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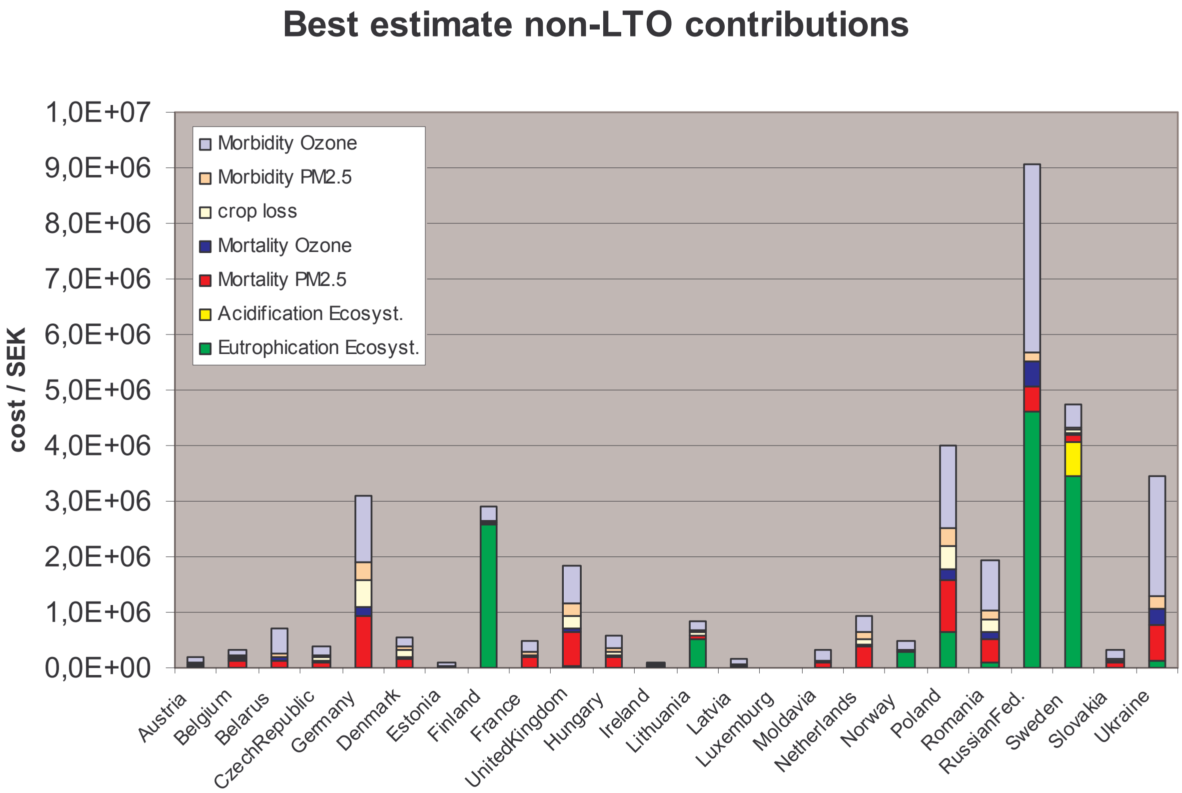

Final report of the project:

Economic valuation of environmental effects of NOx-emissions from air traffic at different altitudes

(Ekonomisk värdering av miljöeffekter av NOx-utsläpp från flygtrafik på olika höjd)

Commissioned by:

The Swedish Civil Aviation Authority (SCAA) and the Swedish Institute For Transport and Communications Analysis (SIKA)

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1. Summary

This report presents the final results of the project “Economic valuation of environmental effects of NOx-emissions from air traffic at different altitudes”. Estimates of the environmental costs due to NOx emissions from aviation in Swedish air space have been made using the impact pathway approach.

Marginal costs for NOx emissions from aviation are estimated to be higher for LTO emissions (ca 15 SEK/kg NOx) than for non-LTO emissions (ca 7 SEK/kg). The costs due to acidification and eutrophication of natural ecosystems dominate but are very uncertain. If ecosystem effects are excluded the marginal costs become almost the same for LTO (2.4-5.7 SEK/kg) and non-LTO emissions (1.9-5.1 SEK/kg) of NOx. These costs are mainly due to health effects due to particles (PM) and ozone. Costs due to crop loss and effects on agricultural soils were estimated to be much smaller.

Comparison with earlier studies for LTO emissions in Sweden show that the results are comparable for effects on ecosystems and agriculture, as well as for health effects due to ozone, provided that the same exposure response relations and valuation is applied. The estimated costs due to health effects of PM are however substantially lower than earlier estimates.

2. Goals and deliverables

The study included three parts:

1. A workshop where underlying information and methods for economic valuation of environmental effects were discussed to form a first assessment of the environmental costs due to NOx-emissions from the air traffic in Sweden on the European scale.

2. Preparation of a spreadsheet application where the information in 1) is included and where the sensitivity of different parameters can be studied.

3. Comparison with earlier estimates of environmental costs of NOx from air traffic in Sweden, analysis and reporting.

3. Background

The prevailing Swedish transport policy implies that the external marginal costs of transport should be the basis for taxes and charges. The purpose is to base efficient infrastructure charges on information about the marginal costs. The external marginal costs include the cost of the wear and tear of infrastructure, environment, noise, accidents and congestion.

The Swedish Civil Aviation Administration (LFV, luftfartsverket) has been given the task to investigate marginal cost pricing of environmental effects of air traffic. On January 1st 2005 the Swedish Civil Aviation Administration was divided and a new authority, the Swedish Civil Aviation Authority (SCAA) was formed. SCAA carries on the work related to pricing of environmental effects due to aviation.

SCAA and other European aviation authorities, within the ECAC (European Civil Aviation Conference) organization, have created a harmonised model for calculation of emissions from aircraft, the ERLIG model. ERLIG generates an emission index, in units of kg NOx emitted, based on aircraft certification data and a standardised Landing and Take-Off (LTO) cycle. ERLIG is based only on emissions during the LTO-cycle. The ERLIG charging scheme aims at addressing local air quality issues. In line with the prevailing Swedish transport policy the Swedish Civil Aviation Administration introduced ERLIG in the national system of aviation emission charges in April 2004 (LFV, 2004).

Since only part of the aircraft emissions are released during the LTO-cycle there is an interest in investigating the influence of higher altitude emissions on different local and regional environmental problems. If the contributions from non-LTO emissions to local and/or regional environmental problems are significant it is reasonable to introduce NOx emission charges also for these emissions.

SCAA and formerly LFV have therefore initiated a series of studies aimed at providing the information needed for an economic valuation of the environmental effects caused by aircraft emissions in Sweden.

The present study provides estimates of the environmental costs due to the impact of NOx emissions from aviation in Swedish air space on surface concentrations of ozone (O_3), fine particles (PM2.5) and deposition of oxidised nitrogen, on the European scale. The study is based on two previous studies (Langner et al., 2004a, Langner et al., 2004b) developing detailed emission data from aviation in Swedish air space and model calculations on the European scale using a regional transport-chemistry model. The underlying data in terms of exposure data, exposure-response relationships and valuation of impacts were discussed at a workshop in Stockholm 2005-02-08. This report summarises the outcome of the workshop and documents the information put into a spreadsheet application that can be used to estimate the environmental costs due to aviation emissions of NOx in Sweden.

4. Application of the impact pathway method

The ExternE family of research projects has developed the Impact Pathway (IP) Method for the assessment of external impacts and associated costs resulting from the supply and use of energy. The methodology is described in detail in several publications, e.g. Friedrich and Bickel (2001, from now on referred to as FB2001). An impact pathway is the sequence of events linking a burden or emission to an impact and subsequent valuation. The Impact Pathway Method includes four phases for assessing the damages caused by atmospheric emissions related to energy production or transport. The phases consist of making an inventory of emissions, modelling the dispersion of the pollutants, estimating physical impacts caused by pollutant concentrations by empirically defined exposure-response relationships, and carrying out a monetary valuation of the physical damages (Figure 1). Recent developments in environmental science (e.g. epidemiology) and economics (valuation techniques) and improvements in computing power (dispersion modelling) have allowed developing the Impact Pathway Method into a realistic, logical and transparent approach.

In this study the Impact Pathway method has been used to assess the environmental costs due to NOx-emissions from aviation in Swedish air space. The study is focused on valuation of the costs on the regional scale following the IP approach. There were several reasons for making this choice: a) Valuation of costs due to aircraft emissions on the local scale has recently been performed for a Swedish airport (Otterström et al., 2003), The local scale study indicated the importance of the costs due to regional scale dispersion of the emissions, b) The non-LTO emissions are released high above ground and are therefore dispersed and transformed on the regional scale before reaching the surface of the earth and there causing damage.

Information about aviation emissions in Swedish airspace and the dispersion, chemical interactions and deposition has been derived in two preceding studies. Details regarding emission modelling and dispersion modelling are given in (Langner et al., 2004a, Langner et al., 2004b).

An important part of the present study was the development of a spreadsheet application in which all the necessary information for the assessment of the environmental costs from aviation emissions of NOx in Swedish airspace can be calculated. In the design of the present study it was decided that aggregation of the assessment on a country by country basis would be a sufficient first step. The spreadsheet application is therefore built on national data and nation average results.

