ECONOMIC EVALUATION OF HEALTH IMPACTS DUE TO ROAD TRAFFIC-RELATED AIR POLLUTION

An impact assessment project of Austria, France and Switzerland

by H. SOMMER, N. KÜNZLI, R. SEETHALER, O. CHANEL, M. HERRY, S. MASSON, J-C. VERGNAUD, P. FILLIGER, F. HORAK Jr., R. KAISER, S. MEDINA, V. PUYBONNIEUX-TEXIER, P. QUÉNEL, J. SCHNEIDER, M. STUDNICKA

Summary

In preparation for the Transport, Environment and Health Session of the WHO Ministerial Conference on Environment and Health in London (June 1999) a tri-lateral project was carried out by Austria, France and Switzerland.

The project assessed the health costs of road-traffic related air pollution in the three countries using a common methodological framework.

Based on the average yearly population exposure to particulate matter with an aerodynamic diameter of less than 10 μm (PM10) and the exposure-response function for a number of different health outcomes, the number of cases attributable to (road traffic-related) air pollution was estimated.

Using the willingness-to-pay as a common methodological framework for the monetary valuation, material costs such as medical costs and loss of production or consumption as well as the intangible costs of pain, suffering, grief and loss in life quality were considered. The monetary valuation provided the following results (see Summary Table).

All three countries together bear some 49'700 million EUR[1] of air pollution related health costs, of which some 26'700 million EUR are road-traffic related. In each country, the mortality costs are predominant, amounting to more than 70 %.

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[1] 1 EUR ≈ 0.94 US $, April 2000

The annual national per capita costs of total air pollution related health effects result in a similar range of values for all three countries. Considering the per capita health costs due to road traffic-related air pollution, the differences between the countries are even lower with a range from 180-540 EUR for Austria (central value 360 EUR), 190-560 EUR for France (central value 370 EUR) and 160-470 EUR for Switzerland (central value 304 EUR).

Summary Table. **Health costs due to road traffic-related air pollution in Austria, France and Switzerland based on the willingness-to-pay approach (1996)**

	Austria		**France**		**Switzerland**	
	Total costs with road traffic share	Costs attributable to road	Total costs with road traffic share	Costs attributable to road	Total costs with road traffic share	Costs attributable to road
Costs of mortality (million EUR)	5'000 3'000 - 7'000	2'200 1'300 - 3'000	28'500 17'300 - 39'900	15'900 9'600 - 22'200	3'000 1'800 - 4'200	1'600 1'000 - 2'200
Costs of morbidity (million EUR)	1'700 400 - 3'000	700 200 - 1'300	10'300 2'800 - 18'500	5'700 1'500 - 10'300	1'200 300 - 2'100	600 200 - 1'100
Total costs (million EUR)	**6'700** 3'400 - 10'000	**2'900** 1'500 - 4'300	**38'800** 20'100 - 58'400	**21'600** 11'100 - 32'500	**4'200** 2'100 - 6'300	**2'200** 1'200 - 3'300

	all three countries	
	Total costs with road traffic share	Costs attributable to road
Costs of mortality (million EUR)	36'500 22'100 - 51'100	19'600 11'900 - 27'500
Costs of morbidity (million EUR)	13'200 3'500 - 23'700	7'100 1'900 - 12'800
Total costs (million EUR)	49'700 25'600 - 74'900	26'700 13'700 - 40'200

1. Introduction

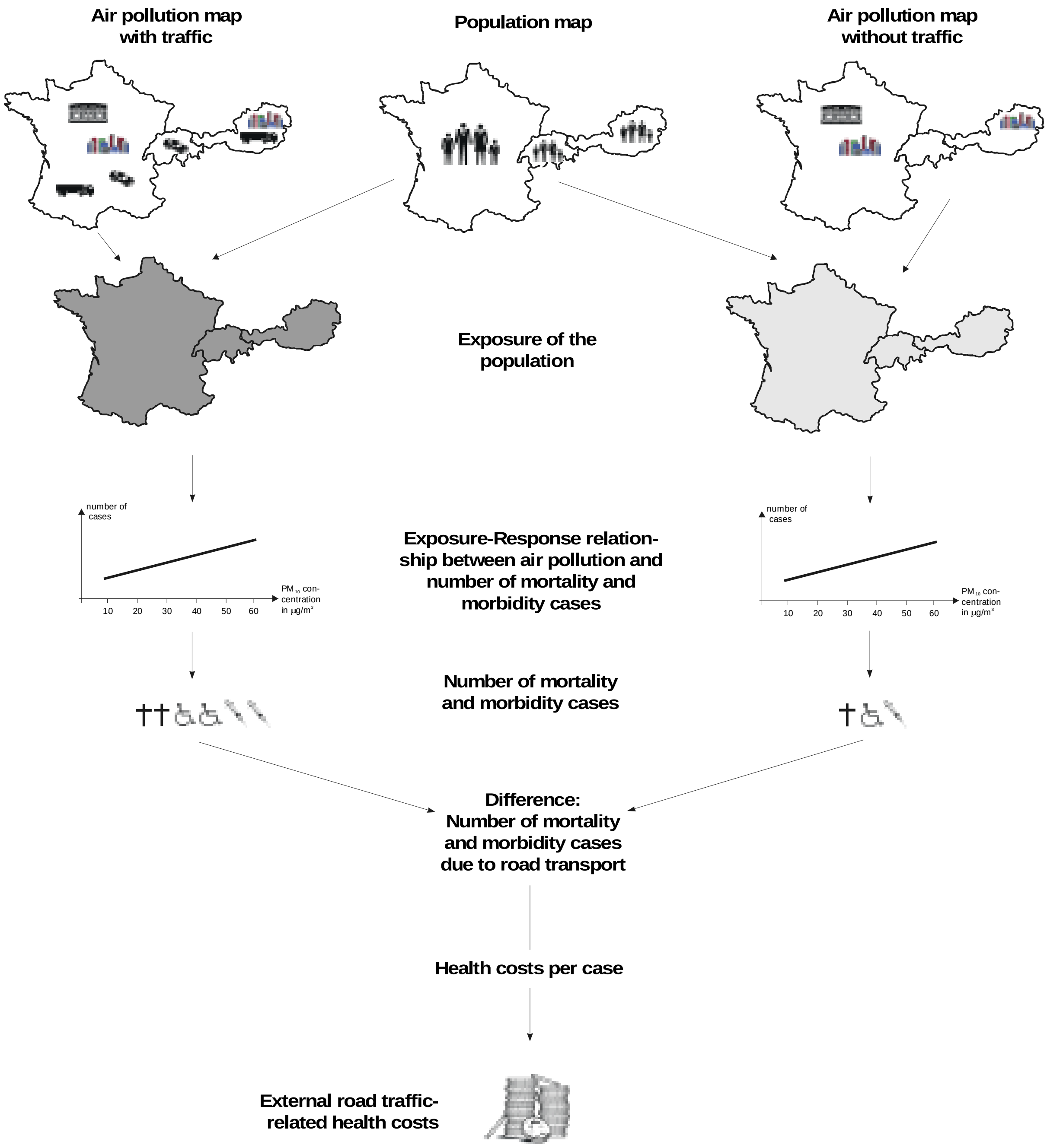
The objective of this tri-lateral research project was to quantify the health costs due to road traffic-related air pollution. The project was carried out by Austria, France and Switzerland. The results of this co-operation provided an input for the WHO Ministerial Conference in June 1999.[2]

The monetary evaluation of the health costs is based on an interdisciplinary co-operation in the fields of air pollution, epidemiology and economy. Figure 1 presents an overview of the different tasks of the three domains.

- **Air pollution**: Evaluation of the (traffic related) exposure to particulate matter: The starting point of the study is the determination of the pollution level in 1996 to which the population was exposed. The entire population of Austria, France and Switzerland is subdivided into categories of exposure to different classes of pollution levels from a superposition of the mapping of ambient concentration of particulate matter (average annual PM_{10}) with the population distribution map. In addition, a scenario without road traffic-related emissions is calculated and the exposure under these theoretic conditions is estimated.
- **Epidemiology**: Evaluation of the exposure-response function between air pollution and health impacts: The relationship between air pollution and health has to be assessed. Thereby it has to be shown, to which extent different levels of air pollution affect a population's morbidity and mortality. This evaluation is based on the latest scientific state of the art presented in the epidemiologic literature and comprehends the results of extensive cohort studies as well as time series studies.
- **Economics**: Evaluation of the traffic-related health impacts and their monetarisation: Using epidemiological data regarding the relation between air pollution and morbidity and premature mortality, the number of cases of morbidity and/or premature mortality attributed to air pollution is determined for each of the health outcomes separately, using specific exposure-response functions. The same operations are carried out for the theoretical situation in which there is no road traffic-related air pollution. The difference between the results of these two calculations corresponds to the cases of morbidity and premature mortality due to road traffic-related air pollution. The morbidity and mortality costs arising from road traffic-related air pollution are then evaluated for each health outcome separately by multiplication of the number of cases with the respective cost estimates (willingness-to-pay factors for the reduction of the different health risks).

