction).

First well-being is formulated as being dependent on a number of variables, among them family situation, income, age, noise nuisance and several other living conditions, such as the isolation of the houses where people live in. Sample data have been obtained using questionnaires for over 16,000 households, of which almost 3,400 responded. The estimations show that well-being is enhanced by the amount of income he or she earns and hampered by aircraft noise, as expected. Subsequently the study investigates equivalent levels of well-being for different levels of noise nuisance. In other words: the study investigates how much *income* a person would require in order to bare a higher level of noise. This gives the implicit shadow price for noise nuisance. By comparing this implicit price for houses with or without isolation and implicit price for isolation is obtained.

The Well-being evaluation method hence determines an implicit shadow price for the environmental good by *assuming* that an increase in noise can be traded off against a higher income.

The results of this approach can be given as follows.

[1] All 'types' deal with passenger transport, except for the '6,000 km' category aircraft that also carry freight.

Table 8 Shadow prices due to the increase of the noise level of 10Ke (monthly compensations in € to achieve a similar level of well-being)

category of house	Isolation	initial noise level		
		20 Ke	30 Ke	40 Ke
House of € 68,000, € 340 living costs p.m.	no	53	33	23
House of € 204,000, € 680 living costs p.m.	no	162	99	70
House of € 68,000, € 340 living costs p.m.	yes	8.1	5.0	3.5
House of € 204,000, € 680 living costs p.m.	yes	25	15	11

By subtracting the results from the investigation for houses with isolation from those without isolation, an implicit shadow price can be found for isolation. So the implicit shadow price of isolation for a house of € 68,000 laying in the zone of 20 Ke is about € 45 monthly. Interesting is moreover that in this study the additional loss in income due to an increase in aircraft noise diminishes with higher levels of initial aircraft noise: the authors interpret this as an evidence of diminishing marginal disutility as known in the economic literature.

Table 9 Overview of results of SEO (1999)

Ke-value lower limit	corresponding Lden dB(A) value (approx.)	# households*	average monthly compensation per household (€)	total annual compensation (€M)
> 20 Ke	> 49	134,705	52	**84**
> 25 Ke	> 51.5	49,052	35	21
> 30 Ke	> 54	10,041	31	3,7
> 35 Ke	>56.5	5,086	28	1.7
> 40 Ke	>59	3,511	21	0.88

* These numbers do nor correspond very well figures presented elsewhere in this annex; the cause is not clear.

NYFER, Schiphol; sea of space ['Schiphol, zee van ruimte'], Breukelen, 1999

Aim: to establish costs of indirect land use due to cordon sanitaire.

Nyfer calculates that completely moving Schiphol to another location would imply that finally about 80 km^2 of land (out of the total 258 km^2 of the cordon sanitaire) would become available for other functions. This is a net figure including all current water, infrastructure, and recreational areas, and includes reservations for rural activities. The value is well consistent with the estimate of Bruinsma et al (2000), but is criticised in a report by the Dutch CPB 'Towards a more efficient environmental policy', 2000) stating that the real value should be about one third lower. For our purposes, from this amount the direct land use of Schiphol (27 km^2) should be subtracted, leading to a net figure of about 30 km^2 of usable land currently made unavailable by the cordon sanitaire. NYFER estimates the net present value of this land, based on an average rise in land prices of € 90 per m^2, to be about €M 16 to 48 per km^2 depending on the economic scenario. The net present value of 30 km^2 would then amount to €M 480 to €M 1,440. On an annual basis (discount rate 4%) this is € 14 to € 58 mln per annum

Hamelink, P., 1999, *The cost of noise from aviation: a hedonic price study for the Schiphol region* ['De kosten van geluidhinder door vliegverkeer: een hedonische prijsstudie voor de regio Schiphol'], KUB/CE Delft

Method: hedonic pricing

This study used two different approaches, of which one has finally been published.

Approach 1: based on NDIs from international literature
Based on the arithmetic average of 29 primary international hedonic pricing studies, it was concluded that the average fall in house prices (NDI, Noise Depreciation Index) per Ke additional noise exposure on top of 20 is 0.0036. This means that for each additional Ke, the value of a house will drop by 0.36%. It should be noted that the 29 studies are based on a variety of different noise units and conversion to Ke was therefore necessary. Once converted, the results of the 29 studies were remarkably consistent. They yielded 26 NDIs of between 0.17 and 0.63. Three studies had outliers of 1.06, 1.12 and 1.36. The arithmetic average of the 26 'low' NDIs was then multiplied by the numbers of dwellings within different Ke zones at Schiphol, their value and a social discount rate of 4%.