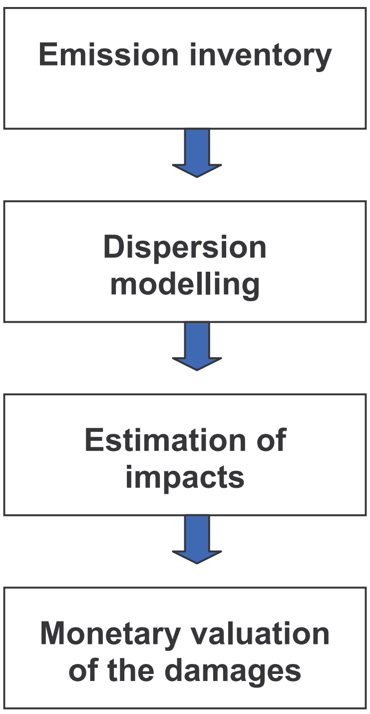

Figure 1. Schematic view of the Impact Pathway approach.

5. Emission inventory

The emission inventory is closely linked to the dispersion model that is used. In this study a regional scale model is used to model the fate of the NOx-emissions in Swedish airspace. The model covers Europe and the north Atlantic in order to properly simulate the relevant atmospheric chemistry. This means that an emission inventory is needed, not only for the aviation emissions in Sweden, but also for all other emissions in the model domain.

5.1. Emission data for Europe and the north Atlantic

Anthropogenic surface emissions of sulphur dioxide (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO), ammonia (NH_3), volatile organic compounds (VOC) and particulate matter (PM), for Europe and surrounding areas, are reported by the nations under the Convention on Long Range Transport of Air Pollutants (CLRTAP) and are compiled by EMEP (Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air pollutants in Europe, EMEP, 2004). EMEP expert estimated emissions (Vestreng, 2004) for 2001 on a 50x50 km grid in 11 emission sectors were used in this study.

Aircraft emissions for Europe and the north Atlantic area were derived from the ANCAT/EC2 (Abatement of Nuisances Caused by Air Transport; Gardner, 1998) and DLR (Deutsches Zentrum für Luft und Raumfahrt; Schmitt and Brunner, 1997) inventories, assuming emission increases of 2.6% per year from 1991/1992. These emissions were used outside Swedish air space when simulating the effects of NOx emission from aviation in Sweden while the aviation emissions from (Langner et al., 2004a, Langner et al., 2004b) were used in Swedish air space.

5.2. Emission data for air traffic in Swedish air space

Emission data for air traffic in Swedish air space for the year 2002 were prepared in two preceding projects. The modelling carried out to derive the emission data used in this study is described in depth in Langner et al. (2004a, b). Detailed emission data for aviation in Swedish air space was derived. The horizontal resolution was 20x20 km and the vertical resolution 500 m. The emissions were divided into international, domestic and overflight traffic. Temporal variations were estimated from detailed data at Arlanda airport. Swedish aviation emissions for 2002 are given in Table 1 along with official emission data. The differences between the study figures and the official data are due to adjustments made to the method for calculating the official emissions, as discussed in Langner et al. (2004a).

Note that the official LTO-emissions for 19 LFV owned airports have been used in this study. This means that the total emissions in Swedish air space reported here are slightly different from the total emissions reported in Langner et al. (2004a) where also the LTO-emissions were modelled.

Table 1. Total aircraft emissions for 2002 in Swedish air space used in this study compared to official Swedish emissions reporting in 2002

Traffic		Fuel (ton)	NOx (ton)	CO (ton)	HC (ton)	SO_2 (ton)
LTO	Study/official	83 890	910	1125	143	83
non-LTO	Study	343 517	4866	3304	408	344
Total	Study	427 411	5778	4430	551	427
	Official	441 449	5862	4071	574	441
Difference %		-3.2	-1.4	8.8	-4.0	-3.2

6. Regional transport/chemistry/deposition modelling

The regional scale transport/chemistry/deposition model MATCH (Multi-scale Atmospheric Transport and Chemistry model) was used to model the fate and chemical interactions of the NOx emissions in Swedish airspace. MATCH is an Eulerian grid point model, which describes the physical and chemical processes that govern emissions, atmospheric transport and dispersion, chemical transformation and wet and dry deposition of pollutants. Based on meteorological data, emission data and chemical boundary conditions MATCH calculates the temporal evolution of concentrations and deposition, hour by hour, of all included chemical components for a selected period, e.g. a month or a year. Based on these hourly values all relevant measures of pollutant concentrations and deposition can be derived. The application of MATCH to model the dispersion of NOx emissions from aviation in Swedish airspace is described in detail in Langner et al. (2004b).

Model simulations for one year were performed to determine the contribution of NOx emissions from aviation in Swedish airspace to a wide range of pollutants and their deposition. Three model runs were performed: a base case with all emissions included, one case excluding LTO emissions and one excluding non-LTO emissions in Swedish air space. The difference between the base case and non-LTO and LTO cases respectively represent the contribution from non-LTO and LTO emissions of NOx to the concentration and deposition of various chemical compounds. The simulations were performed for the domain shown to the right in Figure 2, using meteorological data for the year 2000. Chemical boundary conditions for the simulations were taken from large-scale model simulations over the model domain to the left in Figure 2.

The impacts of aircraft NOx emissions on a wide range of chemical compounds were calculated in MATCH. Impacts and valuation has however only been assessed for surface ozone (O_3), surface concentration of fine particles (PM2.5[a]) and deposition of oxidised nitrogen.

It should be pointed out that only the impacts of *NOx* emissions have been investigated. All other aircraft emissions (sulphur, VOC, soot) have been kept constant in all three model simulations. This means that only part of the environmental impact of emissions

[a] PM2.5 = particulate matter with aerodynamic diameter less than 2.5 micrometers.

from air traffic in Sweden is included in this study. The sulphur emissions will mainly contribute to acidifying deposition and to PM-concentrations in air. VOC-emissions influence ozone and other oxidant concentrations and thereby, indirectly, also all of the other parameters.

Figures 3-6 show the calculated contributions of LTO and non-LTO NOx-emissions to the deposition of oxidised nitrogen, the annual average of the daily maximum 8-hour concentration of surface ozone, the accumulated exposure of surface ozone to vegetation, AOT40[b], and surface PM2.5 concentrations. The details in the calculated distributions are commented in Langner et al. (2004b).

In order to make a first order assessment of the impacts of Swedish aviation emissions the results presented in figures 3-6 were aggregated to averages per country. Tables with average values are presented in Langner et al. (2004b) and were integrated into the spreadsheet application used for economic assessment of environmental costs.

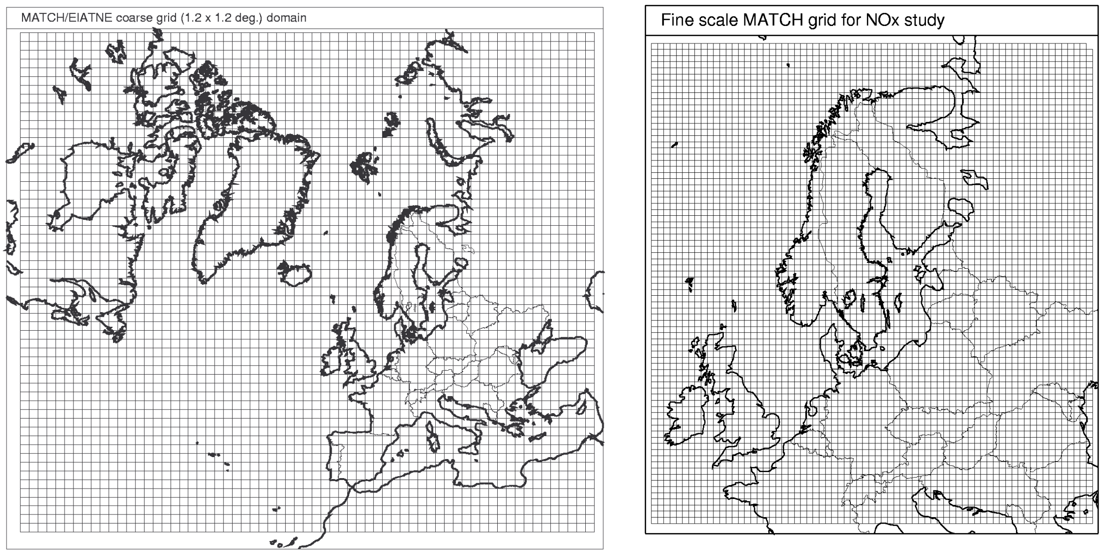

Figure 2. Modelling domains for MATCH

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[b] AOT40 (Accumulated Ozone exposure over Threshold 40 ppb(v)) 1 ppb(v) = 2 μg/m³