[2] Third WHO Ministerial Conference on Environment and Health, London, 16-18 June 1999.

Figure 1. **Methodological approach for the evaluation of mortality and morbidity due to road traffic-related air pollution**



Throughout the entire project many assumptions and methodological decisions had to be made along the various calculation steps in the domains of air pollution, epidemiology and economics. On each level, the method of dealing with uncertainty had to be defined. The research group decided that the main calculation ought to apply an **"at least" approach**, thus consistently selecting methodological assumptions in a way to get a result which may be expected to be "at least" attributable to air pollution. Accordingly, the overall impact of air pollution is expected to be greater than the final estimates. To unambiguously communicate the uncertainty in the common methodological framework, the final results will be reported as a range of impacts rather than as an exact point estimate.

2. Epidemiology - the air pollution attributable health effects

In the last 10-20 years epidemiology has dealt extensively with the effect of outdoor air pollution on human health. A considerable number of case studies in different countries and under different exposure situations have confirmed that air pollution is one of various risk-factors for morbidity and mortality.

In general, air pollution is a mixture of many substances (particulates, nitrogen oxides, sulfur dioxides). Knowing that several indicators of exposure (eg. NO_2, CO, PM_{10}, TSP etc.) are often highly correlated, it is not accurate to establish the health impact by a pollutant-by-pollutant assessment, because this would lead to a grossly overestimation of the health impact. The objective is therefore to cover as best as possible the complex mixture of air pollution with one key indicator. Based on various epidemiological studies, in the present study PM_{10} (particulate matter with an aerodynamic diameter of less than 10 µm) is considered to be a useful indicator for measuring the impact of several sources of outdoor air pollution on human health. The derivation of air pollution attributable cases has been described in a separate publication.[3] Thus, the key features of the epidemiology based assessment are only summarized.

For the assessment of the health costs it was not possible to consider all health outcomes found to be associated with air pollution. Only those meeting the following three criteria were considered:

- there is epidemiological evidence that the selected health outcomes are linked to air pollution;
- the selected health outcomes are sufficiently different from each other so as to avoid double counting of the resulting health costs (separate ICD[4] codes);
- the selected health outcomes can be expressed in financial terms.

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[3] Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.

[4] ICD: International Classification of Diseases.

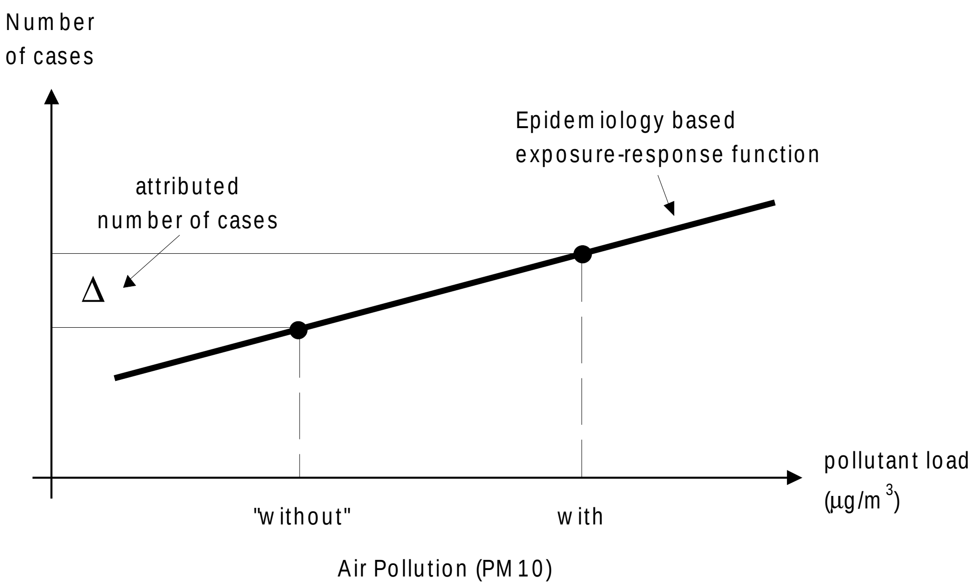
According to these selection criteria seven health outcomes were considered in this study (see Table 2).

Table 2. **Air pollution related health outcomes considered**

Health outcome	Age
Total mortality	Adults, ≥ 30 years of age
Respiratory hospital admissions	All ages
Cardiovascular hospital admissions	All ages
Acute bronchitis	Children, < 15 years of age
Restricted activity days	Adults, ≥ 30 years of age
Asthmatics: asthma attacks	Children, < 15 years of age; Adults, ≥ 15 years of age

The relation between exposure to air pollution and the frequency of health outcome is presented in Figure 3 by graphical means. The number of mortality and morbidity cases due to air pollution can be determined if the profile of the curve (**exposure-response function**) and its position (**health outcome frequency**) are known. These two parameters were determined for each health outcome, separately.

Figure 3. **Relation between air pollution exposure and cases of disease**



The **exposure-response function** (quantitative variation of a health outcome per unit of pollutant load) was derived by a meta-analytical assessment of various (international) studies selected from the peer-reviewed epidemiological literature. The effect estimate (gradient) was calculated as the variance weighted average across the results of all studies included in the meta-analysis.

In this project, the impact of air pollution on mortality is based on the long-term effect. This approach is chosen because the impact of air pollution is a combination of acute short-term as well as cumulative long-term effects. For example, lifetime air pollution exposure may lead to recurrent injury and, in the long term, cause chronic morbidity and, as a consequence, reduce life expectancy. In these cases, the occurrence of death may not be associated with the air pollution exposure on a particular day (short-term effect) but rather with the course of the chronic morbidity, leading to shortening in life.

Accordingly, for the purpose of impact assessment, it was decided not to use response functions from daily mortality time-series studies to estimate the excess annual mortality but the **change in the long-term mortality rates** associated with ambient air pollution.[5]

Contrary to the exposure function which is assumed to be the same for all countries, the **health outcome frequency** (frequency with which a health outcome appears in the population for a defined time span) may differ across countries. These differences may result from a different age structure or from other factors (i.e. drinking and eating habits, different health care systems in the three countries, etc.). Therefore national or European data were used whenever possible to establish the countries' specific health outcome frequency.

For each health outcome included in the trinational study, Table 4 presents the effect estimates in terms of relative risks (column 2) and separately for each country the health outcome frequency (column 3-5), and the attributable number of cases for 10 $\mu g/m^3$ PM_{10} increment.

Reading example:

The relative risk of long-term mortality for a 10 $\mu g/m^3$ PM_{10} increment is 1.043 (column 2), therefore the number of premature fatalities increases by 4.3% for every 10 $\mu g/m^3$ PM_{10} increment. Column 5 shows the number of deaths (adults ≥ 30 years) per 1 million inhabitants in Switzerland (8'260). With an average PM_{10} concentration of 7.5 µg/m3 a baseline frequency of 7'794 deaths would be expected. This proportion depends on the age structure of the population ≥ 30 years and therefore is different for each country.

The absolute number of fatalities (340 cases for Switzerland, column 8) per 10 $\mu g/m^3$ PM_{10} increment and per 1 million inhabitants corresponds to the 4.3% increase in mortality (column 2) applied to the baseline frequency of 7'794 deaths.

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5 Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.

