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Swedish Civil Aviation Administration and
Swedish Institute for Transport and Communications Analysis

ESTIMATION OF ENVIRONMENTAL COSTS OF AIRCRAFT LTO EMISSIONS

- Pilot Study -

ELECTROWATT-EKONO
Jaakko Pöyry Group

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Electrowatt-Ekono Oy

Summary

Information on the environmental costs of transport emissions is necessary for making socio-economic investment analysis and setting charges on the use of transport networks. Marginal environmental emission costs of aviation have been estimated in this pilot study mainly by applying the ExternE methodology. The marginal emission costs caused by one aircraft type are assessed at Västerås airport.

The ExternE methodology applied consists of four phases: emission inventory, dispersion modelling, impact assessment and monetary valuation of the impacts. The emission inventory consists of the emissions caused by the case aircraft. Dispersion modelling calculations are performed for an LTO cycle at Västerås airport.

The impacts of emissions and pollutant concentrations are assessed on local, regional and global scales by receptors, i.e. people, natural resources and property exposed to pollution concentrations. The impacts include mortality and morbidity, impacts on building materials and crops, and global warming. The valuation of impacts is based on research results mainly from European studies i.e. ExternE and UNITE. The recently published BeTa database of the European Commission is also used in this assessment.

The emission costs are presented separately for local, regional and global impacts, as well as by pollutant component.

The results indicate that impacts related to global warming form the main cost category covering approximately 79 - 84% of the total costs when the assessment is based on a shadow value of EUR 65/t CO_2. The variation depends on whether UNITE or BeTa input data for regional impacts is used. The cost of regional impacts contributes 11 – 21% to the total costs and local impacts by approximately 5%. The total costs are EUR 130 - 137 i.e. SEK 1 190 – 1 260 per LTO cycle.

Uncertainties of the results of this study concern primarily the credibility of the exposure-response functions and their application in the case areas, the data on particulate emissions, and the appropriateness of the unit costs used for assessing the costs of health, crop, and material impacts in Sweden. Furthermore, there is no consensus on which unit value should be used for valuing the impacts of global warming.

However, the methodology and unit costs applied in this study represent the state-of-the-art knowledge in the field of applied environmental economics.

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PREFACE

In September 2002, the Swedish Civil Aviation Administration commissioned Electrowatt-Ekono Oy to conduct a pilot study, the purpose of which was to estimate the marginal environmental emission costs caused by aircraft emissions using the Impact Pathway Method, also known as the ExternE methodology. The study was jointly initiated and financially supported by the Swedish Institute for Transport and Communications Analysis (SIKA) and the Swedish Civil Aviation Administration.

The study was co-ordinated by senior adviser Lennart Bergbom from the Swedish Civil Aviation Administration. The contact person at SIKA was Economist Per-Ove Hesselborn. Also Fredrik Jaresved at the Swedish Association of Local Authorities followed the project closely.

The project team of Electrowatt-Ekono Oy consisted of Project Manager, Senior Adviser Tomas Otterström (since March 24th 2003 Executive Vice President at GreenStream Network Ltd), Senior Consultant Kari Hämekoski and Project Assistant Peter Anton. Risto Varjoranta and Harri Pietarila from the Finnish Meteorological Institute (FMI) provided input data for the project.

Input data was also collected by Reidar Grundström at the Civil Aviation Administration and Anette Näs and Anders Hasseltot from the Swedish Defence Research Agency, FOI. SIKA provided geographically distributed population data for Sweden.

1 INTRODUCTION

Information on the environmental costs of transport emissions is necessary for making socio-economic investment analysis and setting charges on the use of transport networks.

When analysing investments, it is important to know the total socio-economic costs of transport emissions during the assessment period of the investments. This purpose is served by more general information on average emission costs expressed e.g. in euro per tonne in the main categories of emissions.

In the case of charging, it is important to know the so-called marginal emission cost, for example caused by an LTO cycle. Such a cost can be expressed, for example, as SEK and EUR per LTO cycle per impact category or per tonne of relevant pollutants. This would allow charging air transportation according to the full social cost.

This pilot study assesses the marginal emission costs caused by a repeated LTO cycle once every hour in a year of a specific aircraft at Västerås airport.

The marginal emission costs are mainly assessed with the Impact Pathway Method (also known as the ExternE method) as spreadsheet calculations. This methodology was developed for assessing the damage costs of energy and transport related atmospheric emissions in a group of consecutive European research projects in the 1990s, the latest being ExternE Core Transport (European Commission, 1997 and 1999c; Friedrich & Bickel, 2001). In addition, results from the UNITE project have been utilized.

Recent information, the so-called BeTa (Benefits Table) database for externalities of air pollution in Europe of DG Environment has also been used at this study. There are some differences both in valuation data (i.e. in monetary values used in the valuation of impacts) and in exposure-response functions between the ExternE and BeTa data sets.

BeTa gives the marginal external costs of air pollution (SO_2, NOx, VOCs and PM) in terms of euros per ton of pollutant for each EU Member State. Furthermore, these external costs are divided between rural and urban areas. The database is largely based on existing data generated by the ExternE project and takes into account the approach to valuation of mortality recommended by economics experts at a workshop convened by DG Environment.

The methodology applied in this pilot study, along with the marginal cost information produced, provides a basis for defining the environmental cost component of an LTO cycle on the local, regional and global scale.

This is in line i.e. with the principles of marginal cost based charging of the use of transport networks, which is currently debated by the European Commission (European Commission, 1999a and 1999b).

The Impact Pathway Method puts a monetary weight on emission detriments, based on assessing the value of damages. This approach is also known as the damage cost approach. The damages include climate change, health impacts, crop impacts and material impacts caused by various pollutants.

However, due to the fact that the damages of climate change have been found difficult to assess, an abatement cost approach based on a combination of CO_2 tax levels and the market price for CO_2-allowances in Europe is used here for providing a value for CO_2 emissions. Alternative value settings for CO_2 emissions are, however, discussed within the report.

The report structure is as follows. Chapter 2 describes the methodological background and its application in this pilot study. Also, the basis for valuing physical damages caused by emissions is discussed according to the features of the case study.

Chapter 3 presents the results of the dispersion modelling estimations, i.e. pollutant concentrations caused by an LTO cycle of a specific aircraft repeated once every hour in a year. In Chapter 4, the results of the environmental cost assessment are presented. Chapter 5 analyses critically the results and the relevance of the results for policy-making. In Chapter 6, the conclusions of the report are presented.

2 METHODOLOGY

2.1 The principles of the Impact Pathway Method

The ExternE family of research projects has developed the Impact Pathway 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. European Commission (1997 and 1999c) and Friedrich & Bickel (2001).

An impact pathway is the sequence of events linking a burden to an impact and subsequent valuation. The Impact Pathway Method is a four-phased multidisciplinary tool for assessing the damages caused by various 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.

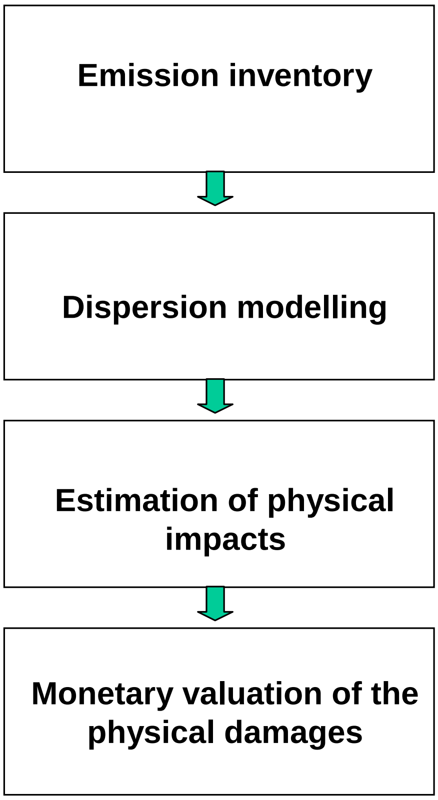

Figure 1. A simplified description of the Impact Pathway Method.

The underlying principles of the Impact Pathway Method are:

- *Transparency*: To present precisely how the results are calculated, the uncertainty associated with the results and the extent to which the external costs have been quantified.

- *Consistency of methodology*: Clearly established models and assumptions (e.g. system boundaries, exposure response functions and valuation of risks to life) are used.

- *Comprehensiveness of analysis*: All of the effects that may raise significant externalities should be identified and analysed.

2.2 Estimation approach in this study

Six pollutants are considered in calculations: nitrogen oxides (NO_x and NO_2), carbon monoxide (CO), hydrocarbons (HC), particulate matter (PM), sulphur dioxide (SO_2). In addition, ozone (O_3) depletion due to the chemical transformation of nitrogen oxides was computed. All PM is assumed to be $PM_{2.5}$ in this study.

The priority impact pathways of airborne pollutants and greenhouse gases addressed in this study are presented in Table 2-2 in Chapter 2.3.3. For a full list of identified impact pathways, see Friedrich & Bickel (2001).