Definition: $$\int_{t=0}^{t=T} \max(c(O_3) - 40\ ppb, 0)\,dt$$

where $\max(x,0)=x$ if $x > 0$, $\max(x,0) = 0$ if $x < 0$

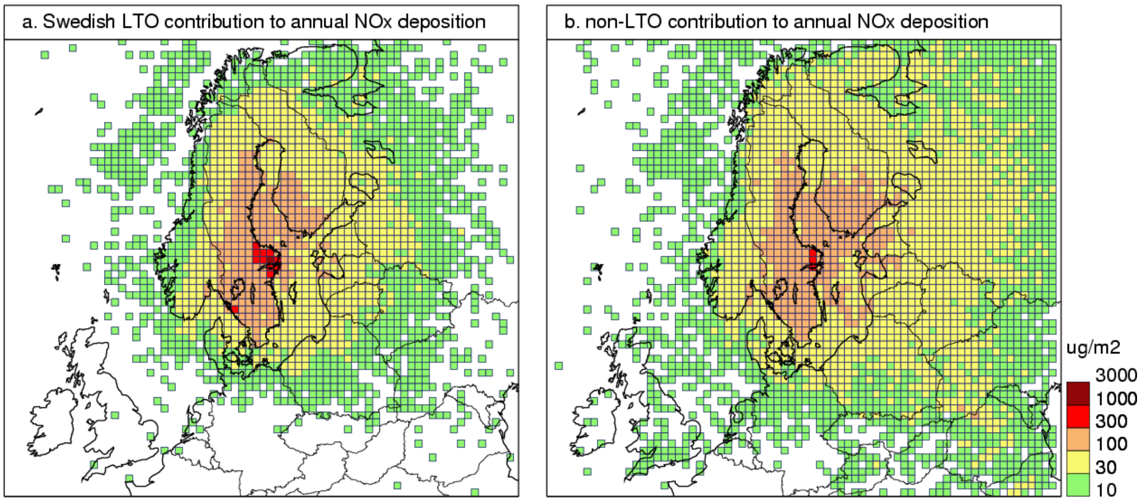

Figure 3. Model calculated contribution to the deposition of oxidised nitrogen for the year 2000 from LTO (a) and non-LTO (b) emissions of NOx in Swedish air space. Only absolute changes above 10 μg/m^2 are plotted. Unit: μg/m^2.

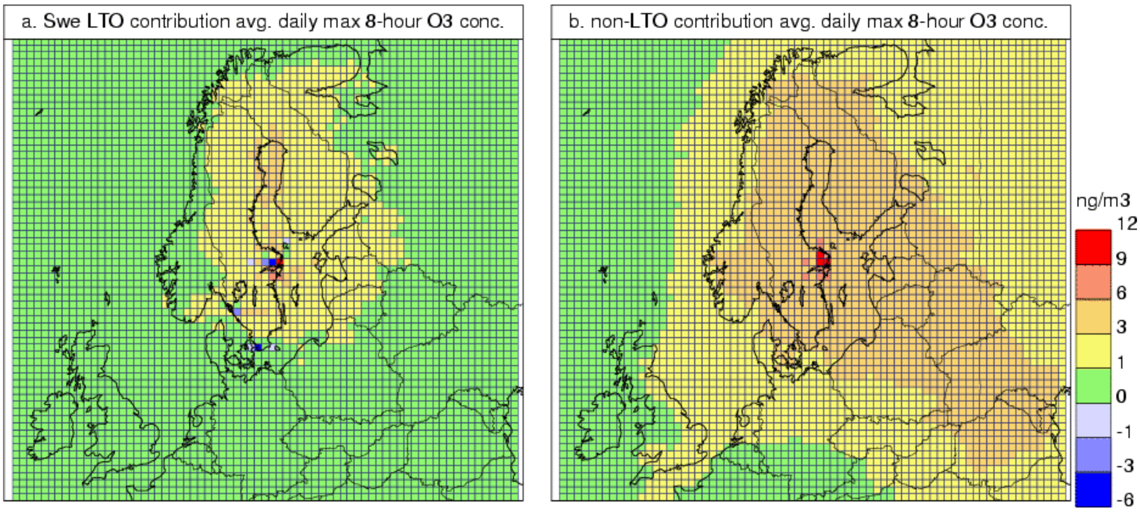

Figure 4. Model calculated annual average contribution to the daily maximum 8-hour average concentration of surface ozone for the year 2000 from LTO (a) and non-LTO (b) emissions of NOx in Swedish air space. Unit: ng/m^3 (at standard temperature and pressure (STP)). 1 ng/m^3 = 0.001 μg/m^3.

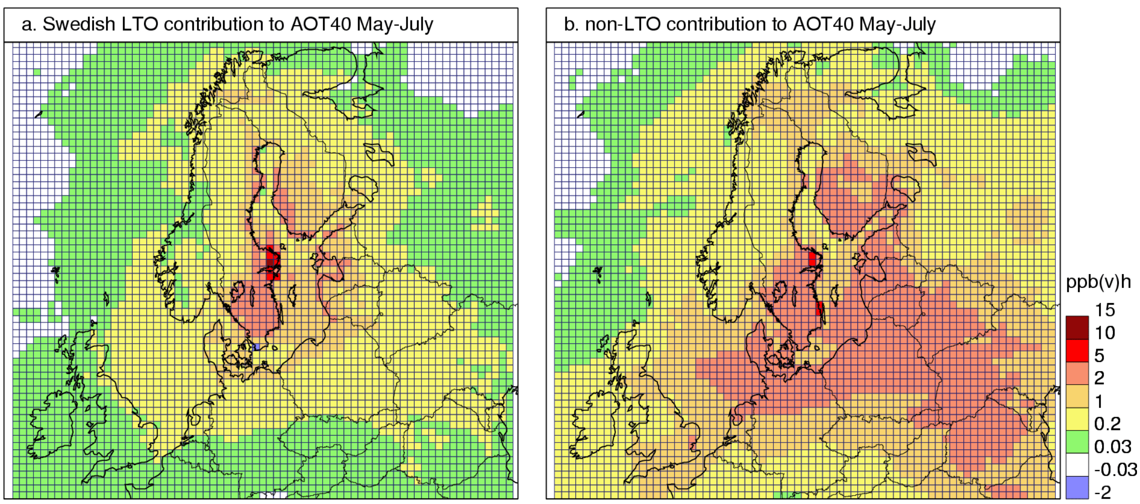

Figure 5. Model calculated contribution to AOT40 for May-July, 2000, from LTO (a) and non-LTO (b) emissions of NOx in Swedish air space. Units: ppb(v) hours.

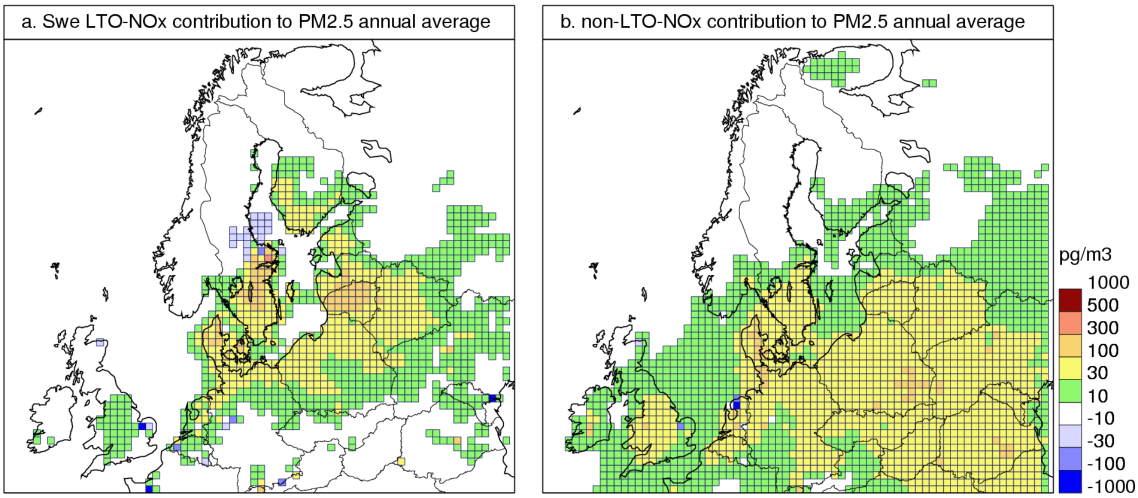

Figure 6. Model calculated annual average contribution to the surface PM2.5 concentration for the year 2000 from LTO (a) and non-LTO (b) emissions of NOx in Swedish air space. Only absolute changes above 10 pg/m^3 are plotted. Unit: pg/m^3 (STP). 1 pg/m^3 = 0.000001 μg/m^3.

7. Estimation of physical impacts

The ExternE family of research projects has compiled dose-response relationships for a range of different physical impacts (Friedrich and Bickel, 2001). The IP approach has been used previously in two studies on transport related emissions in Sweden (Bickel et al., 2002, from now on referred to as BI2002, and Otterström et al., 2003, from now on referred to as OT2003). Based on the results from these studies and the results reported in FB2001 the following impacts were judged to be most important and were included in this study:

- Impact on human health from surface ozone and PM.
- Impact on agricultural crops from surface ozone.
- Impact on agricultural soils and natural ecosystems from acidifying and eutrophying deposition of oxidised nitrogen.

Special emphasis was put on human health since this was known to be an important driver for the air quality policies developed by the European Commission through the Clean Air For Europe (CAFÉ) programme.

Known impacts that were not assessed include impacts on building materials through corrosion and soiling and impacts on crops from SO_2. The impact of NOx emissions on SO_2 concentrations is very small and was considered insignificant.