Table 4. **Additional cases per 1 million inhabitants and 10 $\mu g/m^3$ PM_{10} increment**[6]

	Effect estimate relative risk (±95% confidence interval)	Observed population frequency, P_e Per 1 million inhabitants and per annum			Fixed baseline increment per 10 $\mu g/m^3$ PM_{10} and 1 million inhabitants (±95% confidence interval)		
		Austria	France	Switzerland	Austria	France	Switzerland
Long-term mortality (adults ≥30 years; excluding violent death)	1.043 (1.026-1.061)	9'330	8'390	8'260	370 (230-520)	340 (210-480)	340 (200-470)
Respiratory hospital admissions (all ages)	1.0131 (1.001-1.025)	17'830	11'550	10'300	230 (20-430)	150 (20-280)	130 (10-250)
Cardiovascular hospital admissions (all ages)	1.0125 (1.007-1.019)	36'790	17'270	24'640	450 (230-670)	210 (110-320)	300 (160-450)
Chronic bronchitis incidence (adults ≥25 years)	1.098 (1.009-1.194)	4'990	4'660	5'010	410 (40-820)	390 (40-780)	430 (40-860)
Bronchitis (children < 15 years)	1.306 (1.135-1.502)	16'370	23'530	21'550	3'200 (1'410-5'770)	4'830 (2'130-8'730)	4'620 (2'040-8'350)
Restricted activity days (adults ≥20 years)[a]	1.094 (1.079-1.109)	2'597'300	3'221'200	3'373'000	208'400 (175'400-241'800)	263'700 (222'000-306'000)	281'000 (236'500-326'000)
Asthmatics: asthma attacks (children < 15 years)[b]	1.044 (1.027-1.062)	56'700	62'800	57'500	2'330 (1'430-3'230)	2'600 (1'600-3'620)	2'400 (1'480-3'340)
Asthmatics: asthma attacks (adults ≥15 years)[b]	1.039 (1.019-1.059)	173'400	169'500	172'900	6'280 (3'060-9'560)	6'190 (3'020-9'430)	6'370 (3'100-9'700)

a: Restricted activity days: total person-days per year
b: Asthma attacks: total person-days per year with asthma attacks
P : Frequency as observed at the current level of air pollution

6 Table printed with permission from Lancet, Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.

3. Air Pollution - the PM_{10} population exposure

In addition to the epidemiological data need, information on the population's exposure to PM_{10} is a further key element for the assessment of air pollution-related health effects. Information about the sources and the spatial distribution of PM_{10} is still sparse in Austria, France and Switzerland as it is in many other European countries. Therefore it was necessary to calculate the spatial distribution of PM_{10} by using empirical dispersion models or statistical methods. The general methodological framework for the air pollution assessment consisted of four main steps:

- acquisition and analysis of the available data on ambient concentration of particulate matter (Black Smoke BS, Total Suspended Particulate TSP and PM_{10}) for model comparison or correlation analysis between different particle measurement methods

    - PM_{10} mapping by spatial interpolation with statistical methods or empirical dispersion modelling;

    - estimation of the road traffic-related part of PM_{10} (based on emission inventories for primary particles and for the precursors of secondary particles);

    - estimation of the population exposure from a superposition of the PM_{10} map on the population distribution map.

The differences between the countries concerning the procedures for measuring ambient particulate matter and the availability of emission data led to an adaptation of the general framework to the individual country specific case.

In **Austria**, particulate matter is measured in agreement with national legislation as Total Suspended Particulate (TSP) at more than 110 sites, whereas PM_{10} measurements are not yet available. It was assumed that ambient air TSP levels can be attributed to the contribution of local sources and regional background concentrations. Both of them were modelled separately. The starting point for the modelling of local contributions was the availability of a spatially disaggregated emission inventory for nitrogen oxides (NO_x). An empirical dispersion model was established for NO_x whose results could be compared with an extended network of NO_x monitors. The spatial distribution of NO_x was converted into TSP concentrations, using source specific TSP/NO_x conversion factors. The regional background TSP levels were estimated from measurements and superimposed on the contributions from local sources. These results were compared to measured TSP data. Finally, PM_{10} concentrations were derived from TSP values by applying source specific TSP/PM_{10} conversion factors. The model is able to provide an estimate of the traffic-related part of PM_{10} concentration.

The **French work** was based on the available Black Smoke (BS) data. A correlation analysis between BS and PM_{10} (TEOM method[7]) was first carried out. It was found that at urban background sites, BS and PM_{10} (TEOM) are about equal. Following this, linear relationships were sought between the BS data and land use categories in the areas surrounding the measurement sites. Multiple regression analysis was performed for three categories of sites: urban, suburban and rural. Based on these regressions and using the land use data set, a PM_{10} map was established. A correction factor for secondary particles was defined using the European scale EMEP[8] model. This was necessary because BS and TEOM considerably underestimate the amount of secondary particles in PM_{10}. The percentage of PM_{10} caused by road traffic was determined in each grid cell using results from the Swiss PM_{10} model.

The **Swiss work** was based on a provisional national PM_{10} emission inventory. It was first disaggregated to a km^2 grid. Dispersion functions for primary PM_{10} emission were defined in an empirical dispersion model which was used to calculate the concentration of primary PM_{10}. The contribution of secondary particles was modelled by using simple relationships between precursor and particle concentration. The long-range transported fraction was taken from European scale models. The PM_{10} fractions were then summed to create the PM_{10} map. The traffic related part was modelled separately, using both the road-traffic related portion of PM_{10} emission and the respective portion of the precursor emission for secondary particles.

The determination of the **regional PM_{10} background** was critical to the PM_{10} mapping procedures. The estimates of all three countries are in line with measured and modelled data from EMEP. The large-scale transported fraction of PM_{10} is considerable. At rural sites, over 50 % of PM_{10} may originate from large-scale transport. Furthermore, the contribution of traffic to PM_{10} background concentration is substantial and it may vary in space.

The **population exposure to total PM_{10}** is presented in Figure 5. Around 50% of the population live in areas with PM_{10} values between 20 and 30 $\mu g/m^3$ (annual mean). About one third is living in areas with values below 20 $\mu g/m^3$. The rest is exposed to PM_{10} concentrations above 30 $\mu g/m^3$. The high concentrations are found exclusively in large agglomerations.

7 TEOM: Tapered element oscillating microbalance. Method for measuring continuously particle concentration.

8 EMEP: Co-operative Programme for the Monitoring and Evaluation of Long-Range Air Pollutants in Europe.

Figure 5. **Frequency distribution of total PM_{10} population exposure (with share attributable to road traffic)**[9]

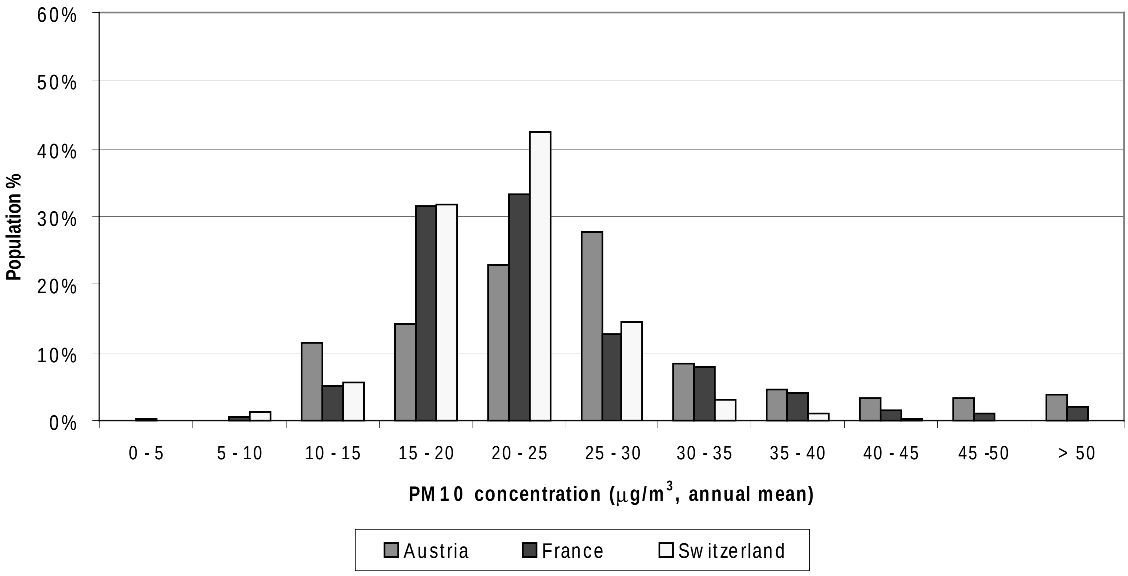
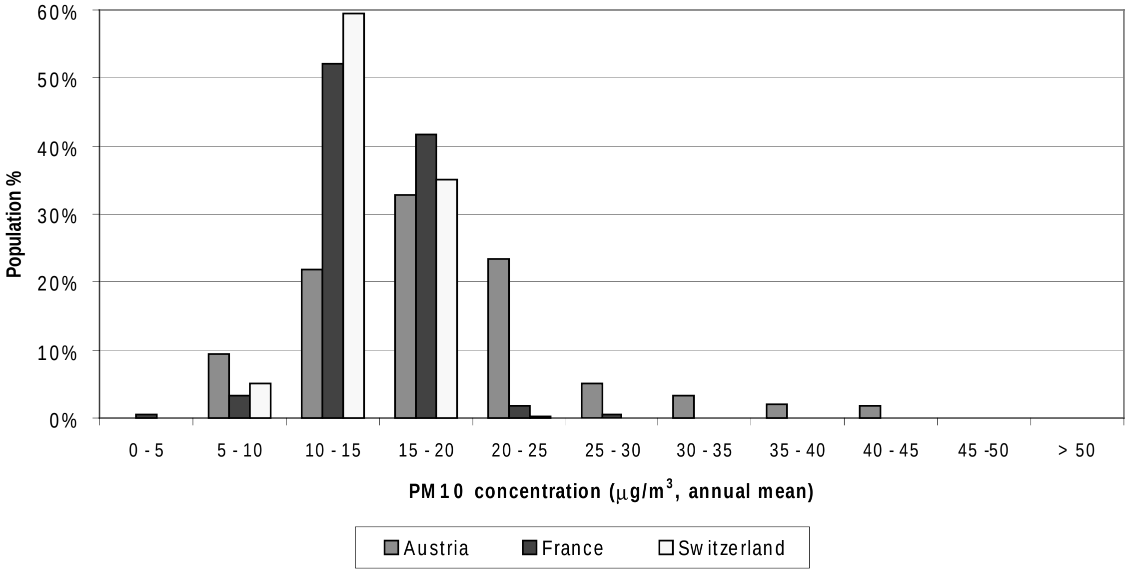