2.2.1 Local impacts

For assessing *local impacts*, all four phases of the Impact Pathway Method are conducted, i.e. emission data for the case aircraft is obtained, dispersion modelling is performed (see Chapter 2.3) and detrimental impacts of pollutant concentrations are assessed according to case specific receptor data. The resulting damages are valued by unit values by damage category presented in the latest literature and research. Only health effects are considered in this phase. Based on the results of other studies (e.g. Friedrich & Bickel, 2001 and Gynther *et al.*, 2000) concerning external costs, the significance of other impacts (materials) is typically minor on a local scale compared with health impacts.

2.2.2 Regional impacts

For assessing *regional impacts*, not all phases of the impact pathway are conducted. Emission data from the case aircraft is obtained, but dispersion modelling not is performed, nor is case-specific receptor data obtained. In turn, regional damages are valued according to the damage values of pollutants per tonne applicable in the study area presented in the latest literature and research (UNITE, 2000) as well in BeTa database[1].

2.2.3 Global impacts

Global impacts (global warming) are assessed and discussed separately as the impacts are different in comparison with 'classical' air pollutants. There are also major uncertainties related to climate change impacts. Emission data for the case aircraft is used. Dispersion modelling is not performed because it is of no relevance in the case of CO_2. The method of calculating costs of CO_2 emissions basically consists of multiplying the amount of CO_2 emitted by a cost factor.

In addition to CO_2 emissions, aviation has other notable effects on global warming via O_3, CH_4, contrails, cirrus clouds, sulphate aerosols and soot (see Penner *et al.* 1999). These impacts are not discussed in this context, since they are relevant for the en route phase rather than the LTO-cycle analysed in this study.

2.3 Application of Impact Pathway Method in this study

2.3.1 Emissions

In this study the dispersion and dilution of emissions caused by the repeated LTO cycle once every hour during the year of one aircraft type (B737-800) at the Västerås airport are considered. The dispersion of emissions during a straight approach to the airport including retardation and all taxing phases to the station and takeoff from the airport including

[1] http://europa.eu.int/comm/environment/enveco/studies2.htm

all preceding taxing phases to the station were considered. The landing and take off were in direction 190°, off runway at taxiway B.

The aircraft studied was of type B737-800 with a CFM56-7B24 engine. Emission data supplied by the client (FOI, 2002) formed the basis for the emission modelling and dispersion calculations in this study (Table 2-1). One LTO cycle every hour of the study period (one year) was assumed. The calculations were based on hourly averages of emissions.

The PM data used in this study are based on information in the LAX Master Plan, 1999, which uses results from a number of underlying scientific studies. The LAX Master Plan reports measured PM data for a few jet engines. The engine that most resembles the engines of the aircraft used by Ryanair is the CFM56-5C2, which belongs to the same engine ICAO certification data for jet engine emissions do not include what is usually meant by particulate matter (PM). The engine certification process does involve measurement of the so-called smoke number (SN), which reveals the exhaust content of visible soot. Soot consists of particles but the size and mass distributions of soot are, however, not comparable with the particulate indices PM_{10} and $PM_{2.5}$, which refer to particles with a diameter of less than 10 µm and 2.5 µm, respectively. The alternative, when there are no certified PM data available, is to look for non-certified data of scientific quality. Unfortunately, the number of such alternative data sources is also very limited. For some engines and for some (non-standardised) thrust settings, PM emissions have been measured.

family as the engine actually used at Västerås (CFM56-7B24). It is, however, important to note that there may be substantial differences in PM emission characteristics between the two engines. Furthermore, the measured emissions for this engine only refer to one thrust setting (or rather one level of fuel flow), which means that one has to rely on correlations between smoke number and PM emissions established in the LAX Master Plan in order to simulate particulate emissions for the different thrust settings used during the LTO-cycle. To conclude, this means that there are significant uncertainties involved in the measurement and in the empirical modelling of PM emissions. Hence, also the effects of PM emissions shown in this study are afflicted with notable uncertainties.

Table 2-1. Emissions from one takeoff, one approach, one LTO cycle and 8760 LTO cycles annually.

	CO_2	CO	HC	NO_X	SO_2	Particles
Takeoff, kg	1 058	1.1	0.11	6.1	0.33	0.46
Approach, kg	613	1.4	0.11	1.4	0.19	0.057
LTO cycle, kg	1 671	2.5	0.22	7.5	0.53	0.52
Per year, 8760 cycles, t	14 640	22	1.9	66	4.6	4.5

2.3.2 Dispersion modelling

Local scale dispersion model for mobile/line sources

A special version of a local scale dispersion model for vehicular pollution of CAR-FMI (Härkönen *et al*. 1995, Karppinen *et al*. 2000b) applicable to elevated vehicular sources like air transportation has been developed by the Finnish Meteorological Institute (Pesonen *et al.* 1996). The dispersion model is based on a partly analytical solution of the Gaussian diffusion equation for a finite line source (Luhar *et al*. 1989). The model includes the chemical transformation of the basic reactions of nitrogen oxides, oxygen and ozone. The model can be adjusted to site-specific conditions. (Varjoranta & Pietarila, 2002)

The model is based on the steady-state hypothesis, where emission rates and meteorological conditions are assumed to remain constant during each time-step (one hour) of the model. The one-hour air concentrations for each pollutant in the emission inventory are computed for the whole period of meteorological data on a receptor grid, which covers the whole application area. The one-hour mean values form the basis for statistical analyses for other averaging periods.

The input data for the model includes meteorological time series, emission data and regional background values for ozone and nitrogen dioxide. The meteorological time series includes hourly data for every parameter necessary for the application of the model.

For the model applications meteorological data sets of at least one year are usually used. The different parts of the LTO cycle are divided into a sequence of lines, with which the emissions of the aircraft are modelled. The emission data includes the location of each line, the traffic volume and the variation of speed of the aircraft on each line of the LTO cycle. The regional background concentrations are used in modelling the chemical transformation of nitrogen oxides. The applications of the line source model can be combined with a local scale model for stationary sources (UDM-FMI). The concentration fields calculated with the local scale model can be used to describe the background concentration of the application area of the line source model.

The primary results of the model are hourly time series of concentrations on each point of the calculation grid. Statistical analysis of this time series is used to evaluate statistical concentration quantities (for instance various means and percentile values). Isolines for these quantities can be analysed over the grid points. The results can be presented either as isolines on a map or in tabular form. The isolines give an instant indication of those areas, where the expected concentration levels of the pollutants exceed some prescribed value(s). By suitable choice of isoline separation these plots can be compared with national air quality guidelines or limit values.

The local scale model for mobile sources has been tested and validated against urban air quality measurements and against the experimental data of the field dispersion trials (e.g. Härkönen *et al*. 1997, Kukkonen *et al*. 2001, Karppinen 2000c, Pesonen *et al*. 1996).

Modelling the chemical transformation of nitrogen oxides

The traffic-originated NO is chemically transformed in the atmosphere into a substantially more harmful compound, NO_2. The transformation rate of NO into NO_2 depends on emission characteristics, meteorological conditions, solar radiation, local environment and background concentrations of various pollutants (NO, NO_2, O_3 and hydrocarbons).

The dispersion model applied in this study allows the chemical transformation of nitrogen oxides and takes into account all the above mentioned factors except for the background concentration of hydrocarbons. For this application, a modelling system has been developed which allows the chemical interdependence of the NO_x concentrations originating from various sources and the O_3 concentrations. See Karppinen *et al. (2000b*) for details.

The time variation of the regional background concentrations is evaluated. The regional background concentrations are based on the data from the monitoring stations. In order to filter out temporally high episodic concentrations originating from local sources, diurnal hourly average concentrations for each month are computed. Spatial pollution distribution from all stationary sources is summed to the regional background. This procedure produces a so-called first-order spatial background distribution for each pollutant (NO, NO_2 and O_3).

The contribution to the background caused by the mobile sources is evaluated. The mobile sources and the receptor points are sorted out in terms of their location with respect to the wind direction. The plume from the most upwind mobile source is allowed to interact chemically with the first-order spatial background distribution. This produces a second-order spatial background distribution. Correspondingly, the second most upwind mobile source is allowed to interact with the second-order spatial distribution. All the mobile sources are then treated consecutively.

The modelling system takes into account the depletion of O_3 in the oxidation process of NO to NO_2. However, the modelling system does not consider the reverse reaction, in which ozone is formed by the photolysis of NO_2. As the concentrations are considered on the local scale (time scale less than 1 h, length scale less than 20 km) the formation of O_3 is probably less significant than the dissociation of O_3. In that sense the simple treatment of chemical transformation of nitrogen oxides can be considered reasonable. In this study only the emissions of one LTO cycle were taken into account. No other local emissions were considered in the calculations. See also Appendix 2 for further discussion.

In this analyses the background concentrations of ozone and nitrogen dioxide are based on the measurements of the available monitoring stations nearest to the investigation area. For the ozone background, data from Aspvreten, and for the nitrogen dioxide, background data from Hoburgen, were used. Both ozone and nitrogen data were not available at the same station.