Impacts on climate related gases were excluded in the design of the project series. Climate effects of changes in tropospheric ozone and methane, due to Swedish air traffic NOx emissions, could be of significance but would require a global, or at least a hemispheric, atmospheric chemistry model for a proper assessment. Impacts on climate due to aerosol particles are very uncertain and are therefore very difficult to assess.

To be able to compare our results with those from previous studies we have estimated the different impacts based on the dose response functions used in the earlier ExternE studies for Sweden. Whenever possible we have tried also to provide an updated or best estimate assessment based on recent scientific findings.

7.1. Human health

In recent years health effects due to air pollution have come increasingly in focus. In the most recent WHO Air Quality Guidelines (WHO, 2000) the important step of abandoning the concept of a threshold concentration for health effects of particulate matter (PM) was taken. After this, additional air pollutants, like ozone, have been shown to have effects without a shown threshold at population level.

An increasing number of epidemiological studies have become available that can be used to estimate dose-response relationships that are necessary for calculating the resulting health effects. Toxicological studies are important primarily in order to understand causal relationships and mechanisms. The new epidemiological studies have prompted organisations like the US Environmental Protection Agency (US-EPA), EU and WHO to update their assessments of the knowledge base (WHO 2003; WHO 2004a; WHO, 2004b). The assessments are to an increasing degree built on so-called meta studies involving a thorough review of the scientific knowledge base.

7.1.1. Calculation of impacts

The basic principles for quantifying the health effects of air pollution exposure on the population has been described in several WHO reports (WHO 2000b; WHO 2001). The necessary calculations are based on the application of dose-response relationships to data on population exposure and usually also on the frequency of incidence of different health outcomes. The exposure-response (ER) relation describes how the concentrations of the pollutant affect the frequency of the studied effect (the cases). Exposure data describes the exposure as a distribution or as a total (time and/or population weighted) exposure. The frequency data gives the number of cases at the current level of exposure and is needed when using ER relations giving relative changes in the number of cases.

When estimating the health impacts we have chosen to present two different calculations one in which the ER functions given in FB2001 are used (this calculation is denoted ExternE) and an alternative updated calculation were ER functions are based on the latest WHO reviews (denoted Best estimate).

7.1.2. Impact on mortality

Effects on mortality are considered as very important and are weighted heavily in studies of societal cost. Up to date meta-analyses on the effect of PM and ozone on mortality are available from WHO as part of the "Systematic review of health aspects of air pollution in Europe", which was funded by the European Commission and was intended to provide input to the CAFÉ programme.

The analyses confirmed statistically significant relationships between levels of PM and ozone in ambient air and mortality, using data from several European cities. Updated risk coefficients, in relation to ambient *short-term* exposure to PM and ozone, were obtained for all-cause relative risk for a 10 µg/m^3 increase in PM10[c]: 0.6% (95% confidence interval: 0.4-0.8%) and ozone: 0.3% (95% confidence interval: 0.1-0.4%), respectively. The same coefficient for ozone (max daily 8-h and 1-h mean) has been found also in a recent analysis of 23 European cities in the APHEA2 study published after the WHO review.

For mortality due to *long-term* exposure of PM the recommended value for the relative risk of a 10 µg/m^3 increase in PM10 is 6%, based primarily on the study by Pope et al. (1995). This value is ten times higher than the value for short-term effects of PM. Our Best estimate for effects on mortality due to PM is based on the value for long-term effects. We have chosen to only include the long-term effect to avoid double counting.

For ozone we have only included short-term effects on mortality and use the value 0.3% in our Best estimate This is due to the limited number of European studies in the review using other endpoints.

The Swedish case studies in ExternE (BI2002) and also the study for Västerås airport (OT2003) have used the dose-response relationships from FB2001. The ER relationship for long-term effects of PM on mortality was taken from Pope et al, (1995) but transformed to allow for differences between European and US conditions. The coefficients used in BI2002 and OT2003 were 1.29% per 10 µg/m^3 for nitrate and 2.14% per 10 µg/m^3 for primary particles and sulphate (PM2.5). It should be noted here that OT2003 only apply the IP approach directly for local effects due to PM and ozone. The effects due to NOx emissions are only considered on the regional scale and then using average damage cost values from UNITE (2000).

The motives put forward in ExternE to correct the coefficients by Pope et al. (1995) do no longer hold but we have chosen to present calculations using the same values as BI2002 and OT2003, in order to be able to make comparisons with these studies.

For short-term effects on mortality FB2001 have used a value of 0.59% for a 10 µg/m^3 increase in ozone and we have used this value when making comparisons with earlier studies.

The calculations for mortality have been made only for persons older than 30 years, since the ER functions were derived for this group.

7.1.3. Impact on morbidity

The ExternE studies have, with varying degree of underpinning, estimated impacts on morbidity (Friedrich and Bickel, 2001). This was also done in the Swedish ExternE studies on transport (BI2002). Morbidity effects include chronic bronchitis, hospital admissions, cerebrovascular disease, heart failure and respiratory diseases (chronic cough, restricted activity). The valuation of these health impacts shows that the impacts on chronic bronchitis and hospital admissions are most important.

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[c] PM10 = particulate matter with aerodynamic diameter less than 10 micrometers. Several of the exposure-response relationships for PM are based on PM10 exposure rather than PM2.5 exposure. PM10 contains all particles in the PM2.5 fraction (plus particles with diameters between 2.5 and 10 µm). This means that changes in PM2.5 also contribute to changes in PM10. The calculated contribution to PM with diameter between 2.5 and 10 µm was negligible for aircraft NOx-emissions and, thus, the effects on PM2.5 and PM10 were essentially identical in this study.

The literature study performed by WHO for CAFE (WHO, 2003) suggest that the long-term effects on mortality are better characterized than effects on respiratory diseases: ”Long-term exposure to current ambient PM concentrations may lead to a marked reduction in life expectancy. The reduction in life expectancy is primarily due to increased cardiopulmonary and lung cancer mortality. Increases in lower respiratory symptoms and reduced lung function in children, and chronic obstructive pulmonary disease and reduced lung function in adults are likely”.

The exposure-response relationships used in the ExternE calculations regarding the effects of PM on hospital admissions can be questioned for several reasons. The ER relations have been taken from a limited number of studies and the base frequencies of incidence of hospital admissions used in the calculations were not taken from the nations or regions to which the calculations were applied. The scientific basis for using different coefficients for nitrates and PM10 and sulphates and PM2.5 can also be questioned.

There are new risk factors for hospital admissions due to respiratory diseases and hospital admissions for cerebrovascular diseases due to PM10 available from the EU-projects Air Pollution and Health: A European Information System (APHEIS) (www.apheis.net) and APHEA2 (Short-term effects of Air Pollution on Health: a European Approach using epidemiological time-series; LeTertre, 2003). For chronic bronchitis updated risk factors are lacking.

According to the recent WHO review there are few European ozone studies using other endpoints than daily number of deaths. A few studies on hospital admissions did not show a significant overall estimate in single pollution models, which may be a result of a negative correlation between ozone and primary combustion products. Neither did studies on admissions for asthma in children find conclusive associations, which may be explained by increased medication when ozone levels are high. Studies of ozone exposure and asthma incidence and prevalence in children and adults are not consistent. Available evidence suggests that long-term exposure possibly reduces lung function growth in children. There is little of support for an independent long-term effect of ozone on cancer or total mortality.

For the purpose of this study we have chosen not to update the ER relations for impacts on morbidity due to PM and ozone. Instead we indicate that the morbidity effects are in the range between zero and the ExternE estimates. This is in line with the assumptions made in the CAFÉ work where morbidity is excluded in the analysis of abatement costs.

7.2. Crops and agricultural areas

Impacts on crops and agricultural areas include crop loss, due to elevated concentrations of ozone, possible reduction of fertiliser usage, due to deposition of oxidised nitrogen, and the effects of acidification, due to deposition of oxidised nitrogen.

7.2.1. Impacts on crops

FB2001 have compiled dose-response relationships for different crops. Crop loss is assumed to depend linearly on AOT40 for the growing season with an adjustment due to the sensitivity of different crops. Slightly sensitive crops include rye, oats and rice, sensitive include wheat, barley, potato and sunflower and very sensitive include tobacco. The dose-response relationships have been implemented in the spreadsheet application.

As an alternative, updated dose-response relationships for the same crops have been taken from Holland et al. (2002). The updated relationships also use AOT40 for the growing season as the driver for the impacts. It should be noted that recent scientific findings indicate that crop loss due to ozone in some cases is better related to ozone uptake through stomata than AOT40. Ozone uptake modelling is, however, still at the forefront of research and in this study we have chosen to stay with AOT40 as the driver for the impacts.

7.2.2. Reduced fertiliser usage

Nitrogen is an essential plant nutrient applied by farmers to their crops. The deposition of oxidised nitrogen to agricultural soils is therefore beneficial if one assumes that the farmer has good control on the nutrient status of the farmland and does not fertilise excessively. Reduction in fertiliser usage was calculated by the following formula, based on FB2001:

$$\Delta F = A \cdot \Delta D_N$$

with ΔF = reduction in fertiliser nitrogen requirement in kg N / year

A = agricultural area in ha

ΔD_N = nitrogen deposition in kg N / ha / year

The areas of agriculture for the different European countries in the year 2000 were taken from the Statistical Office of the European Communities (EUROSTAT) and implemented in the spreadsheet application.