Figure 6. **Frequency distribution of PM_{10} population exposure without share attributable to road traffic**[10]



9 Filliger P., Puybonnieux-Texier V., Schneider J. (1999), Health Costs due to Road Traffic-related Air Pollution, PM_{10} Population Exposure, p. 10.

10 Filliger P., Puybonnieux-Texier V., Schneider J. (1999), Health Costs due to Road Traffic-related Air Pollution, PM_{10} Population Exposure, p. 10.

The population exposure without PM_{10} fraction attributable to road traffic is shown in Figure 6. Compared to total PM_{10}, the frequency distribution changes considerably. Most people would live in areas with PM_{10} values less than 20 μg/m³. In France and Switzerland, less than 3% of the population would live in areas with PM_{10} greater than 20 μg/m³. In Austria, this portion is higher due to an increased non-traffic caused regional PM_{10} background. However, in all three countries, the reduction of the percent values in higher PM_{10} concentration classes is substantial and indicates that road traffic contributes considerably to these PM_{10} concentration classes.

Population weighted PM_{10} averages are summarised in Table 7. Interpreting the figures one has to be aware of the fact that PM_{10} due to road traffic varies considerably spatially. In city centres, the relative contribution of road traffic to total PM_{10} is higher than in rural areas. Typical values, derived from the Swiss model are: 40 - 60% in cities and < 30% in rural areas.

Table 7. **Population weighted annual PM_{10} averages for the three countries (calculated from the original grid values of the PM_{10} maps)**[11]

	PM10 concentration in μg/m³ (annual mean)		
	Austria	France	Switzerland
Total PM_{10}	26.0	23.5	21.4
PM10 without fraction attributable to road traffic	18.0	14.6	14.0
PM10 due to road traffic	8.0	8.9	7.4

Despite the different methods used, the **results of the three countries are similar**, especially concerning PM_{10} levels caused by road traffic. The differences in total PM_{10} can be explained by the fact that (a) the background concentration is higher in Central and Eastern Europe than in the Western parts of Europe and (b) for Switzerland, large areas at higher altitudes have significantly lower PM_{10} levels. Furthermore, the sulphate fraction of the background concentration may increase from Western to Eastern Europe, resulting in an increase of the non-traffic related PM_{10} fraction. However, further investigations including measurements of PM_{10} as well as PM_{10} components are needed to explore in detail the significance of the differences found.

4. The monetary valuation of air pollution related health effects

Monetarising health effects or even fatalities is often criticised outside the community of economic science. In the general public’s opinion it is argued, that human life cannot be expressed in monetary terms. This criticism is based on a misunderstanding as the economic science does not try to assess the value of a specific life. What is being measured is the benefit of a risk reduction due to a lower level of air pollution leading to a decrease in frequency of the different health outcomes.

[11] Filliger P., Puybonnieux-Texier V., Schneider J. (1999) Health Costs due to Road Traffic-related Air Pollution, PM_{10} Population Exposure, p. 11.

For this type of assessment, the term „value of preventing a statistical fatality” (VPF) is often used in economic theory. It reflects the fact that a decrease in risk is valued before the negative results have already taken place. Hence, it dos not value „ex post” a specific human being’s life lost due to an air pollution related disease.

4.1 Monetary Evaluation of Mortality

There are two main different approaches to asses the monetary value of mortality[12]:

- **The gross production / consumption loss:** The costs of additional mortality cases are assessed according to the loss in income / production or the loss of consumption. This valuation concept - sometimes refered as discounted future earnings - is based on the loss resulting from a premature death for the economy as a whole. It is a concept based on the general society, without regarding the individual difference in valuing lower or higher risks of mortality or fatal accidents. The measurement is limited to material aspects of life only, it neglects the intangible costs such as pain, grief and suffering of the victims and their relatives. The main advantage of this approach lies in its simple and transparent calculation concept. Therefore it may be a suitable input for political discussions on policy measures for a reduction of air pollution or other environmental impacts.

However, the main disadvantages are the following:

- The individual aversion against premature death is not considered in this approach, since it only covers material consequences of a fatality.
- Based on the loss for the society as a whole, the concept is in conflict with a basic principle of (welfare-) economic theory according to which each valuation has to be based on the variations in the utility of the concerned individuals.
- An appropriate discount rate has to be chosen which has major implications for the valuation.

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12 For a detailed discussion see: Sommer H., Seethaler R., Chanel O., Herry M., Masson S., Vergnaud J.-Ch. (1999), Health Costs due to Road Traffic-related Air Pollution, p. 22-26.

- **Willingness to pay (WTP) / Value of preventing a statistical fatality (VPF):** This approach attempts to estimate the demand (the willingness-to-pay) for an improved environmental quality. The central question is, how much individuals are ready to pay to improve their own security or the security of other people. Thus, the sum of individual willingness-to-pay indicates how much value is attributed to an improvement in security or a reduction of environmental impact by the society as a whole. The valuation of a risk reduction in mortality or the value of preventing a "statistical" fatality is calculated by dividing the individual willingness-to-pay values for a risk reduction by the observed change in risk.[13]

The main advantage of the willingness-to-pay approach lies in evaluating the individual preferences for risk reductions of morbidity and premature fatalities. It therefore meets the requirements of welfare economics, since it reflects the individual point of view.

However, a number of arguments against this method are often raised:

- The willingness-to-pay approach depends on the level of income which may pose ethical problems when applied to very different countries (OECD vs. less developed countries).

- If part of income losses are borne by the social insurance system of the country, this loss will not be considered by the individual, although it is part of the society's costs.

- It is often difficult for the individual to be sufficiently aware of the risk level at stake and the consequences on health. Individuals may not be familiar with small variations in risk which may imply large discrepancies between individual valuations.

- The main difficulty of the WTP approach lies in obtaining reliable and correct empirical estimations, because results are highly sensitive to the survey design.

Nevertheless, recent research provides promising results. The chosen WTP values for the present study are based on a contingent valuation method, in which the direct comparison between money and risk of mortality is replaced by a sequence of chained interviews.[14]

Based on this discussion the Willingness-to-pay (WTP) for the Value of a Prevented Fatality (**VPF**) was used as common methodological approach.[15]

Unfortunately, so far no empirical studies have been carried out specifically for air pollution related mortality risk. Furthermore, under the prevailing budget and time constraint it was out of scope to conduct an empirical survey within this project. Therefore, empirical results of road accident related WTP were used as a starting point.