Table 10 Review of dwelling numbers in different Ke zones in 1990

	> 65 Ke	40 – 65 Ke	35 – 40 Ke	30 – 35 Ke	20 – 30 Ke	total
average Ke	65	52,5	37,5	32,5	25	
(average dB(A))	71.5	65	58	55	51.5	)
Ke above cut-off	45	32,5	17,5	12,5	5	
# houses 1990	53	7,012	8,025	36,229	189,908	
deprec. (€/house 1990)	13,608	9,828	5,292	3,780	1,512	
deprec. 1990 (€M)	1	69	42	137	287	536
# houses 1999 (approx.)	40	6,000	7,000	18,000	92,000	
deprec. (€/house 1990)	34,020	24,570	13,230	9,450	3,780	
deprec 1999 (€M)	1	147	93	170	348	759

The average price of a house in the Schiphol area in 1990 was about € 80,000, in 2000 it was about € 210,000. Thus, we arrive at an approximate depreciation of house prices of €M 536 in 1990 and €M 759 in 2000. The latter figure has been multiplied by a 10% discount rate to convert it to an annual amount, in between the 5 and 15% values used in ECMT (1998). This yields a shadow price for the impact of noise at Schiphol of €M 76 per annum. We emphasise that this is merely an initial estimate.

Approach 2: new assessment of house prices
Hamelink conducts a hedonic pricing study on houses located in Amstelveen and Aalsmeer, nearby Schiphol Airport in the Netherlands. The study contains a model describing the sales price of houses in general with variables such as floorspace, number of rooms, year of construction, garden, proximity to the centre, etc. By collecting data from real estate agencies on sales, data have been gathered for 1997 on all houses sold in the two vicinities. By adding variables on noise levels stemming from airplanes to these data, the study can estimate the loss in real estate prices due to noise pollution.

Two measures for noise are examined in this study: Kosten-eenheden (Ke), a Dutch measure which (in short) measures noise levels outdoors weighted by the time of the day, and the LAeq-night, a weighted measure for noise levels indoors at night. Both measures are calculated measures for noise nuisance and are difficult to connect directly to international measures, such as dB(A).

The sample in this study consists of all houses sold in 1997 in the two vicinities, 796 in Amstelveen and 81 in Aalsmeer. The study finds no significant effect of noise on house prices for lower levels of noise, measured in Ke. For higher levels of noise (40-55Ke) there exists a significant negative effect on house prices in Amstelveen (which has the most observations). For lower levels of noise (below 40Ke), this study finds no significant influence on the sale prices. Also for the nuisance because of the night flights, the study finds a significant negative effect on house prices in Amstelveen. For the smaller sample of houses sold in Aalsmeer, the study finds no significant effects.

The depreciation in real estate prices because of living in the 40-55Ke zone is equivalent to almost 10% of the house prices (€ 14,700 at an average price of € 156,000). The depreciation in prices because of nuisances because of night flights consists of about 9% (€ 13,700).

The study finds that the total depreciation of house prices due to Schiphol Airport noise was €M 106 for all houses with a noise load over 40 Ke, and €M 680 for all houses with a night time load of over 20 LAeq. The €M106 is well consistent with the figure in Table 10, (€M 1+147) given the fact that in this table calculations take place with about 30% higher house prices. This €M 106 is most probably an underestimation because the largest amount of damage costs is found among households that suffer less than 40 Ke.

The conclusion can be drawn that €M 70 seems a reasonable estimate of annual costs of losses of house values die to the noise of Schiphol Airport.

Schipper, Y., 1999, Market structure and environmental costs in aviation: a welfare analysis of European air transport reform, Free University, Amsterdam

Method: literature survey, meta-analysis, mainly on hedonic pricing studies, statistical analysis of noise nuisance

Schipper presents an overview of 32 case-studies on the social costs of aircraft noise, mainly expressed in housing prices. The vast majority of them are hedonic price studies; only 2 studies have used the CVM method.