Meteorological methods and data

The basic meteorological parameters relevant for dispersion simulations are wind, ambient air temperature, boundary layer stability and mixing height. Wind determines the speed and direction of dispersion. Stability gives indication of the turbulent mixing rate inside the boundary layer. Turbulent mixing is the most important factor for pollutant dilution during transport. The mixing height describes the vertical extent of the plume.

Turbulence data and boundary layer height are not available from any routine base measurements. Indirect methods have therefore been introduced to calculate those parameters. The meteorological pre-processing model MPP-FMI developed at the Finnish Meteorological Institute (Karppinen *et al.* 1997 & 2000a) has been utilised in this study. This pre-processing model is based on a slightly modified version of the energy budget method of van Ulden and Holtslag (1985). This method evaluates the turbulent heat and momentum fluxes in the atmospheric boundary layer (ABL) by utilising routinely available synoptic weather observations. The parameterisation of the ABL height is based on the boundary layer scaling and meteorological sounding data. In absence of sounding data the mixing height is estimated analytically (Arya 1981, Kitaigorodskii & Joffre 1988). The output of the pre-processor consists of estimates of the hourly time series of the relevant atmospheric parameters (the Monin-Obukov length scale, the friction velocity and the convective velocity scale) as well as the boundary layer height.

Meteorological time series for dispersion simulations are compiled by interpolating the weather data to the site of application with a straightforward distance-weighted interpolation. Several synoptic stations can be included in the interpolation. Time series normally cover from 1 to 3 years of data, depending on the application. For model applications outside the Finnish borders, the pre-processor incorporates a routine to fetch synoptic data from the data archives of the European Centre for Medium Range Weather Forecasts (ECMWF) at Reading, the UK.

For this application the meteorological time series were compiled from the synoptic observations of the weather station situated on the Västerås airport. The observations of the year 1996 were used. After the year 1996 usable data was not any more available from Västerås due to the lack of nocturnal observations. The meteorological data used as input of the dispersion model was interpolated to a temporal resolution of one hour from the available routine observations made eight times a day.

The most frequent wind direction is southwest (about 25%) followed by north (16%) and west (15%). The speed distribution is dominated by speed categories of 2–4 m/s and 4–6 m/s.

The monthly boundary layer mixing rate and estimated mixing heights are computed by a pre-processing model. The mixing rate is computed as a function of the inverse of the Monin-Obukhov length. As usual strong mixing (unstable conditions) and high mixing heights occur mostly in summer time. Poor mixing is most frequent in wintertime. The most dominant mixing rate category is moderate corresponding to neutral or near neutral conditions in the atmospheric boundary layer of the investigation area.

2.3.3 Estimation of physical impacts

The exposure-response functions used, along with the reference studies, are presented in Table 2-2. Impacts of nitrates and sulphates (both secondary particles) as well as O_3 are assessed only on a regional level, as are material and crop impacts. Climate change impacts are relevant on a global level. Damages due to acidification are not assessed due to lack of reliable valuation methods (see next Chapter).

Table 2-2. Quantification of impacts (Friedrich & Bickel, 2001).

Pollutant	Impact category	Receptor	Reference study*
Morbidity			
PM, nitrates, sulphates	Asthma, bronchodilator usage	Adults	Dusseltorp et al., 1995
PM, nitrates, sulphates	Asthma, cough	Adults	Dusseltorp et al., 1995
PM, nitrates, sulphates	Asthma, wheeze	Adults	Dusseltorp et al., 1995
PM, nitrates, sulphates	Asthma, bronchodilator usage	Children	Roemer et al., 1993
PM, nitrates, sulphates	Asthma, cough	Children	Pope and Dockery, 1992
PM, nitrates, sulphates	Asthma, wheeze	Children	Roemer et al., 1993
PM, nitrates, sulphates	Congestive heart failure hosp. ad.	Elderly	Schwartz and Morris, 1995
O_3	Asthma attacks	All asthmatics	Whittemore & Korn, 1980
PM, nitrates, sulphates	Chronic cough	Children	Dockery et al., 1989
PM, nitrates, sulphates	Restr. activity days	Adults	Ostro, 1987
PM, nitrates, sulphates	Chronic bronchitis	Adults	Abbey et al., 1995
PM, nitrates, sulphates	Respiratory hospital admission	Entire pop.	Dab et al., 1996
PM, nitrates, sulphates	Cerebrovascular hospital admis.	Entire pop.	Wordley et al., 1997
CO	Congestive heart failure	Entire pop.	Schwartz and Morris, 1995
SO_2	Respiratory hospital admission	Entire pop.	Ponce de Leon, 1996
O_3	Respiratory hospital admission	Entire pop.	Ponce de Leon, 1996
O_3	Minor restr. activity days	Adults	Ostro & Rotschiled, 1989
O_3	Symptom days	Entire pop	Krupnick et al., 1990
Mortality			
PM, nitrates, sulphates	Chronic mortality (YOLL)	Entire pop.	Pope et al., 1995 (scaled down)
SO_2	Acute mortality (YOLL)	Entire pop.	Anderson et al., 1996 Touloumi et al., 1996
O_3	Acute mortality (YOLL)	Entire pop.	Sunyer et al., 1996
Material damage			
SO_2, acid deposition	Ageing of galvanised steel, limestone, natural stone, mortar, sandstone, paint, rendering, zinc		Tidblad et al., 1998
Crops*			
SO_2	yield change for wheat, barley, rye, oats, potato, sugar beet		Baker et al., 1986
O_3	yield loss for wheat, potato, rice, rye, oats, tobacco, barley, wheat		Fuhrer, 1996 (mod.)
Acid deposition	increased need for liming		CEC, 1993
N, S	fertiliser effects		Hornung, 1997
Climate change			
CO_2	Global warming		See chapter 2.3.4

* See Friedrich & Bickel (2001) for list of references.

Acidification and eutrophication

The European Commission has adopted a strategy to combat acidification, which is expected to significantly reduce the extent of the areas in the European Union where the tolerance of sensitive ecosystems to acidity is exceeded by 2010 (COM 1997). The target set out already in the Commission's Fifth Environmental Action Programme is that critical loads should not be exceeded. For each ecosystem acid deposition should be lower than the critical load. A critical load is defined as the highest deposition of a compound that will not cause chemical changes leading to long-term harmful effects on ecosystem sustainability.

While many of the direct impacts of SO_2, NO_x, NH_3 and VOC and related secondary pollutants (sulfate and nitrate aerosols and ozone) on health, building materials and crops have been estimated and quantified as well as valued in money, the impacts of acidification on ecosystems can not currently be quantified in a way which would support monetary valuation. The acidification strategy and later agreements, such as the Göteborg Protocol in 1999, demonstrate that policy makers across Europe place a high value on ecosystem protection. It could therefore be concluded that the associated benefits of acidification abatement would be noticeable.

There are no effect models available to quantify the expected damage to ecosystems resulting from the exceedance of critical loads. What can be estimated with modelling tools, is the change in ecosystem area, in which critical loads for acidification and eutrophication are exceeded. The unability to monetarise the impacts of acidification and eutrophication has been pointed out in the latest major valuation studies on EU-level (for example the ExternE projects and analysis for the European Commission on Economic Evaluation of Proposals under the UNECE Multi-effects and Multi-pollutant Protocol and Emission Ceilings for Atmospheric Pollutants).

The impacts of acid deposition of soils of managed agricultural systems (effects of fertilisation, ground disturbance, harvest) have been counteracted with liming. Although liming may eliminate the possibility of soil degradation by acidic deposition in well-managed land, the effectiveness of applied lime may be reduced, and application rates may need to be increased. Liming of lakes and waterways is also done. Sweden has the largest liming programme in the world. The Swedish government has allocated SEK 218 million for this purpose in 2003 corresponding to some 200 000 tonnes of limestone. Naturvårdsverket will distribute the major part of this to the counties.

Nitrogen is an essential plant nutrient, applied in large quantity to crops. The deposition of additional nitrogen to agricultural soils is thus beneficial, as long as the dose is not excessive. There is concern that prolonged deposition of N can lead to nutrient imbalance and that benefits from enhanced productivity are not sustainable.

The assessment of forest damage due to air pollutants is even more difficult than assessment of crop damage, largely because of the different life cycles involved. Forest soils are more or less undisturbed, allowing acidification to proceed over time. Despite alarm over forest decline, European trees are typically growing faster than before. One reason for this could be enhanced N deposition, providing a steady source of fertilisers to the forests. A possible negative consequence of this is that the increased growth might cause nutrient imbalance as the supply of other nutrients may not be sufficient to keep up with growth. Much of the forest in Europe is subject to stress of low temperatures, high winds, low soil nutrient status etc. and is therefore subject to a variety of stress. This is one of the reasons for the lack of unambiguous effect models for assessing the net effect of acid deposition on forests.

Naturvårdsverket recently published a study (Naturvårdsverket, 2002) on acidification in Sweden. According to the study the critical load for acidification and eutrophication is exceeded on a 2 – 3 times larger area than estimated earlier with a courser modelling grid. The areas in question are primarily surface waters and forest area soils. The analysis shows that the exceedance area with respect to critical load for acidification has diminished from 60% in 1980 to 41% in 1990 and further to 22% in 1997. Assuming that European emissions of sulphur and nitrogen emissions are reduced in accordance with the Göteborg Protocol, the critical load will still be exceeded on 13% of Sweden, equalling to 5.1 million hectars.