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13 Example: A policy measure is able to reduce the yearly risk of fatal road accidents from 4 cases per 10'000 to 3 cases per 10'000. For this risk reduction of 1 case per 10'000, the affected individuals are ready to pay an average amount of 100 US $. In this case, the value of a statistical prevented fatality amounts to 1 million US $ (100 US $ /0.0001 risk reduction). Again, it needs to be recognised that the respondents are not asked about their willingness-to-pay for the avoidance of their own death but about the willingness-to-pay for a change in risk.

14 See Sommer H., Seethaler R., Chanel O., Herry M., Massons S., Vergnaud J.-Ch. (1999), Health Costs due to Road Traffic-related Air Pollution, Annex 7 p. 77-83.

15 According to the country specific needs, in addition to the WTP-approach an alternative partial assessment approach was conducted, based on the loss of production or consumption (see chapter 5.3).

The most recent studies from the 1990's indicate a range of WTP values for the prevention of a statistical fatality of 0.7 to 6.1 million EUR.[16] The latest empirical study, conducted by Jones-Lee *et al.*[17] provides a VPF of 1.42 million EUR (range: 0.7-2.3 million EUR).

Based on these most recent results and the experience of former studies **a starting value of 1.4 million EUR** is adopted for the value of preventing a statistical fatality. This choice is supported by the use of a similar starting value (1.2 million EUR) in a recent study on behalf of the UK Department for Environment, Transport and Regions (DETR) and the fact that it lies in the lower part of the range of the majority of recent empirical evaluations.[18] This choice is in line with the "at least" approach prevailing throughout the entire project.

Road accident related fatal risk differs from air pollution related risk. The latter is to a large extent involuntary and beyond the responsibility and control of those exposed to it. In addition, while taking the risk of a traffic accident, driving itself offers a direct personal benefit. On the other hand, air pollution related risk is less often connected to a direct personal benefit, although it is to some extent transport induced. Because of this **different risk context**, air pollution related risk aversion is likely to be higher than for fatal road accidents.[19] The impact of the contextual difference between road accident and air pollution related risk on individual aversion is subject of several empirical studies and has produced factors in the range of 1.5 to 2. However, the empirical evidence is not considered to be sufficient and following the "at least" approach, the contextual adaptation of the WTP value is abandoned in the present study.

Based on the available epidemiological literature, a direct conclusion about the age structure of the air pollution related premature deaths is not yet possible. It is, however, known that these fatalities are mostly related to respiratory and cardiovascular diseases and lung cancer. In Austria, France and Switzerland, the average age of these respiratory and cardiovascular fatalities lies between 75 and 85 years (see Figure 8).

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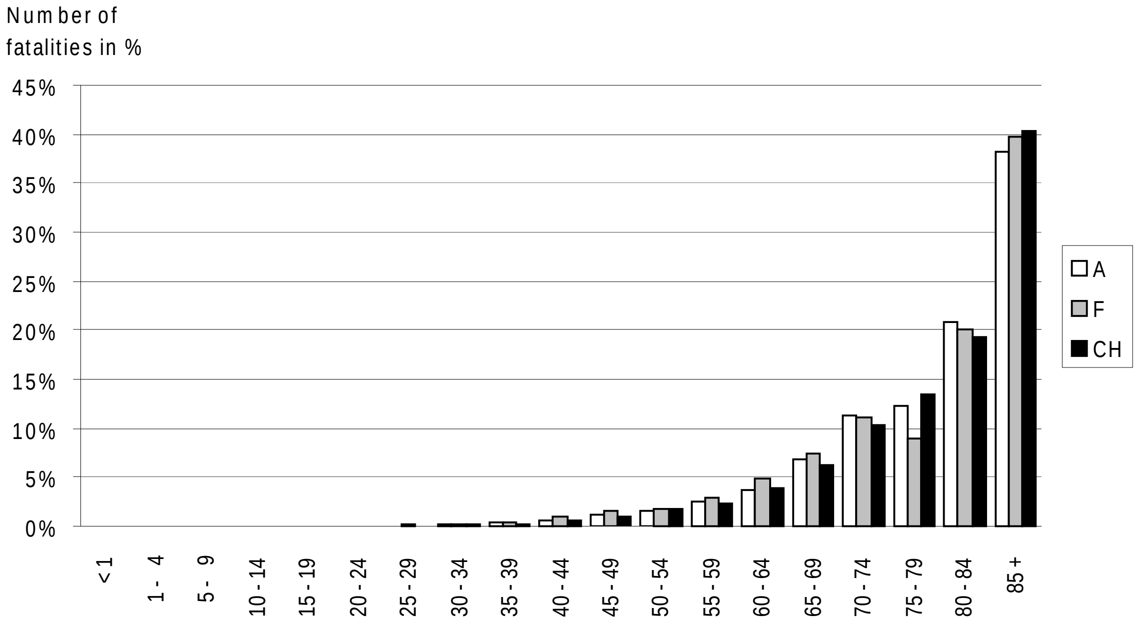
16 Viscusi W.K. (1993, The Value of Risks to Life and Health; Beattie J. et al (1998), Valuing Health and Safety Controls: A Literature Review; Institute of Environmental Studies, Norwegian Institute for Air Research, International Institute for Applied System Research (1997), Economic Evaluation of quality targets for sulphur dioxide, nitrogen dioxide, fine and suspended particulate matter and lead; ZEW / ISI (1997), External Quality Evaluation; ExternE (1995), Externalities of Energie, Vol 2: Methodology.

17 Jones-Lee M. et al (1998), On the Contingent Valuation of Safety and the Safety of Contingent Valuation: Part 2 - The CV/SG "Chained" Approach; Chilton S., Covey J., Lorraine H., Jones-Lee M., Loomes G., Pidgeon N., Spencer A. (1998), New Research Results on the Valuation of Preventing Fatal Road accident Casualties.

18 For example, the ExternE-Project, a very extensive project on behalf of the European Community on the external costs of energy use, is based on a meta-analytical value of 2.6 million Euro with a range from 2.1 to 3.0 million Euro. See: ExternE (1995), Externalities of Energy, Vol. 2: Methodology.

19 This view is adopted by a number of authors. See: Jones-Lee et al (1998), On the Contingent Valuation of Safety and the Safety of Contingent Valuation: Part 1 - Caveat Investigater and Department of Health (1999), Economic Appraisal of the Health Effects of Air Pollution, p. 63-66.

Figure 8. **Age structure of fatalities due to respiratory, cardiovascular diseases and lung cancer in Austria[20], France and Switzerland (1996)**



Hence, the average age of the air pollution related fatalities is much higher than for victims of fatal road accidents (30-40 years of age).

Theoretical as well as empirical evidence indicates a decreasing WTP with increasing age, with reduced remaining life expectancy and with reduced quality of life. For the present study, the relationship adopted is provided by the latest research of Jones-Lee.[21] Weighting the age structure of the fatalities due to respiratory and cardiovascular disease and lung cancer in all three countries by the age factor, an average adaptation factor of 61% is obtained for the present willingness-to-pay for a prevented fatality value.

Based on the preceding discussion we used a value of **0.9 Mio. EUR** (=61% x 1.4 Mio. EUR) for the value of preventing a statistical fatality. Hence, the cost reducing adjustment for age is maintained, meanwhile the cost increasing adjustment for the risk context is abandoned. This implies a very strict application of the "**at least**" **approach**.

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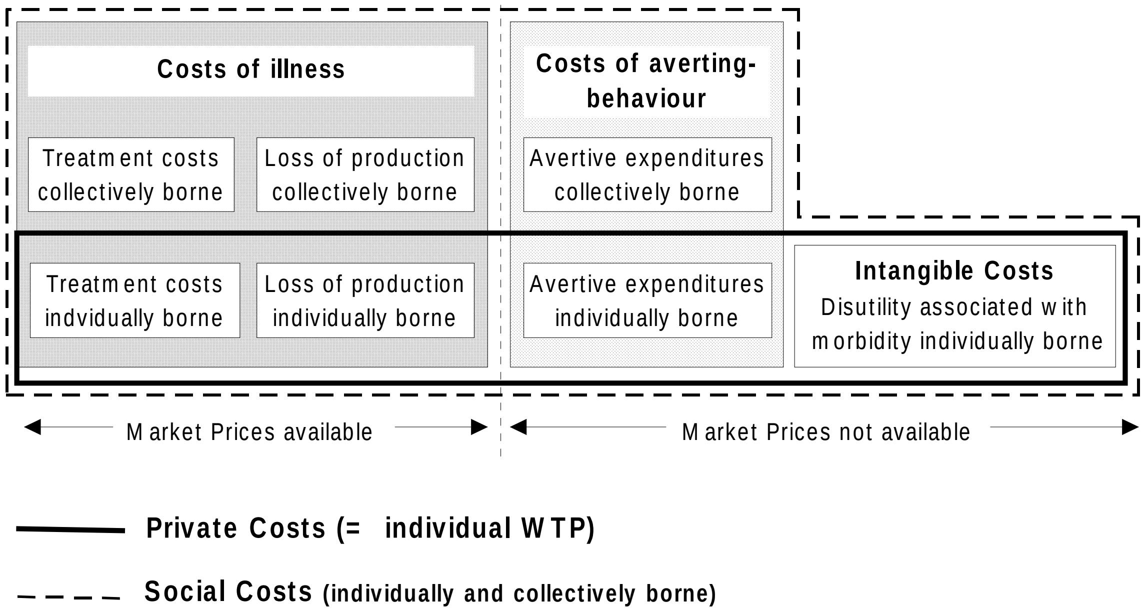
[20] Only respiratory and cardiovascular diseases without lung cancer.