The hedonic price studies show that the Noise Depreciation Index (NDI), an internationally used standard which shows the price elasticity of noise nuisance (in dB(A)), in general moves between the 0.5 to 0.75%. This indicates that every dB(A) additional noise exposure results in a loss of property values of 0.5-0.75%.

Schipper then asks himself the question whether the results of these 30 hedonic price studies are so homogenous that the results can be transferred to other locations. For this he conducts a 'meta-analysis' on the results of these studies, which is a modern tool to answer such questions. His results show that there is no homogeneity in the results: i.e. the figure of 0.5-0.75% is not consistent without taking into account location specifics (such as income levels, average size of houses, etc.). Subsequently, Schipper identifies two types of variables which explain the variation in the NDI: location specific variables such as the average price of houses near the airport; and study specific variables which explain the differences in methodology of the studies conducted. In general, the study specific variables have more influence on the NDI estimate than the location specific variables. Studies that have not been published in scientific journals tend to find higher NDIs and the discovered NDIs tend to become lower over time. This latter result is somewhat surprising as one would expect that increasing scarcity of 'silence' would result in a higher NDI over time. The most important thing, however, is that the estimate for the NDI which, given all differences in study methodologies, are consistent with the data, is 0.48%: lower than the sample mean of the studies involved[2].

Schipper compares his estimates of the NDI from hedonic price studies with the results from WTP-studies and finds that the results from WTP-studies in general show higher external costs than hedonic price studies.

Subsequently, Schipper conducts a statistical analysis in which OECD data on the number of people living within certain noise contours nearby airports are regressed on the aircraft movements round a number of OECD-airports[3]. Schipper defines noise nuisance as the difference between the exposed noise levels and the background noise. He does not take into account noise nuisance lower than 57 dB(A) Leq.

His results show that the noise nuisance per person increases significantly with the aircraft movements at an airport and diminishes over time. This latter effect may reflect technological improvements in aircraft engines. At this place we only present the estimates for the more recent years, which have a value of 3.9 person-Leq per ACM as a basis. By applying his results to the previous results from the hedonic pricing studies, the cost estimates of an aircraft movement are estimated.

Table 11 Noise costs per aircraft movement (ACM) in 1995 €/ACM from Schipper (1999), taken for data after 1985, not differentiated for aircraft type of population density

Hedonic pricing, avg. house price of € 110,000	1,028
WTP in the USA	4,771

It should be noted that the differences between various aircraft are quite substantial. So will a Boeing 747-200 result in more than ten times higher noise costs than a Boeing 757-200. Finally, it should also be noted that these results are averages from the selected European airports. The total external costs per ACM is of course mainly influenced by the amount of houses located nearby the airport and their respective prices.

2 However, this figure of 0.48% is not significant. Nevertheless, Schipper uses it subsequently in his study. It should also be noted that such meta-analysis, as conducted by Schipper, are not free of problems. Many studies have not reported their data, as the authors had not expected that their studies could be the object of another studies dealing with their results. See van den Bergh and Button (1999).

3 These data can be found in the Environmental Data Compendia of 1987 and 1993 from the OECD. The airports which have been taken into account are: Copenhagen, Paris, Frankfurt, Dusseldorf, Munchen, Hamburg, Amsterdam, Rotterdam, Maastricht, Oslo, Geneve, Zurich, London and Manchester.

The WTP results refer here to a study of Feitelson et al. (1996) which presented cost estimates using willingness to pay for a certain number of airports in the US. It is interesting to notice that the hedonic pricing studies come to estimates 4 to 20 times lower than those in the WTP studies. Schipper claims that this may be due to the fact that WTP estimates include non-use values or recreational values and the non-committing character of questionnaires through which the WTP is established which results in a much steeper marginal external cost curve.

Table 12 Noise costs per aircraft movement (ACM) in 1995 ECU/ACM from the average European airport from Schipper (1999), differentiated according to aircraft types, and expressed in € per LTO per passenger capacity of the aircraft

aircraft type	capacity (pax)	ECU 1995 per take-off		€ 1999 per seat LTO	
		average	range	average	range
B737-300	150	555	152-2,577	8	2-37
B757-200	200	150	41-697	2	0.4-7
B767-300	275	297	81-1,380	2	0.6-11
B747-400	420	1170	320-5,430	6	2-28

It can clearly be seen that the valuations per passenger capacity per LTO (=trip) are much lower for the newer aircraft types considered (i.e. B757 and B767).