Concerning eutrophication by nitrogen of forest soils, the critical load was exceeded on 40% of Sweden's area in 1990 and on 30% of the area in 1997. A further reduction to 19% equalling 4 million hectars forest land, is expected by 2010.

The estimates given provide an overview of the current state of the deposition compared with critical loads. They do not, however, provide information on the actual state of the environment. This is because the areas have been subject to different depositions over the time.

In the Swedish environmental quality goals adopted by the Parliament, the target is that the deposition of acidifying substances should not exceed the critical load for soils and water. An interim target is that at the most 5% of all lakes and maximum of 15% of the total length of waterways should be subject to man-made acidification by 2010. The interim target for forest land is that the trend of increasing acidification should be broken and recovery should start by 2010.

2.3.4 Monetary valuation of damages

The fourth and final phase of the Impact Pathway Method consists of monetary valuation of the physical impacts caused by pollution concentrations locally, regionally and globally. In theory, this would call for information on damage costs at the particular location where the impacts are assessed. However, such information in complete form cannot be obtained quickly or at low cost for each assessment. Therefore, benefit transfers are commonly used, i.e. values are transferred from existing empirical studies to the case site following certain transfer rules.

As there is no complete, detailed list available on Swedish monetary values for damages of atmospheric pollutants, this pilot study has used the ones available from Friedrich and Bickel (2001). However, these values have been refined for the use of another research project, UNITE (2000).

In UNITE, the empirical findings on local and regional damage cost estimates listed by Friedrich & Bickel (2001) have been first refined into a list of average European damage cost values by purchasing power parity (PPP) weighting, and then transferred to Finland by another PPP adjustment (Bickel & Schmid, 2002; Tervonen *et al.*, 2002). Now, these Finnish values have been transferred from Finland to Sweden by another PPP adjustment.[2] For regional impacts, however, Sweden specific BeTa values have also been used.

It should be noted, that the origin of the monetary values for damages is in selected case studies from different countries. These results may not necessarily represent the true monetary value of damages taking place in Sweden, or in any other specific country as such. However, these values represent state-of-the-art studies, and are consistent with theory in the structure of cost components, including material costs, medical expenses, lost income, lost production and reduced/lost immaterial well-being. Better representation of Swedish damage costs can only be obtained by local valuation studies. The marginal cost assessment performed in this report can be repeated if values better representing Swedish circumstances would appear later.

The so-called end points of health impacts are described and valued e.g. as years of life lost (YOLL), or cases of illness or appearance of symptoms. The values used in this case study are presented in Table 2-3.

Table 2-3. Monetary values used for the valuation of local impacts at price level of 2000, factor cost without taxes (ExternE/UNITE values with modifications).

Impact	EUR	SEK
Year of life lost (chronic effects), 3% DR	79 502	733 231
Year of life lost (acute effects), 3% DR	136 769	1 261 391
Chronic bronchitis	146 447	1 350 649
Cerebrovascular hospital admission	14 792	136 426
Respiratory hospital admission	3 846	35 473
Congestive heart failure	2 911	26 844
Chronic cough in children	208	1 917
Restricted activity day	104	959
Asthma attack	74	681
Cough	36	336
Minor restricted activity day	36	336
Symptom day	36	336
Bronchodilator usage	34	316
Lower respiratory symptom	7	67

[2] Adjusted by OECD current purchasing power parity (USD 1999 - PPP; Finland 22743 - Sweden 23038), which is the most recent ratio available at http://www1.oecd.org/std/gdpperca.htm. The exchange rate EUR - > SEK used is 9,2228 based on January 2003 data.

The monetary values used for assessing regional impacts are usually presented as the value of damages per tonne of pollutant, as in Table 2-4.

Table 2-4. Monetary values used for the valuation of regional impacts, value per tonne of pollutant, at price level of 2000, factor cost without taxes. (ExternE/UNITE values with modifications).

	NO_x			SO_2	NMVOC	$PM_{2.5}$ **
	via nitrates	via ozone	Total	SO_2 and sulfates	via ozone	
			EUR/t			
crops		131	131	-8	94	
material				71	0	
morbidity	387	117	503	221	90	882
mortality	890	79	969	561	61	2 029
Total	**1 277**	**326**	**1 603**	**845**	**245**	**2 911**
			SEK/t			
crops	0	1 204	1 204	-77 *	866	0
material	0	0	0	659	0	0
morbidity	3 567	1 076	4 643	2 035	831	8 130
mortality	8 210	729	8 939	5 178	563	18 714
Total	**11 776**	**3 009**	**14 785**	**7 794**	**2 260**	**26 844**

* Sulphur has a fertilizing effect and therefore produces a positive externality for crops.
** Direct aircraft particle emissions can be assumed to have a diameter of less than 2.5 micrometer

BeTa database values

In addition to UNITE values, regional impacts are also assessed using the recent BeTa (Benefits Table) data. DG Environment has prepared the Benefits Table database of externalities for air pollution in Europe.[3] BeTa gives the marginal external costs of air pollution (SO_2, NO_x, VOCs and $PM_{2.5}$) in terms of euros per ton of pollutant for each EU Member State. Furthermore, these external costs are divided between rural and urban areas. The database is largely based on existing data generated by the ExternE project and it takes into account the approach to valuation of mortality recommended by economics experts at a workshop convened by DG Environment. Differences in these two approaches are discussed in Chapter 5.5.

Urban values of BeTa are not used in this study. This is due to that fact that urban values can be considered to be poorly applicable to aircraft emissions. Some tentative calculations (EUR and SEK per tonne) are, however, provided in Chapter 4.2.

[3] http://europa.eu.int/comm/environment/enveco/studies2.htm

Table 2-5. Monetary values used for the valuation of regional impacts, value per tonne of pollutant, according to the BeTa database, EUR/t.

	SO_2	NO_x	$PM_{2.5}$	VOCs
Austria	7 200	6 800	14 000	1 400
Belgium	7 900	4 700	22 000	3 000
Denmark	3 300	3 300	5 400	7 200
Finland	970	1 500	1 400	490
France	7 400	8 200	15 000	2 000
Germany	6 100	4 100	16 000	2 800
Greece	4 100	6 000	7 800	930
Ireland	2 600	2 800	4 100	1 300
Italy	5 000	7 100	12 000	2 800
the Netherlands	7 000	4 000	18 000	2 400
Portugal	3 000	4 100	5 800	1 500
Spain	3 700	4 700	7 900	880
Sweden	**1 700**	**2 600**	**1 700**	**680**
UK	4 500	2 600	9 700	1 900
EU-15 average*	5 200	4 200	14 000	2 100

* No values are defined for Luxembourg

Global warming

The monetary value given to a tonne of CO_2 is perhaps the most uncertain monetary weight that must be chosen when the impacts of emissions are valued. The harmful impact of global warming has traditionally been valued according to the damage values presented for a tonne of CO_2. As the estimates on damage costs tend to vary considerably from study to another, the use of avoidance cost estimates or estimates on the price of carbon emission permits have become attractive.

In the UNITE project for example, a European average shadow value of EUR 20/t CO_2 is used. This value represents a central estimate of the range of values for meeting the Kyoto target in the EU by 2010. The value is based on estimates by Capros and Mantzos (2000) and reflects abatement costs and therefore also the gives an indication of the possible price level of emissions allowances in the EU-wide system for trading with emissions allowances proposed by the European Commission. The possible impact on the CO_2-value of changes to the current energy taxation or emission charges is not reported.

The recent climate change policy literature suggests CO_2 allowance prices of below EUR 10 until the year 2012 (Varilek & Marenzi, 2001). Ultimately, all estimates concerning the price of CO_2 are highly uncertain.

Kågeson (2002) argues that the total marginal cost of achieving the emission reduction goal in Europe is the sum of the price on emissions allowances and the existing taxation on coal, oil and natural gas in Europe, which Kågeson estimated at EUR approximately 45/t CO_2 on the average. Taking this value and adding the UNITE-estimate of EUR 20/t CO_2 on emissions allowances results in EUR 65/t CO_2 – this value is used in the calculations in this study. It should be noted that this approach leads to a high CO_2 value in comparison with values based on damage or avoidance costs.

In addition to the general uncertainties in estimates of the market price on emission allowances, it should be pointed out, that the transport sector is not included in the EU emissions trading scheme. It should also be noted that the average European tax level referred to concerns fuels for energy use. The emissions reduction targets and costs of CO_2 abatement measures and CO_2 taxes in the transport sector on a European level are not specified.

3 AIR QUALITY IMPACTS OF AIRCRAFT LTO EMISSIONS

3.1 General

This chapter describes the fundamentally important intermediate results of the Impact Pathway Method, i.e. the pollution concentrations produced by the case aircraft. Six air pollutants were considered in the modelling, namely nitrogen oxides (NO_x), nitrogen dioxide (NO_2), carbon monoxide (CO), hydrocarbons (HC), particulate matter (PM) and sulphur dioxide (SO_2). In addition, the used models compute the ozone (O_3) depletion due to the chemical transformation of nitrogen oxides.