[21] Several studies by Jones-Lee show a reversed U-shaped relationship between the age and the willingness-to-pay. See: Department of Health (1999), Economic Appraisal of the Health Effects of Air Pollution, p. 67 and direct information of M. Jones-Lee (1998).

4.2 Monetary valuation of morbidity

From an economic point of view, the costs of morbidity may be subdivided by two main criteria, namely by the cost component and by the entity in charge of paying them. As shown in Figure 9, the costs of illness, the costs of averting behaviour and the intangible costs are three different components. They are either borne privately or in the case of cost of illness and costs of averting behaviour collectively as well.

Figure 9. **Overview on the costs of morbidity**



Costs of illness (COI) contain the loss of production due to a possible incapacity to work and the medical treatment costs. They determine the “material part” of the health costs and may be assessed on the basis of real market prices (loss of earnings, costs for medicaments, costs per day in hospital, etc.).

Costs of averting-behaviour result from changes in behaviour due to air pollution. The abstention from outdoor sport activities during a summer day with high ozone concentration, the installation of air filters or a different choice of residential location to avoid high levels of air pollution are some current examples. The higher the costs of avoidance measures, the smaller will probably be the number of air pollution related morbidity cases. Considering the extent of avoidance measures taken so far, neglecting the costs of averting-behaviour may result in a considerable underestimation of the morbidity costs. However, for the assessment of these costs market prices are mostly non-existent.

The third essential component of morbidity costs are the **intangible costs** reflecting the individual loss of the victims utility and consisting of pain, grief and suffering due to a disease. Based on empirical evidence, the risk aversion of morbidity is mainly determined by these inconveniences (losses in utility).

In order to draw a complete picture of the total morbidity costs, individually borne private costs and the costs borne collectively, e.g. by a social security system, have to be considered. All components together constitute the social costs of morbidity.

Similar to the methodological possibilities for the monetary valuation of mortality, the morbidity may be assessed with different methodological approaches. For the costs of illness (COI) containing the production loss and medical treatment costs, the damage cost approach is used. Based on market prices, it assesses all individually as well as collectively borne material costs. However, for the costs of averting-behaviour and the intangible costs, this approach is not suitable, since market prices are mostly non-existent.

The willingness-to-pay approach focuses on the individually borne costs (private costs). It establishes the individuals utility of a risk reduction in air pollution related morbidity and reflects all costs the individual expects to bear in case of a disease, such as loss of earnings, costs of averting behaviour or intangible costs.

As mentioned above, the advantage of the willingness-to-pay approach consists of its integration of material and intangible costs, that cannot be measured by any other method but are often considerably higher than material costs. However, the disadvantage is its limitation to individually borne costs, especially when a large part of health costs is borne by collective means.

In spite of this limitation the willingness-to-pay approach is considered to be a better approximation of social costs of morbidity than the COI approach. Therefore we used the WTP-approach as the main common methodological framework to assess the morbidity costs.

Unfortunately, the literature on WTP based, air pollution related, morbidity costs is very rare in Europe and most available studies refer to the US context. Their application to Europe is not completely unproblematic, since a recent research study provide lower results for a European country.[22] The different socio-cultural background and the difference in health care and insurance systems ask for an application of country specific WTP results. In spite of this problem, the present study had to be based on existing values since the available resources did not allow for an empirical survey within this project.

Table 10 presents the WTP for avoiding different air pollution related health outcomes.

[22] Navrud S. (1998), Valuing Health Impacts from Air Pollution in Europe - New Empirical Evidence on Morbidity.

Table 10. **TP for the avoidance of air pollution related health outcomes**

Health indicator	**WTP-Value (EUR)**	
Respiratory Hospital Admission	7’870	per admission[23]
Cardiovascular Hospital Admission	7’870	per admission[23]
Chronic Bronchitis	209’000	per case[24]
Bronchitis	131	per case[25]
Restricted Activity Day	94	per day[25]
Asthmatics: Asthma Attacks (person day)	31	per attack[26]

5. Results

5.1 Quantitative results of PM_{10} related health effects

From the epidemiological data (fixed base line increment per 10 $\mu g/m^3$ PM_{10} per 1 million inhabitant) on the one hand and the average exposure level of the population on the other hand, the number of health outcomes can be determined.

These calculations may be done for the current exposure to particulate matter as well as for a hypothetical situation **without** road traffic-related air pollution. The difference between the two results corresponds to the number of morbidity and mortality cases attributable to road traffic-related air pollution.

In Table 11 for Austria, France and Switzerland, the health effects considered are presented for the average annual exposure to total air pollution and for the average annual exposure to road traffic-related air pollution. According to the epidemiological foundations, for each health outcome the respective age group is considered. Knowing the distribution of the different population groups across exposure classes (chapter 3) and the parameters of the exposure-response function (chapter 2), the absolute number of health outcomes may be established for each country with or without the road traffic-related share of air pollution.

23 Based on ExternE (1995), Externalities of Energy, Volume 2, Methodology, Part II: Economic Valuation, p. 519, adjusted for inflation according to Nilsson M., Gullberg M. (1998), Externalities of Energy, Swedish Implementation of the ExternE Methodolgy.

24 Chestnut L.G. (1995), Human health benefits from sulfate reductions under Title IV of the 1990 clean air act amendments, p. 5-20, WTP for an average chronic bronchitis case.

25 Maddison D. (1997), Valuing the morbidity effects of air pollution, p. 8.

26 Ostro B., Chestnut L. (1997), Assessing the Health Benefits of Reducing Particulate Matter Air Pollution in The United States, p. 100.

It needs to be emphasized that the health effects are **only considered from the exposure class of 5-10 μg/m^3 PM$_{10}$ onwards** (average 7.5 μg/m^3 PM$_{10}$). This restriction reflects the fact that epidemiological studies have not yet included the exposure-response relationship below this level. In addition, it needs to be considered that there is a natural background concentration level which is not man made. For Austria, France and Switzerland this natural baseline pollutant level is estimated to be $<$7.5 μg/m^3 PM$_{10}$. For the further assessment of air pollution measures it is adequate to only consider the air pollution of human activities.

In Table 11, the negative effects of air pollution are divided into the number of health outcomes related to total air pollution and those related to the road traffic share only.

5.1.1 Mortality

In 1996, air pollution caused **5’600 cases of premature death** in Austria, **31’700 cases** in France and **3’300 cases** in Switzerland. In Austria 2’400, in France 17’600 and in Switzerland 1’800 cases are attributable to road traffic-related air pollution.

According to the epidemiological foundations, the increase in premature mortality is only considered for adults ≥30 years of age and for the exposure class of 5-10 μg/m^3 PM$_{10}$ (class mean 7.5 μg/m^3) onwards.