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

Method: literature survey

ECMT in their literature survey find no evidence against the assumption that the average and marginal costs of noise changes, measured on the dB (A) scale are equal. This means that the actual noise level does not influence the marginal social cost.

The literature survey is based on a couple of hedonic pricing studies, in which the income characteristics and property prices of houses are used to arrive at an estimate for the social cost of noise. For comparison purposes ECMT has converted all values into a lump-sum value. For this purpose, yearly estimates have been converted to a lump-sum estimate by assuming that persons live in a house for 50 years. Taking a shorter or longer time period does not influence the results substantially. The following studies and corresponding results are presented in ECMT (1998).

Table 13 Summary of studies cited in ECMT (1998)

Study	Lump sum value for a 1 dB (A) noise reduction (in €$_{1995}$)	Remarks
Soguel (1994)	1,044	5% discount
	384	15% discount
	231	25% discount
Colins and Evans (1994)	209	Apartment value: 32,000
	771	Semi-detached house, garden: 60,200
	1,185	detached house: 113,000
Levesque (1994)	809	house value: 44,700
Uyeno et al. (1993)	733	house value: 110,250

From these hedonic pricing estimates, ECMT calculates a shadow price per dB(A) of € 22 per person or € 58 per household per year. When using this value for other countries and sites it is necessary to adjust the figure for Purchasing Power Parities and house values, although the income elasticity of noise valuation is stated to be fairly low.

ECMT furthermore calculates the total costs from transport noise in different European countries by estimating the amount of people living in certain noise bands, i.e. the amount of people that is exposed to a certain level of noise. Noise levels under 50 dB (A) are not valued in monetary terms. The estimate for each of the transport modes under consideration is fully based on the amount of people that are exposed to a certain noise level of a certain transport mode. The external noise cost per kilometre can then be calculated, because there is no evidence from empirical studies that average costs and marginal costs are not equal.

ECMT also presents another method to estimate the external costs of noise from different transport modes. This involves a 'top-down' approach in which the total noise costs from transport in a country are expressed as a percentage of GDP. These estimates only concern noise from road transport and ECMT has also estimated the total external cost from rail transport. Unfortunately, the total external noise costs from air transport are not calculated in ECMT.

Institut für Verkehrswissenschaft, 1991, Kosten des Lärms in der Bundesrepublik Deutschland, UBA-FB 91- 076, Erich Schmidt Verlag Berlin, Germany

Method: combination of damage cost, prevention cost and willingness to pay

The study uses different methods to estimate the social cost of traffic in Germany. However, the cost estimates have not been related to a certain reduction in noise and therefor it is not possible to estimate, in the scope of our study, the social cost per certain noise unit, which can be translated to other airports.

Qualitative summary of the literature

To arrive at a common estimate for external costs from these studies is far from straightforward. Nevertheless, this literature survey can come up with some results:

1 Studies that have estimated the costs of road transport proved not to be useful for estimating aircraft noise because the hindrance from aircraft

noise has a typical peak intensity, largely absent in road transport noise, which can be described as a more general 'humming'. Proost et al. (1999) have shown that this effect is so substantial that we do not recommend to use figures from road transport to evaluate aircraft noise. This implies that the studies from Proost et al. (1999) and Bleijenberg et al. (1994) are not useful for this study.

2 External costs have been estimated different in most of the studies. This is mainly due to the different valuation approaches that have been chosen. Approaches that are dominant are either the hedonic price method (HPM) or the contingent valuation method (CVM).[4] The disadvantage of the hedonic price method is that it assumes that the true value of external costs may be underestimated. Schipper (1999, p39) concludes that the revealed preference techniques (as hedonic pricing) are only able to uncover a part of the total economic value of environmental goods. For example: the loss in recreational values for non-habitants nearby airports is not counted in hedonic pricing studies. The disadvantage of the contingent valuation method is that this method does not involve a real but a hypothetical transaction. As the filling in of questionnaires has no binding force, the answers may not reflect true market prices. Furthermore the results may be influenced by the amount of people who, under no circumstances, are willing to live nearby the airport. Such unwilling persons may influence the housing market, as housing prices may fall due to a lack of demand for houses nearby airports. Especially when taking into account the happiness of people living nearby airports, as in SEO/Baarsma (1999/2000), an underestimation of the true value of damage may occur.