3.2 Results

The pollutant concentrations were computed in an area of 50 km x 60 km around the Västerås airport. The base grid mesh was 1000 meters. Around the airport the grid mesh was 100 meters on a rectangular area of 4500 m $\times$ 4500 m. Also some additional points were chosen so that the average concentrations for each statistical population area could be calculated. The total number of receptor points was 5757 with a population of 286 000.

The 1-hour air concentrations were computed over this grid for each pollutant and for all hours of the considered year (8784 hours). Other mean concentration fields like annual mean and moving 6-hour and 8-hour averages were accumulated based on the hourly values. Isolines for each quantity were then analysed over the grid points. The basic quantities are defined as follows:

- the annual mean is the true mean value over the whole year
- the highest moving 6-hour mean is the largest true mean value averaged over each moving 6-hour periods of the year for each grid point
- the highest moving 8-hour mean is the largest true mean value averaged over each moving 8-hour periods of the year for each grid point
- highest 1-hour mean is the maximum value for each grid point of the year i.e. the absolute maximum value for each grid point

It should be noted that neither the 1-hour mean nor the 6-hour or the 8-hour means refers to any single continuous pollution situation or single reference time. However, they represent the worst case over the whole period, which can (and usually does) happen at different times at each grid point.

Results of dispersion calculations are presented as isopleths in Appendix 1. The area presented in the figures covers the most part of the actual computation area.

The maximum values of the results from the dispersion calculations of the LTO cycle emissions are presented in Table 3-1. The numbers presented in the table are the highest of the maximum concentration values found in the calculation grid used in each case during the period of one year. The calculated maximum concentration values for annual means of nitrogen oxide (NO_x), nitrogen dioxide (NO_2), hydrocarbons (HC), particulate matter (PM) and sulphur dioxide (SO_2) and the maximum 1-hour and (moving) 8-hour averages for carbon monoxide are presented.

The table also includes (moving) 6-hour means for ozone (O_3). It should be pointed out that these figures for ozone represent just the depletion of ozone in the chemical transformation of nitrogen oxides. The chemical transformation sub-models used in this analysis do not take into account any formation of ozone. Formation of ozone is also very unlikely in the vicinity of emission sources due to the reaction between nitrogen oxide and ozone.

Table 3-1. The maximum calculated air concentrations (µg/m^3) of nitrogen oxides (NO_x), nitrogen dioxide (NO_2), ozone (O_3), carbon monoxide (CO), hydrocarbons (HC) and sulphur dioxide (SO_2) caused by the emissions of LTO cycle on the Västerås airport. Note that the figures calculated for ozone represent maximum depletion values.

Pollutant	Concentration, µg/m^3
NO_x annual mean	59
NO_2 annual mean	19
O_3 6-hour mean (depletion)	62
CO hourly mean	440
CO 8-hour mean	320
HC annual mean	2.8
PM annual mean	4.2
SO_2 annual mean	4.0

The computed air concentrations can be compared with air quality guidelines and limit values. A comparison of the greatest of the computed maximum concentration values in this study with the limit values of the European Council (EC/30/1999 and EC/69/2000) are presented in Table 3-2. Except for the annual mean of the total nitrogen oxides (NO_x) concentration, the greatest of the calculated maximum values are below the applicable EC limit values.

It is pointed out that the highest concentrations used in the comparison are from the receptor points situated on the runway area i.e. the limit value is exceeded only in the airport area. In addition, the NO_x limit value is set for the protection of vegetation. It is therefore highly questionable whether this limit value is applicable in the study area.

Table 3-2. The greatest of the maximum calculated air concentrations compared with EC limit values. Note that the limit value for NO_x is given for the protection of vegetation and the limit value for

SO_2 is given for the protection of ecosystems. The other presented limit values are given for human health protection.

Pollutant	averaging time	Maximum Calculated value [µg/m^3]	EC limit value [µg/m^3]	Calculated value/ EC limit value [%]
NO_x	annual mean	59	30	197
NO_2	annual mean	19	40	48
SO_2	annual mean	4	20	20
PM_{10} (phase 1)	annual mean	4.2	40	11
PM_{10} (phase 2)	annual mean	4.2	20	21
CO	8-hour mean	320	10 000	3

4 EMISSION COSTS

4.1 General

The marginal emission costs of LTO cycles at Västerås airport are presented separately for local, regional and global impacts. Calculations are based on assumptions given in Chapter 2.

Morbidity and mortality are assessed locally and regionally. In addition, impacts on building material and impacts on crops are assessed regionally. The impacts of global warming are presented separately without allocating the costs into any impact category. The marginal cost information produced is summarized in Chapter 4.5.

All results are shown in euros (EUR) and Swedish crowns (SEK). The results are discussed in this section and in the conclusions section. A critical overview on the methodology and background data is included in Chapter 5.

4.2 Local environmental costs

Annual local health impacts due to 8760 LTO cycles amount to approximately EUR 58 000 and SEK 532 000. The cost per LTO cycle is EUR 6.6 and SEK 61.

Table 4-1. Local health impacts, 8760 LTO cycles at Västerås airport.

Morbidity				**Unit cost**		**Damage costs**	
	Impact category	Receptor	Cases	EUR	SEK	EUR	SEK
PM	Asthma, bronchodilator usage	adults	0.56	34	320	20	180
PM	Asthma, cough	adults	0.57	36	340	20	180
PM	Asthma, wheeze	adults	0.21	7	70	2	20
PM	Asthma, bronchodilator usage	children	318	34	320	10 898	100 510
PM	Asthma, cough	children	549	36	340	19 981	184 280
PM	Asthma, wheeze	children	424	7	70	3 082	28 430
PM	Congestive heart failure hosp. adm.	elderly	0.00	2 911	26 840	11	100
PM	Chronic cough	children	0.58	208	1 920	120	1 110
PM	Restricted activity days	adults	25	104	960	2 550	23 520
PM	Chronic bronchitis	entire pop.	0.02	146 447	1 350 650	3 340	30 800
PM	Respiratory hospital admission	entire pop.	0.00	3 846	35 470	10	90
PM	Cerebrovascular hospital admission	entire pop.	0.01	14 792	136 430	90	830
CO	Congestive heart failure	entire pop.	0.33	2 911	26 840	960	8 850
SO_2	Respiratory hospital admission	entire pop.	0.00	3 846	35 470	8	70
Subtotal						41 092	378 970
Mortality							
PM	Chronic mortality (YOLL)	entire pop.	0.20	79 502	733 230	15 570	143 600
SO_2	Acute mortality (YOLL)	entire pop.	0.01	136 769	1 261 390	980	9 040
Subtotal						16 550	152 640
Total						**57 642**	**531 610**

Table 4-2. Local health impacts of one LTO cycle per pollutant at Västerås airport.

	EUR/LTO cycle	SEK/LTO cycle	EUR/a	SEK/a	Emissions, t/a	EUR/t	SEK/t
PM	6.4	58.6	55 694	513 650	4.5	12 323	113 654
SO_2	0.11	1.0	988	9 110	4.6	213	1 966
CO	0.11	1.01	960	8 850	22	43	398
Total	6.6	60.7	57 642	531 610	-	-	-

Environmental costs per tonne of pollutant are EUR 12 300 and SEK 114 000 for PM, EUR 210 and SEK 2 000 for SO_2, EUR 43 and SEK 400 per for CO.

If urban values of BeTa database would be applicable for aircraft emissions, the following values could be used for Västerås (2.86 times original urban values based on population):

- EUR 94 000/t $PM_{2.5}$
- EUR 17 200/t SO_2

It can be seen that these BeTa values are significantly higher than specific EUR/t results (Table 4-2) obtained in this study. The difference can be explained by the fact that aircraft emissions disperse rather well i.e. concentrations caused by 1 tonne of pollutants are considerably lower in comparison, say, one tonne of emissions from road transport. In addition, population density is much lower in areas where the highest concentrations due to aircraft emissions occur.

4.3 Regional environmental costs

The annual regional environmental costs of 8760 LTO cycles are estimated at SEK 1.1 million (Table 4-3) using UNITE based values. Allocated by LTO cycle, the cost is SEK 130.

Table 4-3. Regional impacts of 8760 LTO cycles and one LTO cycle, per pollutant and impact category at Västerås airport based on UNITE values.

Regional impacts	EUR/a	SEK/a	LTO cycles	EUR/cycle	SEK/cycle
crops	8 800	81 156	8 760	1.0	9.3
materials	329	3 034	8 760	0.038	0.35
morbidity	38 501	355 089	8 760	4.4	41
mortality	75 949	700 458	8 760	8.7	80
Total	123 578	1 139 737	8 760	14	130
By pollutant					
NO_x	106 047	978 048	8 760	12.1	112
SO_2	3 913	36 093	8 760	0.45	4.1
NMVOC	462	4 261	8 760	0.05	0.49
$PM_{2.5}$	13 156	121 336	8 760	1.5	14
Total	123 578	1 139 737	8 760	14	130

The annual regional environmental costs based on BeTa database values and 8760 LTO cycles are estimated at SEK 1.7 million (Table 4-4). Allocated by LTO cycle, the cost is SEK 200. BeTa data give generally somewhat higher environmental costs in comparison with UNITE data.