Table 11. **Additional cases of mortality and morbidity due to air pollution in Austria, France and Switzerland**[27]

Health outcome	Cases or days attributable to **total air pollution**			Cases or days attributable to **road traffic**		
	Austria	**France**	**Switzerland**	**Austria**	**France**	**Switzerland**
Long-term mortality (adults ≥30 years; excluding violent death)	**5'600** 3'400 - 7'800	**31'700** 19'200 - 44'400	**3'300** 2'000 - 4'700	**2'400** 1'500 - 3'400	**17'600** 10'700 - 24'700	**1'800** 1'100 - 2'500
Respiratory hospital admissions (all ages)	**3'400** 400 - 6'500	**13'800** 1'400 - 26'300	**1'300** 140 - 2'500	**1'500** 160 - 2'800	**7'700** 800 - 14'600	**700** 70 - 1'300
Cardiovascular hospital admissions (all ages)	**6'700** 3'500 - 10'000	**19'800** 10'400 - 29'400	**3'000** 1'500 - 4'400	**2'900** 1'500 - 4'300	**11'000** 5'800 - 16'300	**1'600** 800 - 2'400
Chronic bronchitis incidence (adults ≥25 years)	**6'200** 600 - 12'000	**36'700** 3'300 - 73'100	**4'200** 370 - 8'400	**2'700** 240 - 5'300	**20'400** 1'800 - 40'700	**2'300** 200 - 4'500
Bronchitis (children < 15 years)	**48'000** 21'000 - 86'000	**450'000** 198'500 - 813'600	**45'000** 20'000 - 82'000	**21'000** 9'000 - 37'000	**250'000** 110'000 - 453'000	**24'000** 11'000 - 44'000
Restricted activity days (adults ≥20 years)	**3'100'000** 2'600'000 - 3'600'000	**24'600'000** 20'700'000 - 28'500'000	**2'800'000** 2'400'000 - 3'200'000	**1'300'000** 1'100'000 - 1'600'000	**13'700'000** 11'500'000 - 15'900'000	**1'500'000** 1'200'000 - 1'700'000
Asthmatics: asthma attacks (children < 15 years, person days)	**35'000** 21'000 - 48'000	**243'000** 149'000 - 337'000	**24'000** 15'000 - 33'000	**15'000** 9'000 - 21'000	**135'000** 83'000 - 188'000	**13'000** 8'000 - 17'000
Asthmatics: asthma attacks (adults ≥15 years, person days)	**94'000** 46'000 - 143'000	**577'000** 281'000 - 879'000	**63'000** 30'000 - 95'000	**40'000** 20'000 - 62'000	**321'000** 155'000 - 489'000	**33'000** 16'000 - 51'000

[27] Table printed with permission from Lancet, Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.

5.1.2 Morbidity

Within the **additional morbidity cases**, the highest incidence in all three countries is registered for **acute bronchitis in children** younger than 15 years. Some 21’000 cases in Austria, some 250’000 cases in France and some 24’000 cases in Switzerland were attributable to road traffic-related air pollution in 1996.

The second highest frequency is obtained for the incidence of **chronic bronchitis in adults**. In 1996, the number attributable to road traffic-related air pollution amounts to ca 2’700 cases in Austria, 20’400 cases in France and 2’200 cases in Switzerland.

The additional cases of **cardiovascular hospital admissions** (all ages) due to road traffic-related air pollution amount to 2’900 cases in Austria, 11’000 cases in France and 1’600 cases in Switzerland. The smallest number of road traffic attributable cases is obtained for **respiratory hospital admissions** (all ages). In 1996, it amounts to ca 1’500 cases in Austria, 7’700 cases in France and 700 cases in Switzerland.

Concerning the **additional days** of air pollution related morbidity, a very large number of **restricted activity days** for adults (≥ 20 years) results in all three countries. In 1996, in Austria, 1.3 million days, in France 13.7 million days and in Switzerland 1.5 million days with restricted activity are attributed due to road-traffic-related air pollution.

In 1996, for Austria 15’000 **asthma attacks in children** (<15 years) and **40’000 asthma attacks in adults** (≥ 15 years) are attributable to road traffic-related air pollution. France and Switzerland attributed 135’000 and 13’000 asthma attacks in children and 321’000 and 33’000 asthma attacks in adults to road traffic-related air pollution.

5.2 Health costs due to air pollution based on the willingness-to-pay approach

Based on the **willingness-to-pay** approach, in 1996 the total air pollution in Austria, France and Switzerland caused a high level of health costs. The **total air pollution related health costs** across the three countries amount to **49’700 million EUR** (Table 12), of which **26’700 million EUR** are attributable to **road traffic-related air pollution**.

In Austria (6’700 million EUR) and Switzerland (4’200 million EUR) the **total air pollution** related health costs reach a similar level. Due to the much larger population, the French costs amount to 38’800 million EUR.

Table 12. **Health costs due to road traffic-related air pollution in Austria, France and Switzerland based on the willingness-to-pay approach (1996)**

	Austria		**France**		**Switzerland**	
	Total costs with road traffic share	Costs attributable to road	Total costs with road traffic share	Costs attributable to road	Total costs with road traffic share	Costs attributable to road
Costs of mortality (million EUR)	5'000 3'000 - 7'000	2'200 1'300 - 3'000	28'500 17'300 - 39'900	15'900 9'600 - 22'200	3'000 1'800 - 4'200	1'600 1'000 - 2'200
Costs of morbidity (million EUR)	1'700 400 - 3'000	700 200 - 1'300	10'300 2'800 - 18'500	5'700 1'500 - 10'300	1'200 300 - 2'100	600 200 - 1'100
Total costs (million EUR)	**6'700** 3'400 - 10'000	**2'900** 1'500 - 4'300	**38'800** 20'100 - 58'400	**21'600** 11'100 - 32'500	**4'200** 2'100 - 6'300	**2'200** 1'200 - 3'300

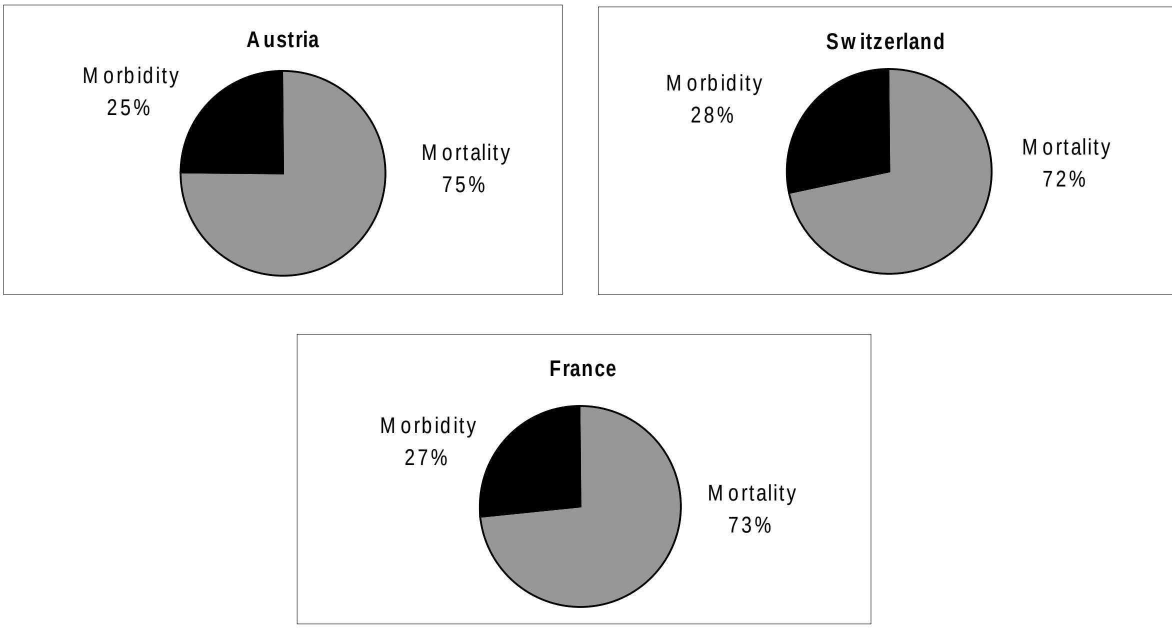
	all three countries	
	Total costs with road traffic share	Costs attributable to road
Costs of mortality (million EUR)	36'500 22'100 - 51'100	19'700 11'900 - 27'400
Costs of morbidity (million EUR)	13'200 3'500 - 23'600	7'000 1'900 - 12'700
Total costs (million EUR)	**49'700** 25'600 - 74'700	**26'700** 13'800 - 40'100

In all three countries, road traffic is a main source of air pollution related health costs. The absolute level of the road traffic-related air pollution amounts to 8.9 µg/m^3 PM_{10} in France, 8.0 µg/m^3 in Austria and of 7.4 µg/m^3 in Switzerland (as population weighted annual averages). It needs to be remembered that tailpipe exhaust is only responsible for part of the PM_{10} concentration. The considerable proportion of other emissions, such as tyre wear, other abrasion products and road dust re-suspension are independent of the share of diesel engines.