3 Closely connected to the various methods that have been used for external costs, there exists different definitions of external costs in the various studies. ECMT (1998), Schipper (1999) and Hamelink (1999) have emphasized the loss in property values. SEO/Baarsma (1999/2000) have emphasised the costs of foregone well-being and Infras/IWW has emphasised the general costs (willingness to pay) and the damage costs of reduced health.

4 The disturbance from noise has been measured differently in most studies. The three most used measures of noise are the Leq, the Ldn and the Ke. The day-night average noise level, or DNL, is a 24-hour average (expressed in decibels). Night-time noise, between the hours of 10:00 p.m. and 7:00 a.m. is weighted, i.e., given an additional 10 decibels to compensate for sleep interference and other disruptions caused by loud night-time noise. (The symbol for DNL that often appears in noise monitoring systems is Ldn.) The community noise equivalent level, or CNEL, is similar to the DNL except that it includes an approximate 5 dB "penalty" for evening noise (7 p.m. to 10 p.m.), in addition to the 10 dB penalty for night-time noise. (The symbol for noise equivalent level that often appears in noise monitoring systems is Leq.). The Kosteneenheden (Ke), finally, is a Dutch measure for aircraft noise, which gives the cumulative yearly weighted noise levels. For each time of the day, a different weighting factor is attached to the maximum dB(A) noise levels of an aircraft that passes by. Nightly passages are in this way 5 times more counted than the passages during rush hours. The Ke units cannot

[4] The HPM establishes a value for external costs through the revealed preferences in associated markets: the price for houses do not only contain components for the quality of the house but also the quality of the environment in which the house is located. Noise nuisance will hence be translated in a lower value of the house than on grounds of the quality of the house could be expected. The contingent valuation method establishes a value for external costs through expressed preferences, for example, with the use of questionnaires. Typical questions are then: "how much compensatory money would you need in order to accept that an airport is located nearby your house".

be recounted into Ldn or Leq without going into details for every passage that has occurred during a year. This is due to the different calculation methods and to the fact that the Ldn and Leq estimates refer to average noise, while the Ke refers to maximum noise levels.[5] As an extremely rough, and preliminary estimate one may state that the relationship is Leq dB(A) = 39+0.5*Ke. In reality there is no linear relationship between Ke and Leq.

5 The minimum level of noise under which no external effects can be expected differs between the studies. While ECMT (1998) has assumed a minimum level of 50 dB(A), Infra/WWW (2000) has estimated that the minimum level is 55 dB(A) and Schipper (1999) has used a minimum level of 57 dB(A). SEO/Baarsma (1999/2000) have taken a minimum level of 20 Ke (i.e. about 49 dB(A). These substantial differences matter for the estimation of the total external costs.

6 Also the shape of the external damage function is ambiguous from the various studies. Bruinsma et al. (2000) and Schipper (1999) assumed linear marginal cost functions. But SEO/Baarsma (1999) found concave marginal cost functions (i.e. decreasing marginal costs for higher levels of disturbance), and also Infras/IWW state that marginal costs of noise are generally 30-60% of average costs.

7 Finally, the slope of the external cost function can be estimated to lay in between 0.4-0.75%. This implies that every dB(A) increase in noise levels result in an increase in external costs by 0.4-0.75%. Schipper (1999) is the only study, which has attempted to compare various results of the slope of the external cost function, and he arrives at a figure of 0.48% (though it is not significantly different from zero).

Quantitative survey of the literature

Estimates of external costs from Schiphol Airport

Noise damage costs from Schiphol Airport are quite extensively studied. In this paragraph we will show a synthesis of the studies considered earlier in this annex.

[5] The Dutch government has launched a study project in which during five years both the Ldn and the Ke estimates will be produced for a period of five years to establish a comparison of Dutch figures for aircraft noise with internationally comparable measures.