7.2.3. Acidification of agricultural soils

Liming of agricultural soils is necessary in order to balance acid inputs. The lime requirement was calculated by the following formula, based on FB2001:

$$\Delta L = 0.5 \cdot A \cdot \Delta D_A$$

with ΔL = additional lime requirement in kg / year

A = agricultural area in ha

ΔD_A = acid deposition in meq / ha / year

The agricultural areas for the different European countries were taken to be the same as in section 7.2.2. In principle only non-calcareous soils should be included. However, for simplicity we have assumed that liming is necessary for all soils.

7.3. Natural ecosystems

Impacts on natural ecosystems due to aviation emissions of NOx include eutrophication and acidification due to excessive deposition of oxidised nitrogen.

The assessment of the impact of nitrogen deposition in this study is based on information about critical loads. A critical load (CL), for deposition, has been defined as "a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge" (Nilsson and Grennfelt 1988). Thus, a CL is an indicator for sustainability of an ecosystem in that it provides a value for the maximum allowable deposition of a pollutant below which the risk of damage is reduced. The information on ecosystem sensitivity can be compared with deposition data to determine which areas receive deposition levels exceeding the CLs in that area.

Information about CLs are collected and summarized by the Co-ordinating Centre for Effects (CCE) in support of the Convention on Long-range Transboundary Air Pollution (CLRTAP) of the United Nations Economic Commission for Europe (UNECE). Since the late eighties, methods have been developed to compute and map CLs of acidity (sulphur and nitrogen based) and of eutrophication (nitrogen based). CCE issues annual reports were the progress in mapping CLs across Europe is presented.

In line with the methodology for the other impacts studied we have chosen to use nation average data on CLs when assessing the impact of oxidised nitrogen deposition to natural ecosystems. We acknowledge that this is a crude approach but, given the limited scope of the study, this simplification seems reasonable. As we will see in Section 8 the uncertainty in the valuation of effects on ecosystems is also large which also justifies a simple approach as a first step.

National data on CLs for eutrophication and acidification, due to nitrogen and sulphur deposition, and ecosystem areas were taken from recent CCE reports (Hettelingh et al., 2004; Posch et al., 2003, 1999). The assumed CL-percentiles and ecosystem areas applied for each country are given in the Appendix.

The information needed when estimating the marginal costs of ecosystem effects is the (aviation caused) change in the area where CLs are exceeded. It is very difficult to calculate this area accurately, for several reasons. The aircraft emissions over Sweden give relatively small contributions to acidifying and eutrophying deposition. This means that the *change* in area with exceedances will be small compared to typical regional scale model resolutions. In the MATCH model each grid point covers ca 1900 km^2 and it is very unlikely that the average deposition in a whole grid point will change from below the grid average critical load to above this value. In reality the CLs are highly variable, which means that, within a given model grid point, the deposition to some ecosystems may be close to their critical loads even if the deposition is far above (or below) the grid (or nation) average CL.

In order to get a rough estimate of the possible changes in areas of exceedances of CLs, two different sets of nation average percentiles of CL were investigated, one corresponding to protection of 95% of the ecosystems (the 5^{th} percentile of the CL) and one to protection of 50% of the ecosystems in the different countries. For grid points where the model total acidifying or eutrophying deposition is within 5% of these CLs, the Swedish air traffic NOx-emissions relative contributions to the areas of exceedance (of the chosen percentile) were calculated. The averages of the contributions at the 5^{th} and 50^{th} percentile of the CLs are used as estimates of the areas of exceedance. The method is described below, using LTO NOx-emissions and eutrophying deposition in Sweden as an example.

For Sweden, the used 5^{th} percentile for eutrophication CL was 125 eq/ha/year. In 13 model grid points (within Sweden) the *total* calculated eutrophying deposition is within

5% of this CL and the calculated average contribution to the deposition in these grid points from LTO NOx-emissions is 0.06 eq/ha.

Assuming that the eutrophying deposition, within the 13 grid points, is evenly distributed between the lower and upper end of the studied interval (118.75 – 131.25 eq/ha) leads to the conclusion that the 5^{th}-percentile is exceeded in ca 12000 ha extra[d] due to the Swedish LTO NOx-emissions.

12 (other) grid points within Sweden are exposed to nitrogen deposition within 5% the 50^{th} percentile (300 eq ha^{-1} $year^{-1}$). For these points the calculated LTO NOx contribution to deposition is 0.20 eq ha^{-1}, giving an estimated area of 15 700 ha of exceedance of the 50^{th} percentile of the CL for eutrophication.

To get an estimate of the *order of magnitude* of the increased area of exceedance of critical loads for eutrophication in Sweden the average of the two computed areas of exceedances (ca 14 000 ha) is used. However, the total ecosystem area in Sweden, which is sensitive to eutrophication, is 182 223 km^2 (Hettelingh et al., 2004) which is ca 40% of the total area of Sweden. To take this into account the area is multiplied by the factor 0.4 giving 5600 ha as an estimate of the area of exceedance in Sweden due to LTO NOx-emissions.

The same methodology, as for Sweden, was applied, for all countries covered by the model calculations, both for eutrophication and acidification of ecosystems. It must be recognized that this way of estimating the increased areas of exceedances is only a very rough estimate. To calculate accurate areas, much more detailed information about the distributions of critical loads and sensitive ecosystems are needed. National averages and national percentiles of the CLs are not enough for this.

8. Economic valuation

A consistent application of the IP approach requires that the cost for a certain impact can be directly related to the damage that is caused, so called damage cost. For most of the impacts included in this study damage costs are available although the costs used in each case can be discussed and further development is needed. For impacts on natural ecosystems however, no damage costs are available. Currently the second best solution of using so-called abatement costs is used in this case. Abatement costs are derived from the costs of reducing emissions to politically agreed targets e.g. the different protocols under the Convention for Long-Range Transboundary Air Pollution (CLRTAP) or the National Emission Ceiling Directive in EU. It should be recognized however, that abatement costs and damage costs may be different and that such differences could be used as an indication that political targets are either too ambitious or too low depending on the sign of the difference.

[d] 0.06 eq/ha × 13 grid points × 193 600 ha/gridpoint / (131.25 - 118.75) eq/ha ≈ 12 000 ha

8.1. Human health

Two different outcomes need to be valued concerning health effects. Those are the risk for reduced life expectancy due to air pollution (mortality) and the risk for different illnesses (morbidity).

8.1.1. Mortality

It is necessary to consider short-term and long-term effects on mortality separately since we have ER functions for both. Short-term effects are however, usually assumed to be included in the long-term effects so it is important not to apply both long-term and short-term effects in the calculation of the effects for the same pollutant in order to avoid double counting.

Valuation of mortality is usually based on studies estimating the value of a statistical life (VSL). VSL is defined as the average willingness to pay (WTP) for a certain risk reduction divided by the risk reduction:

$VSL=WTP/\Delta risk$.

VSL has been estimated in the case of deaths in traffic accidents. The average reduction in life expectancy in this case is around 40 years. However, there are not many studies available that have studied the type of risks that are related to exposure to air pollutants, i.e., reduction in life expectancy by a few years.

The initial approach in ExternE has been to start from estimates of VSL from other fields and recalculate VSL to the value of a life year lost (VOLY) (Friedrich and Bickel, 2001). The VSL value used was 3.36 MEUR. Two different values of a life year lost are used. One for acute mortality (VOLYa) used for short-term effects and one for chronic mortality (VOLYc) used for long-term effects.

In FB2001 VOLYc is estimated by discounting VOLYa with an interest rate of 3% over 20 years. This means that VOLYc is 58% of VOLYa. For a zero interest rate VOLYc and VOLYa have the same value. This way of estimating VOLY from VSL has been criticized because it builds on assumptions about the individual willingness to pay that are not considered valid, but also because the risks regarding air pollution are different (Rowlatt et al., 1998).

New studies have been initiated to improve the information, e.g. NewExt (Taylor, 2004) that is a continuation of ExternE and a British DEFRA (Department for Environment, Food and Rural Affairs) supported study (Chilton et al., 2004). Based on these studies updated valuation of mortality has been suggested by Nerhagen et al. (2005). Table 2 is taken from their study and show values for VOLYa and VOLYc under different assumptions.

For comparison with earlier studies we have used the same values as in the Swedish ExternE study (BI2002). In the updated, or best estimate, calculation we have used 0.67 MSEK for VOLYa and 0.45 MSEK for VOLYc (Nerhagen et al., 2005). These values can easily be changed in the spreadsheet application. To some extent differences in the values can be dependant on differences in income. An adjustment of values for individual countries can be based on consumer price index (UNITE, 2000). This possibility has also been included in the spreadsheet application.