Table 4-4. Regional impacts of 8760 LTO cycles and one LTO cycles per pollutant and impact category at Västerås airport based on BeTa database.

Regional impacts by pollutant	EUR/a	SEK/a	LTO cycles	EUR/cycle	SEK/cycle
NO_x	171 896	1 585 364	8 760	19.6	181
SO_2	7 876	72 641	8 760	0.90	8.3
NMVOC	1 282	11 826	8 760	0.15	1.35
$PM_{2.5}$	7 683	70 859	8 760	0.9	8
Total	188 738	1 740 690	8 760	22	199

4.4 Global environmental costs

The costs related to global warming is based on a CO_2 value of EUR 65/t. The total annual costs are EUR 952 000 and SEK 8.78 million. Per LTO cycle the costs are EUR 109 and SEK 1000. Only impacts of CO_2 emissions on global warming are assessed in this context.

Table 4-5. Global impacts of CO_2 emissions.

	EUR/a	SEK/a	EUR/cycle	SEK/cycle
Global warming, CO_2	951 600	8 776 500	109	1 002

4.5 Summary of emission costs

All costs related to different impacts are summarised in the table below based on this pilot assessment (Table 4-6). The share of local impacts is approximately 5% of the total impacts using UNITE values. Regional impacts are 11% of the total impacts and the share of global warming (based on CO_2 emissions only) is about 84% when using the CO_2 value of EUR 65/t CO_2.

When BeTa input data is used for regional impacts, the share of regional impacts is larger i.e. 16%. The contribution of local impacts is 5% and global warming 80%.

The fairly small share of local impacts can be explained partly by the fact that concentrations caused by LTO cycle emissions are fairly low (see Table 3-1). The highest concentrations occur in the runaway area with negligible population density, and therefore costs are rather low.

Table 4-6. Summary of emission costs based on UNITE values.

	Local impacts			Regional impacts					Local+ regional	Global warming	Total
	morbidity	mortality	total	crops	materials	morbidity	mortality	total			
EUR/a	41 092	16 550	57 642	8 800	329	38 501	75 949	123 578	181 220	951 600	1 132 800
SEK/a	378 985	152 637	531 623	81 156	3 034	355 089	700 458	1 139 737	1 671 360	8 776 500	10 447 900
EUR/LTO cycle	4.7	1.9	6.6	1.00	0.038	4.4	8.7	14	21	109	129
SEK/LTO cycle	43	17	61	9.3	0.35	41	80	130	191	1002	1 193
Percent of total	3.6	1.5	5.1	0.8	0.03	3	7	11	16	84	100%

Table 4-7. Summary of emission costs based on BeTa database values.

	Local impacts			Regional impacts BeTa	Local+ regional	Global warming	Total
	morbidity	mortality	total				
EUR/a	41 092	16 550	57 642	188 738	246 380	951 600	1 197 990
SEK/a	378 985	152 637	531 623	1 740 690	2 272 313	8 776 500	11 048 800
EUR/LTO cycle	4.7	1.9	6.6	21.5	28	109	137
SEK/LTO cycle	43	17	61	199	259	1002	1 261
Percent of total	3.4	1.4	4.8	16	21	79	100%

5 CRITICAL OVERVIEW

5.1 Emissions

Emission data is based on input provided by the client representing state-of-the-art data concerning emissions from the case aircraft type. Note that the $PM_{2.5}$ data is associated with notable uncertainties. Uncertainties concerning PM emissions are discussed in detail in Chapter 2.3.1.

5.2 Dispersion modelling

The meteorological data used in this study is based on the synoptic observations made on the application site of the dispersion model simulations. Observations of the year 1996 were used. After that year the need to fill the lacking nocturnal observations of Västerås would have had a weakening effect on the spatial representativity of the meteorological data. As found earlier in a corresponding analysis made for ship traffic the wind statistics of the year 1996 agree quite well with that of the 30 years long normal period (Varjoranta *et. al. 2002,* World Survey of Climatology, *1981*)

The background concentrations of ozone and nitrogen dioxide are based on the measurements of the nearest available monitoring station to the study area. The ozone background data and the nitrogen background data were from different monitoring stations because the data were not available from the same station.

In the emission modelling it was assumed that emissions of one LTO cycle release every hour of the study period (one year). The calculation is based on hourly averages of emissions and meteorological parameters.

The performance of the model for mobile/line sources near ground level has been evaluated by comparing the calculation results with the available measurement data from the air quality monitoring networks (for instance Karppinen *et al.,* 2000b and 2000c) or from experimental field campaigns (for instance Härkönen *et al.,* 1997). The modified version of the line source model applicable to elevated line sources like the LTO cycle of aeroplanes at airports was developed in the middle of the 1990s.

During that development some evaluations of the predictions of the model against measured concentrations of NO_2 on the roof of a building in the Helsinki-Vantaa airport were carried out with encouraging results (Pesonen *et al.,* 1996).

The modified version of the mobile/line source model used in this study has been applied in evaluating the concentrations caused by the emissions of the ship traffic in several air quality studies (for instance Pesonen *et al.*, 1996, Pietarila *et al.,* 1997, Varjoranta *et al.*, 1999, 2000 and

2002). The model has also been used for the emissions of Helsinki-Vantaa airport (Pesonen *et al.*, 1996).

The modelling system takes into account the depletion of O_3 in the oxidation process of NO to NO_2. However, the modelling system does not consider the reverse reaction, in which ozone is formed by the photolysis of NO_2. As the concentrations are considered on the local scale (time scale less than 1 h, length scale less than 20 km) the dissociation of O_3 is probably much more significant than the formation of O_3. In that sense the simple treatment of the chemical transformation of nitrogen oxides can be considered reasonable.

The effects of other secondary pollutants, i.e. sulphates and nitrate aerosols on the local scale were not modelled. Some tentative calculations were performed during the project using average oxidation rates of SO_2 and NO_x to sulphate and nitrate aerosols. These results were not included in local impacts as the ExternE methodology suggests that damages are rather insignificant (Friedrich & Bickel, 2001) and they are not usually taken into account in studies using the ExternE methodology.

It should be noted, however, that in some studies local impacts of sulphates and nitrates are taken into account due to the fact that urban sulphate and nitrate concentrations can be somewhat elevated in comparison with rural measurement (see Gynther *et al*., 2000). These impacts can be a fairly significant part of local damage costs (even though the share of total costs is minor).

As far as the linearity of the results is concerned, only global results are strictly linear. The change in emissions correlates directly with costs.

Local impacts are non-linear as far as the formation of NO_2 is concerned. The direct effects of NO_2, however, due to lack robust exposure-response functions are not valued. The change in concentrations in other components correlates rather directly with the change in emissions and further with impacts and costs. In reality, the damage is not a strictly linear function of emissions, but with the small emissions in question, linearity may be considered as a reasonable assumption. This assumption is only valid when conditions (i.e. emissions, meteorology, emission height etc.) do not change.

Regional impacts are non-linear as far as ozone, sulphates and nitrates are concerned because of the complex atmospheric chemistry involved. Therefore, the changes of emissions do not necessarily correspond to similar proportional change in concentrations and costs.

The non-linear behaviour does not show in the results, as regional valuation in this study is based on average SEK/t monetary values for different pollutants. The effect of small emissions changes on concentrations from various sources could be theoretically assessed with complex modelling exercises, but it would in any case be very difficult to allocate the changes in an even way. It can be argued that marginal costs in many cases can be calculated accurately enough based on a methodology using an average approach.

5.3 Exposure-response functions

Exposure-response (E-R) functions are based on the latest set of functions available from ExternE projects (Friedrich & Bickel, 2001). These are in turn based on the latest applicable literature. The only major changes in E-R functions in recent years, is the scaling down of the particle functions of mortality in order to not over-estimate the damages related to particles. The E-R functions are based mainly on time series studies in the USA with historical estimates of exposure, which are high compared with current European estimates. There has recently been some discussion about the validity of this down scaling. There are also slight differences between ExterE/UNITE and BeTa data.

The ExternE methodology identifies several additional E-R functions. Due to uncertainties these are recommended only for sensitivity analyses. NO_2 is probably the most interesting component in this sense, especially, when considering that there are European limit values and several national guidelines for NO_2. There is some evidence that NO_2 is linked with respiratory hospital admissions and acute mortality. However, NO_x emissions play a major role in damage costs via regional nitrate impacts. At the local level, ozone depletion is excluded from the assessment.

In some case studies in Finland (i.e. Gynther *et al*. 2000, Otterström 1998, Torkkeli, 2000) forest damages due to elevated regional ozone concentrations have been assessed using regional values of EUR 45/t NO_x and EUR 6.5/t VOC. Due to uncertainties involved in impact assessment, forest damages have not been valued in this context. The inclusion of forest damages would slightly increase the total damage costs.