Table 14 Overview of annual external noise cost estimates results for Schiphol Airport

source	external cost estimate (€M/yr)	remarks
1 estimates with HP (Hedonic Pricing) approach		
Hamelink (1999) approach 1	ca 76	NDIs from literature
Hamelink (1999) approach 2	ca 68	primary HP research
2 estimates with CVM (stated preference) approach		
SEO (1999)	88	WTA, compensation
Infras/IWW	446	all Dutch airports, WTP, we suspect that exposure data have been overestimated.
3 estimates of costs of indirect land use due to 'cordon sanitaire'		
Bruinsma et al, after correction	ca 45	indirect land use from 'cordon sanitaire', no time path
Nyfer (1999), after correction	ca 14-58	indirect land use from 'cordon sanitaire', based on NPV time path 2000-2030
4 estimates of health costs		
INFRAS/IWW	146	all Dutch airports, we suspect that exposure data have been overestimated
	30	indicative correction for health costs from noise from Schiphol Airport
estimates of total external costs (1 or 2 + 3 + 4)		
sources mentioned	100-200	indicative minimum and maximum estimates of external costs from noise from Schiphol Airport

These results show that

- except the Infras/IWW study, CVM and HP approaches show approximately the same order of magnitude;
- the annual costs as a result of the 'cordon sanitaire' seem to be somewhat lower than the damage cost estimates;
- the annual costs resulting from the CVM approach of SEO are the highest values found. This is in line with the conclusions by Schipper (1999);
- the total external costs from noise at Schiphol are about twice as large as the costs that are derived from HP studies.

Based on the methodology for distribution of external costs across different aircraft types from CE (1999) this leads to the following noise costs for different aircraft types.

Table 15 Average noise costs per aircraft type per LTO at Schiphol Airport, based on €M 100-200 of total external noise costs and on the allocation methodology in (CE, 1999)

MTOW (tonnes)	maximum payload	capacity (seats)	typical dist. (km)	noise factor	€ per LTO	€/LTO/seat available	
						average	marginal*
10	3.9	30	150	0.3	140-270	5-9	2-4
50	11	100	500	1.0	450-910	5-9	2-4
70	17	130	1,500	1.3	570-1,130	4-9	2-4
280	48	240	6,000	2.5	1,100-2,300	4-8	2-4

* Marginal costs calculated as 50% of average costs, based on Infras/IWW (2000)

Comparing Schiphol with other airports considered

The number of inhabitants within a radius of 25 km from Schiphol (1,965 km^2) is about 1.8 million people, or about 900 people per km^2. About 8% of this circle is North Sea.

With respect to Charles de Gaulle Airport, within a circle of 25 km radius, more than one half of the city of Paris will be covered (2 million people). The other 75% of the virtual circle adds another 1.8 million people (900 per km^2).

This leads to an estimation of 3.8 million people living within 25 km of Charles de Gaulle Airport, or about 2,000 per km^2.
Frankfurt Airport: the Rhine-Main region, covering major cities like Frankfurt, Mainz and Wiesbaden, has 4.8 million inhabitants. The Rhine Main region is surrounding Frankfurt Main airport and is about 11,000 km^2, which means that the average number of inhabitants is about 431 per km^2.
It can be concluded that Schiphol has a medium position from the point of population density. At Charles de Gaulle the density is about twice that of Schiphol, at Frankfurt it is about half.

Marginal costs of extra aircraft movements
- the external costs of noise per aircraft type per LTO are dependent on aircraft size (MTOW), and even more on aircraft technology level;
- in this study we will base our estimate for the marginal costs on half that of average noise costs (total external costs divided by number of LTOs);
- external costs of noise per LTO are, within a given technology level, more or less linearly dependent on aircraft size in terms of maximum payload, and number of seats (not for freight).

Estimates for the marginal external costs of noise per seat per LTO vary between € 0.2 and € 37, depending on valuation methodology, aircraft technology, and number of people affected. See Table 16.

Table 16 Overview of estimates of external noise costs per seat LTO

	range	average estimate	average or marginal costs?	explanation
IWW/Infras (2000)	0.4-30	5.2	average	low = Norway average high = Netherlands average average = EU average
Pearce and Pearce (2000)		0.3	marginal	B777, Heathrow
		1.3		B747-400, Heathrow
Schipper (1999)	2-37	8	marginal	B737-300, depending on income & location
	0.4-7	2		B757-200
	0.6-11	2		B767-300
	2-28	6		B747-400
CE (2001) (estimate in this annex)		4-9	average	figures apply to Schiphol, to all aircraft sizes, to average technology level

- medium estimates for marginal costs, for an airport with an EU average population density, arrive at about € 3 per seat LTO for aircraft with fleet average technology.

U. S. Standards

Additional literature used

Feitelson, E.I., R.E. Hurd, R.R. Mudge (1996). The impact of aircraft noise on willingness to pay for residences. *Transportation Research* **1D:** 1-14.