The lower relative proportion of traffic-related health costs in Austria may be caused by a higher background of PM_{10} in 1996 which may contain a high sulphate amount (especially in Eastern Austria).

Depending on the country, 72% to 75% of the health costs are related to mortality (see Figure 13). The differences are mainly due to country specific differences in the baseline frequencies of the health outcomes observed.

Figure 13. **Breakdown of air pollution related costs by mortality and morbidity**

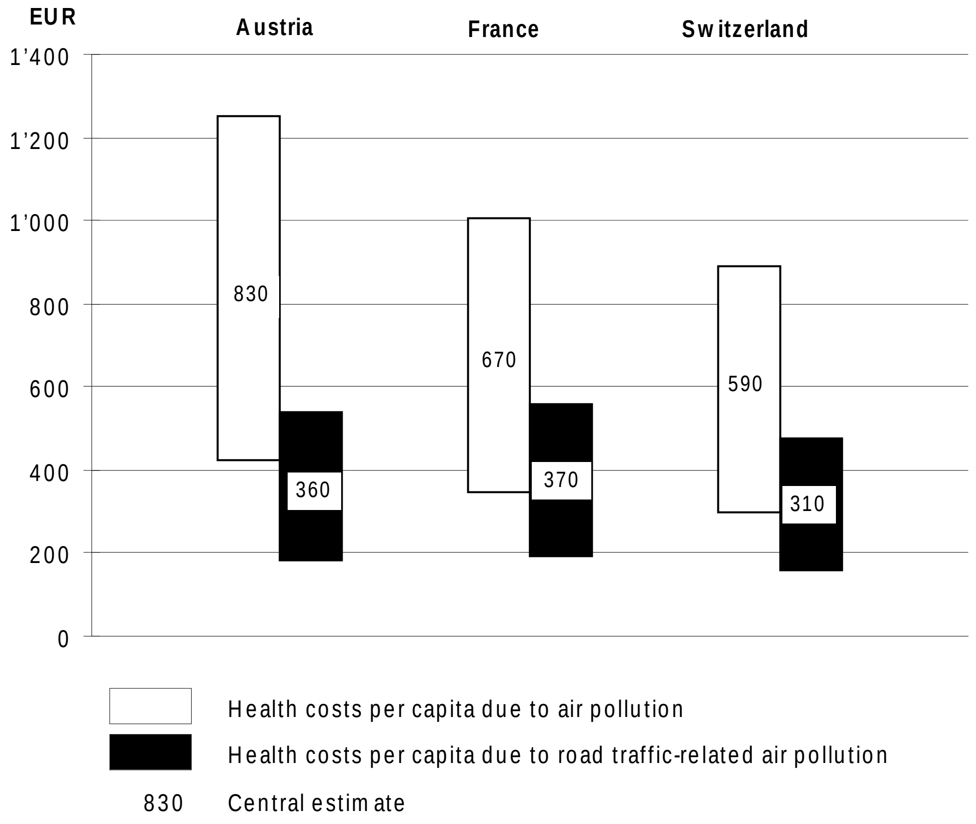


Comparing the **total air pollution related health costs per capita** (see Figure 14) the results of the three countries stay within the same range, **although** the central estimates indicate differences between the three countries. The highest per capita costs are shown for Austria.

For the **road traffic-related health costs**, the per capita results differ much less between the three countries: The highest value is obtained in France with about 370 EUR per capita, followed by Austria with about 360 EUR per capita and Switzerland with about 310 EUR per capita.

These differences are mainly due to air pollution levels (average level of population weighted total PM_{10} exposure and the traffic-related share) and the epidemiological results (different national mortality and morbidity rates in general). However, the results of the three countries stay within the same range. Therefore, the differences in per capita costs mentioned above should not be overinterpreted.

Figure 14. **Air pollution related health costs per capita based on the willingness-to-pay approach (1996)**



5.3 Partial assessment approach: health costs due to air pollution based on gross production loss approach / cost of illness (COI)

According to the country specific needs, in addition to the WTP-approach a partial assessment approach has been used to evaluate the health costs:

- The mortality related health costs are based on the production/consumption loss. The losses are determined on the potential years of life lost.

- The morbidity related health costs are based on the costs of illness, which consist of the production loss due to a incapacity of work and the medical treatment costs.

The partial assessment approach is an extreme implementation of the "at least" approach in so far, as it does not include a major aspect of mortality and morbidity risk related costs, namely the intangible costs. In addition, for some health outcomes (chronic bronchitis, asthma attacks) only the medical treatment costs are included, as for the production loss related to these health outcomes, no data is presently available. In absence of empirical data, for the very great number of restricted activity days no costs of production loss and medical treatment could be established at all.

The per capita costs of the partial assessment approach are shown in Table 15.

Table 15. **Air pollution related health costs per capita based on the partial assessment approach (1996)**

	Austria		France		Switzerland	
	Total costs with road traffic share	Costs attributable to road traffic	Total costs with road traffic share	Costs attributable to road traffic	Total costs with road traffic share	Costs attributable to road traffic
Costs per capita (EUR)	**140** 80 - 190	**60** 30 - 80	**70** 40 - 100	**40** 20 - 60	**160** 100 - 230	**90** 50 - 120

All the above mentioned restrictions for the assessment of health costs due to air pollution reduces the costs by a factor of 3.6 (in Switzerland) up to a factor of 9.1 (in France) compare to the willingness to pay based results.

The differences between the countries are mainly based on the country specific calculation methods. Different cost levels for the production or consumption loss approach have been used: 18'230 EUR per year of life lost in Austria, 12'600 in France and 34'800 in Switzerland.[28] The use of the same valuation per year of life lost for the three countries would have suppressed most of the differences in relative ratios between WTP and partial assessment results.

5.4 Interpretation and sensitivity of the results

For the assessment of air pollution related health costs, different methodological approaches are available. For an integral view, considering the material and intangible costs, the willingness-to-pay approach for the monetary valuation of mortality and morbidity costs comes to the fore.

Based on this approach, the results may be interpreted as follows:

- In all three countries, road traffic is a main source of air pollution related health costs. The absolute level of the road traffic-related costs stay within the same range: 0.9%-2.7% of the GDP in France, 0.8%-2.5% in Austria and 0.6%-1.7% in Switzerland.
- Compared to other road traffic-related negative impacts (noise, accidents, damage to buildings), the health costs are considerable. According to comparative studies in Austria and Switzerland, the health costs exceed the present estimations of accident costs.

28 The Swiss value contains an amount of 14'200 EUR per year of life lost as a low and insufficient proxy for the intangible costs. The proxy is based on compensation payments granted by courts.

- Based on the actual air pollution, a reduction in the average PM_{10} exposure of 10 µg/m^3 would result in the long run in a annual cost reduction of 3’600 million EUR in Austria, 24’300 million EUR in France and 3’000 million EUR in Switzerland. However, it needs to be borne in mind that the health costs (assessed by the willingness-to-pay approach) are mostly borne by individuals through welfare losses and intangible costs. Therefore, the cost savings due to a reduction of air pollution don’t result in a similar reduction of the health budget covered by the social insurance system.

- The cost reduction has to be seen as a long-term effect and that the savings during transition years would be less.

The sensitivity of the overall results is influenced by all three partial steps (the assessment of exposure, the exposure-response relationship for mortality and morbidity, the monetary valuation of mortality and morbidity related risk). The impact of key assumptions and methodological decisions has been quantified in the health impact paper[29], and discussed in more detail in our full reports.[30] In general, for each sensitive assumption an “at least” approach was adopted. The real costs of (road traffic-related) air pollution are considered to be higher than the results of the present study, since:

- various PM_{10} related health effects (e.g. infant mortality) were not considered due to the absence of available data;

- the additional effects of other pollutants (e.g. ozone) were not considered;

- for the monetary valuation generally lower estimates of cost factors were chosen.

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29 Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in Lancet, in press.

30 Filliger P., Puybonnieux-Texier V., Schneider J. (1999), Health Costs due to Road Traffic-related Air Pollution, PM_{10} Population Exposure; Künzli N., Kaiser R., Medina S., Studnicka M., Oberfeld G., Horak F. (1999); Health Costs due to Road Traffic-related Air Pollution, Air Pollution Attributable Cases; Sommer H., Seethaler R., Chanel O., Herry M., Massons S., Vergnaud J.-Ch. (1999), Health Costs due to Road Traffic-related Air Pollution; see also Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.

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