E-R functions for acidification and eutrophication are currently missing leading to an underestimate of the total damages (see also Chapter 2.3.3). Some experiments of valuation of acidification and eutrophication have been reported in literature, but these do not provide a ground for an attempt for valuation in this context.

There are also E-R functions available for some hydrocarbons (i.e. benzene, benzo-[a]-pyrene and 1,3-butadiene). Due to the uncertainties in emissions and concentrations, these E-Rs are not included in this assessment.

Compared with the particle (primary and secondary) related health impacts and the impacts of global warming, acidification, eutrophication and the impacts of hydrocarbons are expected to be of much lesser importance for the LTO-cycle studied here.

5.4 Monetary values

The monetary values given to the damages caused by airborne pollutants in this study must be considered with caution due to several reasons. The values have their origin in selected European studies on the costs of illness and environmental impacts of pollutants. These original studies may not have considered the cost structure of the detrimental impacts identically, since they have been performed in different situations and in different countries.

Although e.g. the cost of illness has a long tradition as a field of science, systematic costing procedures are still missing. For instance, the medical expenses of treating a particular illness or symptom differ by country. Furthermore, earnings per capita, national net production per capita and the value given to changes in personal well-being are empirically found to differ by the standard of living. Thus, the values used here, represent more or less European generalisations on the costs of damages due to air pollution. Only a list of detailed values produced for Sweden alone will represent the economic significance of the detriments of pollutants with the exactly correct weight.

The monetary weight of environmental impacts with most uncertainty, and at the same time with high relative importance, is the value of CO_2.

The CO_2-valuation used in the calculations is based on the abatement cost approach. With this approach the marginal cost to reach a given quantitative emission reduction target is calculated. An optimal emission charge or an optimal quantitative emission restriction is given by the intersection of the marginal damage cost and marginal abatement cost curves[4]. There is therefore, in general, a need to estimate damage costs as well as abatement costs.

In the case of CO_2, the range of marginal damage cost estimates is very large, reflecting the large fundamental uncertainties of the relation between CO_2-emissions and global warming as well as economic difficulties regarding the appropriate choice of discount rate and weighting between countries with very different income levels. There is no reason, in a scientifically based study, to hide neither these uncertainties, nor the differences in values between damage and avoidance cost estimates. Therefore, marginal damage cost estimates are presented here as well. Friedrich and Bickel (2001) present estimates of marginal damage costs using the so-called FUND 2.0 model with a time horizon of year 2100.

[4] See for instance Hanley et al. (1997), chapters 2-4, for a discussion of these concepts.

Table 5-1. Recommended marginal damage costs for ExternE (EUR per tonne of CO_2).

	Minimum	Central estimate	Maximum
Marginal cost	0.1	2.4	16.4
Assumptions			
Discount rate	3%	1%	0%
Equity weighting	EU impacts only	World average	EU values applied globally

These estimates are central estimates. Assuming a log-normal distribution of observations from a Monte Carlo uncertainty analysis[5], an 80 per cent confidence interval for the maximum estimate ranges from EUR 8.3 to EUR 39 per tonne of CO_2. The lower bound of an 80 per cent confidence interval of the minimum estimate equals EUR 0.04 per tonne of CO_2.

To conclude, marginal damage cost estimates span a large range of values from nearly zero to EUR 39. This upper bound is relatively close to some estimates of abatement costs. Also note that the impacts incorporated in models such as FUND 2.0 are only a share (of unknown size) of all climate change impacts. There is, though, a clear tendency that the most recent marginal damage cost estimates are lower than the most recent marginal abatement cost estimates. If this reflects the truth, then, for reasonable assumptions about the shapes and slopes of the marginal damage and abatement cost curves, the emission reduction targets underlying the abatement cost calculations above are too ambitious from an economic point of view.

5.5 Comparison between different studies

When comparing the results of one Impact Pathway exercise with results from other studies, it should be kept in mind that different input data, methodology and geographical coverage and different assumptions in applying the method in some other aspects might have been used. For these reasons, the results are likely to differ.

Especially important factors influencing the results are E-R functions for particles, monetary values (especially for CO_2) and geographical coverage of the study. If for example regional impacts were not taken into account or are only partly covered, this would have a significant impact on the results. This applies to the number of pollutant components considered in the study as well. Furthermore, the monetary values used for costing impacts may have their origin in different studies, and they may have been adjusted for application by different procedures.

[5] As suggested in Table 9.7 in Friedrich and Bickel (2001).

Friedrich & Bickel (2001) reviews two studies assessing aircraft emissions in Berlin and London. It should be noted that greenhouse gas emissions are valued with a low value of 2.4 EUR/t CO_2 (Table 5-2).

Table 5-2. Costs due to an LTO cycle at the Berlin-Tegel and the London-Heathrow airport, EUR/LTO-cycle.

	Local	Regional	Local+Regional	Global warming	Total
Berlin					
Boeing 737-300/400	4.4	26.8	31.2	4.2	35.4
London					
Boeing 737-200	27.2	29.7	56.9	6.8	63.7
Boeing 737-400	6.8	27.7	34.5	6.8	41.2

Without global warming, the results from Berlin and London are of the same magnitude as results obtained in this study. Costs for Västerås are somewhat lower reflecting mainly the lower population density in comparison with Berlin and London.[6]

The use of BeTa database values gives somewhat higher results for regional impacts in comparison with UNITE values. This can be explained by the higher external costs per one tonne of pollutant in Sweden in comparison with Finland (see Table 2-5 for details). This difference, in turn, can be explained by the higher population density and density of other receptors in Sweden.

Urban BeTa values are not used in this study due to the fact that dispersion of aircraft emissions are rather different in comparison with “typical” urban emissions.

5.6 Further research

This pilot study is a first assessment of emission costs of air transportation in Sweden using the Impact Pathway method. It has been performed for demonstrating what kind of damage cost information can be produced with the Impact Pathway Methodology for LTO cycles of a single type of aircraft.

In light of the results of this study, the following areas for further assessments seem justified:

- Firstly, the environmental costs of en route emissions, not addressed in this study, would be important to analyse in order to get a picture of the total emission cost of flights. There is currently weak knowledge in this area and further work should start

[6] Negative values are used for ozone in Berlin and London which lowers regional costs.

by building a knowledge base before attempting numerical assessments;

- Secondly, the value of the regional impacts of emissions in locations in other parts of Sweden, is expected to differ from that for Västerås. Further assessments of regional impacts are justified by the results for Västerås, for which their value was found to be about double the value of the local impacts;
- Thirdly, even if the value of the local impacts of emissions in Västerås were smaller that that of regional impacts, the situation might be different for an airport in a more densely populated area, such as the Bromma airport.

Further, due to its dominance, the value of greenhouse gas emissions deserves continued attention. Better data on particulate emissions is needed, requiring both research and measurements. Also the critical issues concerning valuation of damages (mortality, morbidity, crops, materials and possibly forest damages) with Swedish values would require a more detailed assessment. Furthermore, other environmental impacts of aviation should be assessed in economic respects (e.g. noise, waste and oil spills).

6 CONCLUSIONS

The emission costs per LTO-cycle of an aircraft of type B737-800 with a CFM56-7B24 engine at Västerås airport are estimated at EUR 130 (SEK 1 190) when EU research project UNITE -based values are used for the regional impacts. Using an abatement cost value of EUR 65/t CO_2 for greenhouse gases leads to global impacts dominating the total costs, having a share of 84% of the total emission costs. The value of local impacts (EUR 6.6 or about 5% of the total) is about half of that for regional impacts (EUR 14 or about 11% of the total). Particulate emissions cause almost all of the calculated local emission costs, while NO_x emissions cause 86% of the regional emission costs.

Using the values per tonne of pollutant included in the BeTa-database recently published by the European Commission for regional impacts, increases the value of the regional impacts to EUR 21, that is about one fifth of total emission costs, EUR 137 per LTO-cycle (SEK 1 260). The absolute value for local impacts is unaltered.

The UNITE-values used here are originally calculated to represent an average for Finland - corresponding specific numbers for Sweden were not calculated in UNITE. Both UNITE- and BeTa-values are calculated using the ExternE-methodology, even though there seems to be some differences in exposure-response functions and unit values for endpoint impacts. But more importantly, the BeTa-values are calculated separately for each Member State. The average values per tonne of pollutant are higher in Sweden than in Finland. This can leads to conclude that the results for regional impacts based on the UNITE-values in this study are underestimates.

The results suggest, that it is of interest to control particulate emissions for reducing local impacts (better emissions data and more research on the impacts are however needed), NO_x emissions for reducing regional impacts and CO_2 emissions in order to reduce global scale impacts. It should be pointed out, that the CO_2 value used (EUR 65/t CO_2) reflects abatement costs, not the damage, for which e.g. ExternE currently estimates a value as low as EUR 2.4/t CO_2. It should also be noted, that the knowledge base for valuing impacts on natural ecosystems, for example caused by acidification and eutrophication, still is weak and that they are therefore missing from the results presented here.