Table 2. Monetary values for mortality (MSEK 1998). From Nerhagen et al. (2005)

Alternative discount rate	VSL	$VOLY_{acute}$	$VOLY_{cronic}$ (20 yr)	$VOLY_{cronic}$ (10 yr)
Swedish ExternE (3%)	27	1.18	0.69	-
ASEK* (4%)	12.9	0.67	0.31	0.45
ASEK* (2%)	12.9	0.49	0.33	0.40

*ASEK = Arbetsgruppen för Samhällsekonomiska kalkyler, see Nerhagen et al. 2005 and references therein.

8.1.2. Morbidity

In ExternE values for different types of morbidity outcomes have been used. These are taken from the UNITE (UNIfication of accounts and marginal costs for Transport Efficiency) project (UNITE, 2000) and are listed in Table 3. For comparison more recent results for some outcomes from a study by Ready et al. (2004, published in Eftec, 2004) are also shown. The study included several countries; included in Table 3 are results for Norway.

Table 3. Monetary values for morbidity (EUR)

Effect	UNITE for Sweden	Ready et al.* for Norway
Chronic bronchitis	168 840	-
Respiratory Hospital admission	4400	482
Congestive heart failure	3360	-
Chronic cough in children	240	-
Restricted activity day	120	190
Asthma attack	85	382
Cough	42	58
Minor restricted activity day	42	-
Symptom day	42	50
Bronchodilator usage	40	-
Lower respiratory symptom	8	-

*Published in Eftec, 2004.

The most important health consequence, in terms of valuation, is chronic bronchitis. This value is however questioned since the present valuation is not compatible with the symptoms that the ER relationships are representing (Taylor, 2004). For the remaining effects, except for hospital admission, the values used in ExternE are lower than the corresponding values from Ready et al. The difference is quite large for asthma attacks but this could be due to the fact that the results from Ready et al. are based on asthma attacks leading to a hospital visit while this may not be the case in the studies used in

UNITE. The largest difference is found for hospital admission. The reason for this is unclear but it can be noted that the DEFRA study gave higher values in the range 2096-11376 EUR. In the estimates presented in this study we have chosen to use the ExternE values to give an indication of the possible costs due to morbidity.

8.2. Crop loss, fertilisation and liming

Valuation of crop losses was made based on national producer prices for the year 2000 for the different crops considered. Data were taken from EUROSTAT.

The costs for fertilisation and liming were taken from ExternE (Friedrich and Bickel, 2001) using a price of 0.53 EUR per kg of fertiliser and 0.018 EUR per kg of lime.

8.3. Acidification and eutrophication of natural ecosystems

It is difficult to evaluate ecosystem effects by the IP approach. Therefore abatement cost calculations have been introduced within ExternE.

Vermoote and De Nocker (2003) have developed a so-called ”Standard Price” approach to be compatible with the ExternE methodology. Based on official emission reduction targets they estimate a cost of 100 €/ha (or 10 000 €/km^2) for acidification and eutrophication of ecosystems. In ExternE (BI2002) much higher abatement cost estimates have been used, 176 000 $€_{1998}/km^2$ for acidification and 25 900 $€_{1998}/km^2$ for eutrophication. The reason for the lower estimate of Vermoote and De Nocker is that they have tried to single out the ecosystem part of the total abatement cost to get a cost that is *additive* to the ExternE estimates for impacts on human health, agriculture and building materials.

We use the value 100 €/ha as a starting guess for the “Best estimate” in the spreadsheet application. The user can easily change this to other values. For comparison with other (ExternE) studies the values 176 000 €/km^2 and 25 900 €/km^2 (BI2002) were used for acidification for eutrophication, respectively.

9. Emission costs

The underlying information and mathematical relations discussed in Sections 5-8 above have been implemented into a spreadsheet application in Microsoft Excel. The spreadsheet calculates emission costs for aviation NOx emissions in Swedish air space. The costs are calculated on a nation-by-nation basis using nationally averaged or aggregated information. Aggregation to the European level is also performed.

Calculations have been made for two cases: a) ER relationships and valuation following ExternE and FB2001 and b) Updated, or Best estimate, ER relationships and valuation. This was done in order to be able to make direct comparisons with earlier studies using the ExternE methodology for the transport sector in Sweden.

The marginal costs for the air traffic NOx-emissions were calculated by dividing the total computed costs by the total emissions. This was done both for the LTO-emission and the non-LTO emission scenario.

Table 4 summarise the results on the European level. Table 5 shows a comparison between the results for LTO emissions from the present study and the results for LTO emissions from BI2002 and OT2003.

The Best estimates, of this study, of various costs in different countries from LTO and non-LTO NOx-emissions in Sweden are illustrated in Figures 7 and 8. For LTO-emissions the costs in Sweden are dominating. The non-LTO emissions influence a much larger area and the greatest costs are calculated for Russia.

Table 4. Summary of Swedish aviation NOx emission costs in Europe as estimated in the present study. Two different sets of estimates are given. The first is denoted ExternE and is based on ER-functions from FB2001 and monetary valuations from FB2001, BI2002, and UNITE (2000). The second is denoted Best estimate and uses updated ER-functions for mortality from Nerhagen et al. (2005) and for crop losses from Holland et al. (2002) and alternative valuations for mortality from Nerhagen et al. (2005) and for acidification/eutrophication of ecosystems from Vermoote and DeNocker (2003).

	Total cost (M SEK)				**Marginal cost/kg NOx (SEK)**			
	ExternE		**Best estimate**		**ExternE**		**Best estimate**	
Effect	**LTO**	**non-LTO**	**LTO**	**non-LTO**	**LTO**	**non-LTO**	**LTO**	**non-LTO**
Mortality - PM	1.3	4.6	1.7	5.8	1.5	0.94	1.8	1.2
Morbidity - PM	0.67	2.3	0-0.67	0-2.3	0.74	0.47	0-0.74	0-0.47
Mortality - ozone	1.4	1,3	0.31	1.7	1.6	0.27	0.34	0.36
Morbidity - ozone	2.4	13	0-2.4	0-13	2.6	2.7	0-2.6	0-2.7
Sum health effects:	**5.8**	**21**	**2.0-5.0**	**7.5-23**	**6.4**	**4.4**	**2.2-5.5**	**1.5-4.7**
Crop loss – ozone*	0.70	5.2	0.23	2.0	0.77	1.1	0.25	0.41
Fertilisation of soils	-0.05	-0.09	-0.05	-0.09	-0.05	-0.02	-0.05	-0.02
Acidification of soils	0.01	0.01	0.01	0.01	0.01	0.00	0.01	0
Sum effects agriculture	**0.66**	**5.2**	**0.19**	**1.9**	**0.72**	**1.1**	**0.21**	**0.39**
Ecosystem eutrophication	21	33	8.2	13	23	6.8	9.0	2.6
Ecosystem acidification	37	48	2.1	2.7	41	9.8	2.3	0.56
Sum of costs	**65**	**107**	**13-16**	**25-40**	**72**	**22**	**14-17**	**5.1-8.3**

*Excluding crop loss costs in Belarus, Moldova, Russia and Ukraine, due to lack of data on agricultural production of different ozone sensitive crops.

Table 5. Comparison of emission costs of LTO emissions from different studies for Sweden. It should be noted here that OT2003 only apply the IP approach directly for local effects due to PM and ozone. The effects due to NOx emissions are only considered on the regional scale and then using average damage cost values from UNITE (2000).

Effect	Marginal cost/kg NOx (SEK)			
	Bickel et al. (2002)[*,†]	Otterström et al. (2003)	This study ExternE	This study Best estimate
Mortality - PM	7.8[†]	8.21	1.5	1.8
Morbidity - PM	3.5[†]	3.57	0.74	0-0.74
Mortality - ozone	1.7[*]	0.73	1.6	0.34
Morbidity - ozone	3.5[*]	1.08	2.6	0-2.6
Sum health effects:	**16.5**	**13.58**	**6.4**	**2.2-5.5**
Crop loss - ozone	2.6[*]	1.20	0.77[‡]	0.25[‡]
Fertilisation of soils	-0.09	-	-0.05	-0.05
Acidification of soils	0.0	-	0.01	0.01
Sum effects agriculture:	**2.5**	**1.20**	**0.72**	**0.21**
Ecosystem eutrophication	51	-	23	9.0
Ecosystem acidification	0.0	-	41	2.3
Sum of costs	**70**	**14.79**	**72**	**14-17**

[*]The costs due to ozone effects have been calculated per kg emitted NOx. In Bickel et al. (2002) VOC emissions were also varied but these emissions are relatively small compared to the NOx emissions and for this comparison they have been neglected. [†]For this comparison the PM effects are those related to secondary nitrate formation. [‡] Excluding crop loss costs in Belarus, Moldova, Russia and Ukraine, due to lack of data on agricultural production of different ozone sensitive crops.