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External costs of aviation

Allocating costs to passengers, freight, and aircraft types

Delft, February 2002

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Contents

1 Allocating costs to pax, freight, and aircraft types

In this annex it is discussed how external costs of aircraft movements will be allocated to passenger and freight, in cases both passengers and freight are transported.

Table 1 The characteristics of the four different types of aircraft or market segments, as the case may be, and the distribution of LTO numbers (Landings and Take-Offs, flight movements divided by 2) for scheduled and charter flights to and from airports in the Netherlands in 1997 for these segments

typical distance (km)	MTOW (tonnes)[a]	maximum payload (tonnes)	capacity (pax)	utilisation (%)	number of pax	freight (tonnes)
200 km[c]	17	4.5	40	50%	20	0
500 km	50	12	100	65%	65	1
1,500 km (EU)	110	24	120	70%	140	2
6,000 km ICA[b]	397	72	400	80%	330	25

a Maximum Take-Off Weight (empty weight + fuel + load)

b ICA: Intercontinental

c This segment of the market, as regards characteristics relevant to this study, (MTOW, use of energy, distance, level of capacity utilisation) is defined such that it is representative of domestic flight traffic.

All four types of aircraft have been considered for passenger transport. It can be seen that the difference in *freight* carried between the four is very large. The freight carried varies in the order of 1 tonne for the small types and 17.5 tonnes for the large ones. It is evident from this that most freight is carried in these large aircraft, which generally fly between continents. The average distance for KLM and Lufthansa freight transport, for instance, is about 6,000 km.

Allocation to passengers and freight

For the calculation to be correct, external costs must be allocated to freight and passengers. In aviation it is usual to allow 100 kg per passenger. However, for this study we must view allocation in a broader perspective. It is evident that what are known as *full freighters* have a much higher payload (total maximum permissible load) than those known as *combis*. Thus the full freighter version of the 747-400 has a payload of 129.1 tonnes, whereas the 'combi' version (which can carry 410 passengers) only has a payload (freight plus passengers at 100 kg per person) of 72.2 tonnes. This means that ultimately exactly the same aircraft loses a great deal of its *total* load capacity if it has to be fitted for passengers. Correct allocation requires that the mass of all facilities required for passenger transport be allocated to the passengers. This then results in a representative mass of (129,100 – 72,200)/410 + 100 = 240 kg for one passenger and his or her facilities.

This improved allocation does not affect total costs, but it does mean that air freight is less heavily affected than would have been the case with an allocation of 100 kg per passenger, whilst passenger transport is affected more heavily. Thus for an ICA flight carrying 320 passengers and 25 tonnes of freight it means that the passenger/freight ratio becomes 75/25 instead of 56/44. This adjusted allocation has little effect on passenger transport in smaller aircraft: 100, 97 and 95%, respectively is allocated to passengers instead of 100, 93, 89%, respectively.

A consequence therefore of this allocation method is that it now makes no difference in principle to the outcome for *freight transport* whether a full freighter or a combi is used.

Allocation to aircraft types
This project requires that the costs for infrastructure and noise nuisance be allocated 'top-down' to the four types of aircraft. This was done with weighting factors, which were derived from the current charges for the various types at the airports. For infrastructure costs this means a strong correlation between MTOW and the number of passengers. For noise the proportion of fixed charges was used which Schiphol levies on aircraft over 20 tonnes where airlines do not/cannot submit any dimensional data. These fixed weighting factors depend on (the power of 2/3 of) MTOW and on what is known as the 'k factor' which indicates in which noise class an aircraft is placed in the absence of further information. The same k factor is assumed for all three aircraft types of MTOW over 20 tonnes. For the smallest aircraft (200 km, 17 tonnes MTOW), a weighting factor has been derived based on the prescribed formula for such small aircraft based on the current level of charging of ca. € 10 and the expected future increase is estimated at 30% of that for aircraft of 50 tonnes MTOW. See Table 2.

Table 2 Allocation factors for charges and infrastructure and noise nuisance costs

type of aircraft	infrastructure costs weighting factor	noise costs weighting factor
40 seat 200 km	1	0.3
100 seat 500 km	5	1.0
200 seat 1,500 km	10	1.7
400 seat 6,000 km	25	3.2

It appears that allocation of noise to different aircraft types is practically linearly dependent on the number of seats per aircraft.