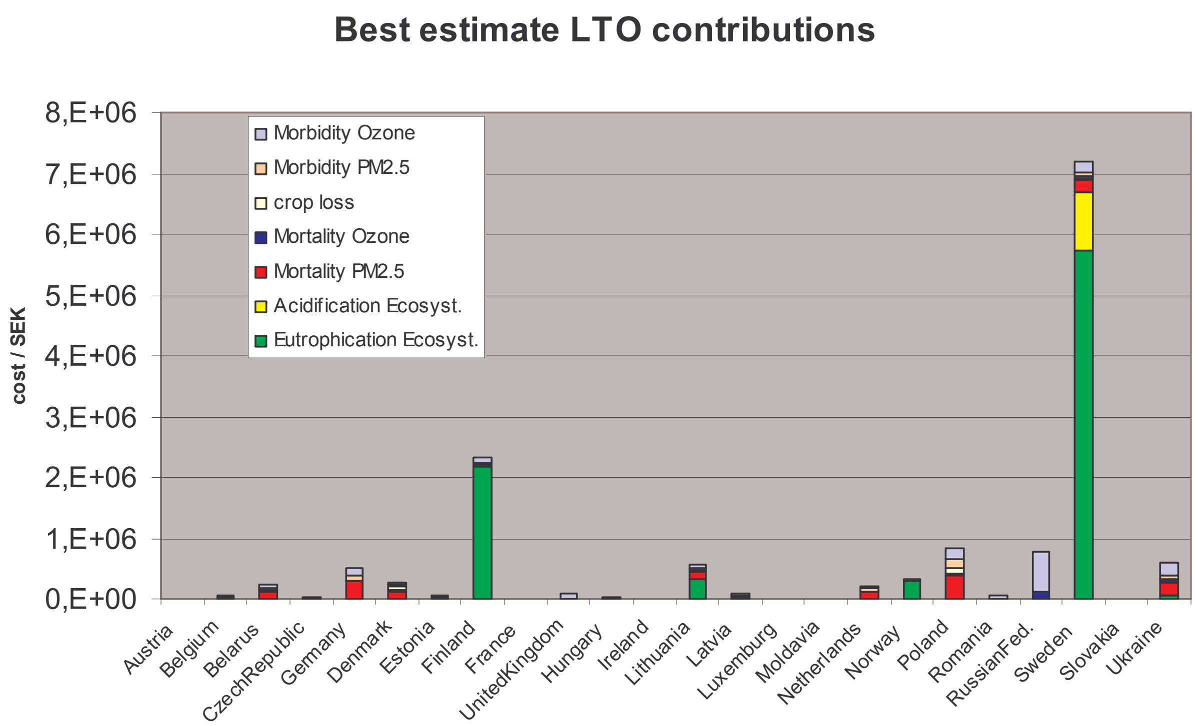

Figure 7. Best estimate of costs for different effects in Europe of Swedish LTO-emissions of NOx.

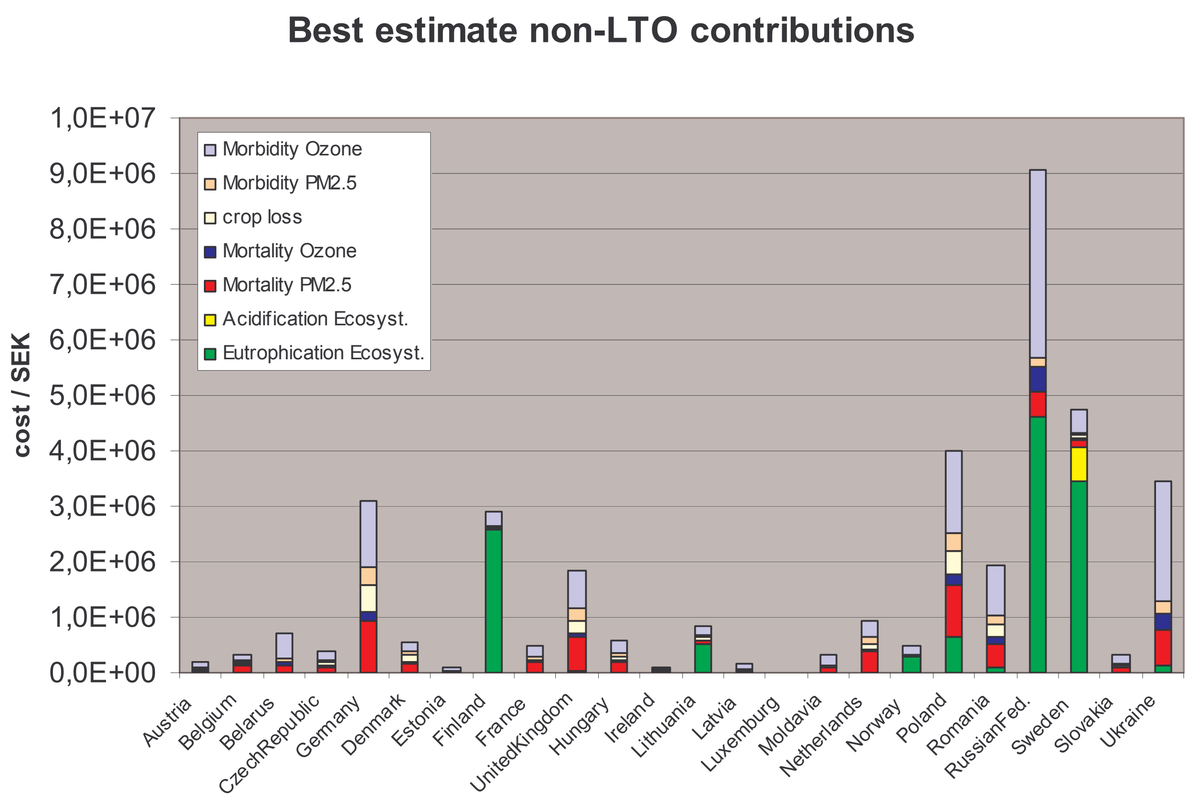

Figure 8. Best estimate of costs for different effects in Europe of non-LTO-emissions of NOx over Sweden.

9.1. Health effects

The costs for health effects due to PM for the ExternE case are smaller than the costs due to ozone in the present study. The difference is largest for the LTO case where ozone gives a two times higher cost than PM. For the non-LTO case the difference is somewhat smaller. For the Best estimate case the differences between ozone and PM are smaller since a higher risk factor for mortality is used for PM and a lower for ozone.

The estimates for ozone are comparable to BI2002 and OT2003 when using the same ER relations and valuation. The Best estimate gives a smaller cost for mortality due to ozone primarily because the risk factor used is a factor of two lower.

For PM the present study gives substantially lower cost than BI2002 and OT2003 even when using the same ER relations and valuation. The major part of this difference must be due to differences in the chemical transport modelling between the EcoSense model used in ExternE and MATCH. The long-range transport model used in EcoSense is quite different compared to MATCH. In order to understand and analyse this difference, detailed results from the EcosSense model would be needed.

Model intercomparison studies that the MATCH model has participated in, e.g. EURODELTA (http://rea.ei.jrc.it/netshare/thunis/eurodelta/), indicate substantial differences between different models for calculations of PM while model results for ozone are more homogeneous.

Another factor of importance is the use of nation average impacts, when calculating the emission costs, in the present study. That is, variations in population density, within countries, have not been taken into account. This is mainly of importance for the valuation of costs in countries close to the emissions studied, i.e., Sweden, Finland and Denmark. However, rough calculations indicate that this factor alone cannot explain the difference in valuation of PM.

The largest health effect costs occur in Russia, Poland and Ukraine both for the LTO emissions and for the non-LTO emissions.

9.2. Effects on crops and agricultural areas

The estimated costs for effects on crops and agricultural areas are dominated by the costs for crop loss. The costs and gains related to acidification and fertilisation of agricultural soils are estimated to be small.

The Best estimate results in a cost that is more than a factor of two lower than the ExternE estimate. This is due to the differences in the ER relations between FB2001 and Holland et al. (2002).

The greatest costs for crop losses are calculated for Poland, Denmark and Sweden for the LTO emissions and for Germany, United Kingdom and Poland for the non-LTO emissions. As can be expected the non-LTO costs include countries further away. It should be noted that, due to lack of data on agricultural production of ozone sensitive crops, no crop losses were calculated for Russia, Ukraine, Moldova and Belarus. As can be seen in Figure 5 relatively large effects on crops can be expected in all of these countries except Russia. This means that the true costs for crop losses are somewhat higher than those presented in this report. If data on agricultural production in the four missing countries become available this can easily be included in the spreadsheet application.

The results can be compared to those from the Swedish ExternE study (BI2002) and the study for Västerås airport (OT2003). BI2002 also give a low gain for fertilisation and the cost due to crop loss dominates. The cost for crop loss in BI2002 is however more than three times higher than the ExternE estimate in this study. The reason for this is unclear but possible candidates include the nation averaging used in this study, lack of crop loss data for several major agricultural nations in our case, differences in the dispersion modelling and differences due to the choice of meteorological conditions (The year for which the modelling is performed). The estimate in OT2003 is lower than BI2002 but still almost a factor of two higher than the ExternE estimate in this study.

If we compare with the Best estimate from this study the differences are larger due to different ER relations. In this case the present study gives costs that are a factor of ten lower than BI2002 and a factor of five lower than OT2003.

9.3. Effects on natural ecosystems

The costs for effects on natural ecosystems given in Table 4 must be considered as very uncertain estimates. This is unfortunate since the calculated ecosystem costs are very high compared to the other impacts. This is true even for the “Best estimate” valuation based on the 100 €/ha cost from Vermoote and De Nocker (2003).

The difference between the ExternE estimate and the best estimate is the cost per area of protected ecosystems. The ExternE estimate (Friedrich and Bickel, 2001) is based on abatement costs derived from the RAINS (Regional Air Pollution INformation and Simulation) model. These are different for acidification and eutrophication. The “best estimate” is based on the value 100 €/ha (both for acidification and eutrophication) given by Vermoote and De Nocker (2003). The ExternE-values are much higher, especially for acidification, which results in higher costs.

The largest estimated ecosystem costs occur in Sweden and Finland for the LTO emissions and in the Russian Federation, Sweden and Finland for the non-LTO emissions.

The results can be compared to those from the Swedish ExternE study (BI2002). BI2002 give a zero value for acidification possibly due to treshold effects in the methodology. The cost for eutrophication in BI2002 is a factor of two higher compared to the ExternE estimate in this study. The total ecosystem effect is however not that different. When using the new valuation the difference becomes larger and exceeds a factor of four.

10. Conclusions

A first assessment of the environmental costs due to non-LTO NOx emissions from aviation in Swedish air space has been presented. The following conclusions are drawn.

- Marginal costs for NOx emissions from aviation are estimated to be higher for LTO emissions than for non-LTO emissions. If ecosystem effects are excluded the marginal costs become almost the same for LTO and non-LTO emissions of NOx.

- The costs due to acidification and eutrophication of natural ecosystems dominate. This is similar to what have been found in earlier studies. There are however substantial uncertainties connected to the valuation of ecosystem effects. Health effects due to PM and ozone were estimated to be next in importance. Costs due to crop loss and effects on agricultural soils were estimated to be smaller.

- Comparison with earlier studies for LTO emissions in Sweden show that the results are comparable for effects on ecosystems and agriculture, as well as for health effects due to ozone, provided that the same exposure response relations and valuation are applied. The estimated costs due to health effects of PM are however substantially lower than earlier estimates. This is most likely due to a combination of differences in the chemical transport models used for calculating the exposure and the simplification of using nation averages when estimating costs in the present study.

Several uncertainties have been identified above and during the workshop. The following are considered most important:

- The valuation of ecosystem effects includes a number of uncertainties. The mapping of ecosystem sensitivity varies to some extent between different countries and the valuation is based on abatement costs. The method for calculating the change in unprotected ecosystem area in the present study is based on country average critical loads. This also introduces some uncertainty. A more detailed approach would be to make calculations on a grid square by grid square basis, using grid square specific critical loads. This is possible but was outside the scope of the present study.

- We have not used updated ER relations for effects on morbidity in this study. There are new European studies for hospital admissions, but for chronic bronchitis updated risk factors are lacking, and the calculations build on one old study from the US. An additional problem is the valuation of morbidity effects. The valuation can be expected to vary between countries but such information is lacking for many health outcomes. We have used the ExternE ER relations and valuation to indicate the possible magnitude of the morbidity costs but the uncertainties here are substantial.

- PM formed from aviation NOx emissions is mostly nitrate in this study. Is nitrate really dangerous? Studies on animals do not indicate that ammonium nitrate in itself is toxic in relevant concentrations. However, when the short-term effect of PM in the US was compared between regions, California with the highest nitrate proportion had an ER-coefficient above the average. A few studies of short-term effects on mortality have also shown that nitrate particulates seem important, but maybe not due to nitrate in itself.

- The sensitivity of different chemical transport models to changes in emissions is an important area of uncertainty. This appears to be the most important factor explaining the differences in valuation of health effects due to PM in this study. Harmonisation of chemical transport models is certainly an important topic here. Part of the explanation could also be related to the choice of time period for the modelling. Meteorological conditions are known to be important and simulations for different years are expected to give different results.

- During the workshop a question was raised regarding the representativity of ozone concentrations calculated on the regional scale for assessment of health effects. Most people live in urban areas and ozone concentrations here are lower due to titration with fresh NO emissions mainly from road traffic. Suggested reductions could be in the range 10-15% but the reduction can be expected to vary with population and traffic density. There are no agreed on relationships that can be used to compensate for this effect. The best approach is to increase the geographical resolution in urban area but this then also implies using high-resolution emission data.

- The model domain chosen for the exposure calculations was a compromise between computational cost and coverage and does not cover all areas where effects can be expected. For health effects and effects on crops this results in an underestimate of the effects due to non-LTO emissions. For LTO emissions the extension of the model domain is a minor problem.

Future work to improve the present assessment of environmental costs due to aviation emissions of NOx are needed in all links of the Impact Pathway chain. A number of uncertainties have been identified above. We suggest that the following topics (in order of importance) should be considered in future studies:

- Use of improved ER-relations and valuation of morbidity due to both PM and ozone. For PM it is important also to differentiate between different sources, sizes and composition. Some important effects are not well studied, such as onset of chronic bronchitis.

- Grid based assessment of all the effects studied instead of nation averaging.

- Calculations for more than one year in order to reduce the impact of meteorological variability.

This study has considered only the effects due to NOx emissions from aviation in Swedish air space since this was the objective. Many other substances are emitted in aircraft exhaust, e.g. particles, hydrocarbons, CO and SO_2, and may also have significant environmental effects. It would be straightforward to analyse the environmental impact of such emissions using the same modelling system as used here.

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12. Appendix

Table A1. Assumed critical loads for eutrophying nitrogen deposition ($CL_{nut}(N)$)for different protection levels and different countries and ecosystem areas sensitive to eutrophication [based on Hettelingh et al., 2004; Posch et al., 2003, 1999]

Country	$CL_{nut}(N)$ 5th percentile (eq ha^{-1} a^{-1})	$CL_{nut}(N)$ 50th percentile (eq ha^{-1} a^{-1})	Ecosystem area sensitive to eutrophication (km^2)
Austria	250	400	37 572
Belgium	500	700	7 282
Belarus	125	750	103 364
CzechRepublic	500	400	18 272
Germany	125	450	105 747
Denmark	550	800	3 149
Estonia	250	400	22 411
Finland	125	250	240 403
France	300	500	180 101
UnitedKingdom	625	1050	74 203
Hungary	450	800	10 460
Ireland	700	700	8 936
Lithuania	250	400	(18 500 est.)
Latvia	250	400	(32 000 est.)
Luxemburg	-	-	(700 est.)
Moldavia	-	-	11 985
Netherlands	300	1250	4 624
Norway	500	550	226 631
Poland	350	450	88 383
Romania	250	400	(54 000 est.)
RussianFed.	125	250	3 517 136
Sweden	125	300	182 223
Slovakia	250	400	19 227
Ukraine	550	400	(145 000 est.)

Table A2. Assumed critical loads for acidifying sulphur deposition ($CL_{max}(S)$)for different protection levels and different countries and ecosystem areas sensitive to acidification [based on Hettelingh et al., 2004; Posch et al., 2003, 1999]

Country	**$CL_{max}(S)$ 5th percentile (eq ha^{-1} a^{-1})**	**$CL_{max}(S)$ 50th percentile (eq ha^{-1} a^{-1})**	**Ecosystem area sensitive to acidification (km^2)**
Austria	1000	1900	37 572
Belgium	1200	2450	7 282
Belarus	150	950	103 364
CzechRepublic	400	750	18 272
Germany	600	1300	105 747
Denmark	950	1950	3 149
Estonia	900	>	21 450
Finland	475	950	266 805
France	400	3800	180 101
UnitedKingdom	350	550	77 672
Hungary	2700	>	10 460
Ireland	400	1300	8 936
Lithuania	1000	>	(0 est.)
Latvia	1000	>	(0 est.)
Luxemburg	-	-	(0 est.)
Moldavia	-	-	11 985
Netherlands	650	1800	7 583
Norway	150	550	453 088
Poland	600	1050	88 383
Romania	900	>	(0 est.)
RussianFed.	300	950	3 517 136
Sweden	50	550	379 261
Slovakia	425	2500	19 227
Ukraine	850	>	(0 est.)

Table A3. Assumed critical loads for acidifying nitrogen deposition ($CL_{min}(N)$ and $CL_{max}(N)$) for different protection levels and different countries [based on Hettelingh et al., 2004; Posch et al., 2003, 1999]

Country	**$CL_{min}(N)$ 5th percentile (eq ha^{-1} a^{-1})**	**$CL_{max}(N)$ 5th percentile (eq ha^{-1} a^{-1})**	**$CL_{min}(N)$ 50th percentile (eq ha^{-1} a^{-1})**	**$CL_{max}(N)$ 50th percentile (eq ha^{-1} a^{-1})**
Austria	150	1300	300	2500
Belgium	300	2000	400	3000
Belarus	100	600	700	1750
Czech Republic	400	875	600	1500
Germany	100	1000	400	1900
Denmark	300	1650	600	3000
Estonia	425	>	450	>
Finland	75	750	100	1400
France	250	950	450	4500
United Kingdom	200	750	550	1400
Hungary	400	>	775	>
Ireland	300	950	550	1900
Lithuania	275	>	300	>
Latvia	290	>	300	>
Luxemburg	-	-	-	-
Moldavia	-	-	-	-
Netherlands	75	800	450	2500
Norway	35	300	45	875
Poland	275	1650	400	2500
Romania	500	>	550	>
Russian Fed.	50	400	550	1500
Sweden	85	225	200	1100
Slovakia	250	900	500	3500
Ukraine	550	>	400	>