The calculated emission costs are linear even though the relation between emissions and resulting concentrations of pollutants is non-linear for some key pollutants. The dispersion model used on the local scale takes into account background concentrations and the non-linear formation of NO_2, but due to the lack of robust exposure-response functions for NO_2, non-linearity is not reflected in the results. The benefit-transfer method used for regional impacts does not take into account non-linearity.

The ER functions used are all linear as well as the unit values of specific impacts. Calculating the emission cost for one LTO cycle based on repeated LTO cycles over the year is also averaging, but this allows taking into account all possible meteorological conditions equally and therefore produces results with wider applications than such calculated for one specific LTO cycle at a specific moment. In general, while the real relations in the impact-pathway described by emission-concentration-impact-value are not strictly linear, the assumptions made in the study can be considered reasonable and the results being fair approximations of marginal costs, in particular since the emissions studied here are small.

The results of this pilot study suggest that further effort should be put on analysing environmental costs of en route emissions, assessing the value of regional impacts of emissions at airports in other areas in Sweden and assessing the value of local impacts of emissions from an airport in a more densely populated area than Västerås. Further, the valuation of global impacts is clearly a key issue.

The inclusion of other environmental burdens of aviation, in particular noise, would probably change the share of total local, regional and global impacts and would certainly give a more complete picture of the total environmental costs at stake.

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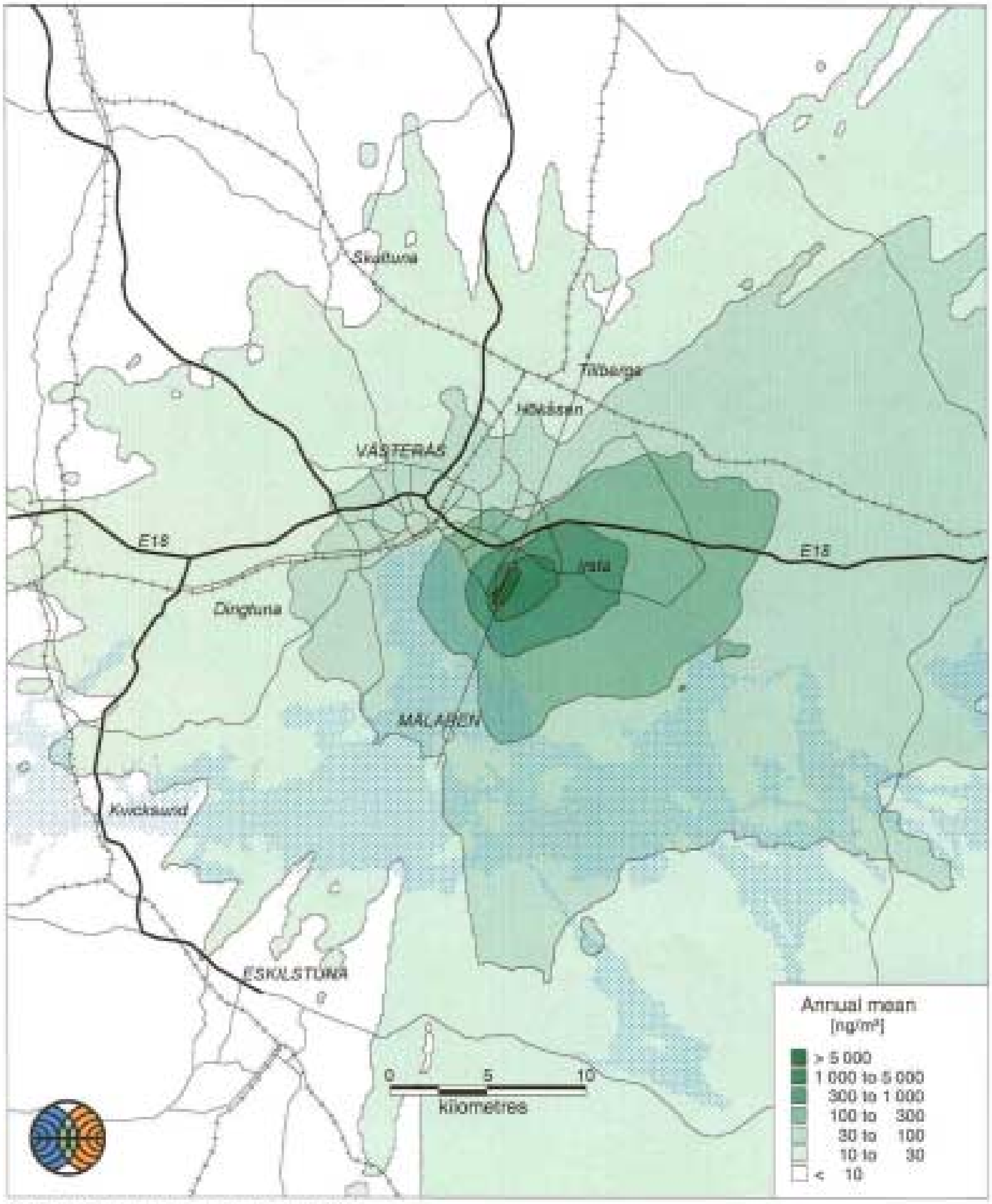
APPENDICES 1-3

Final Report, June 9th, 2003

Appendix 1 Results of the dispersion modelling

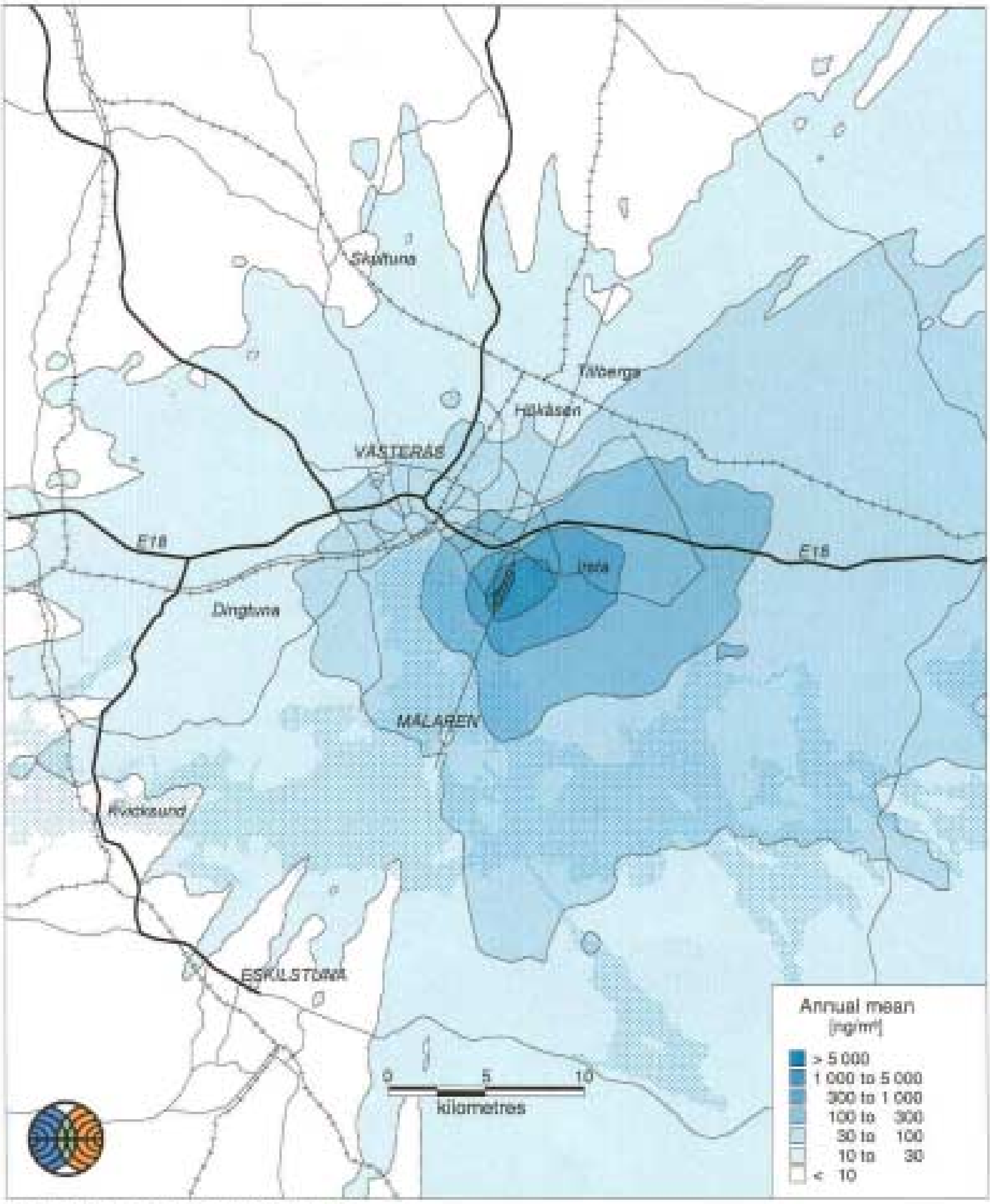
VÄSTERÅS AIRPORT



Finnish Meteorological Institute 2002

Figure 1. The annual mean NO_x concentration (ng/m^3).

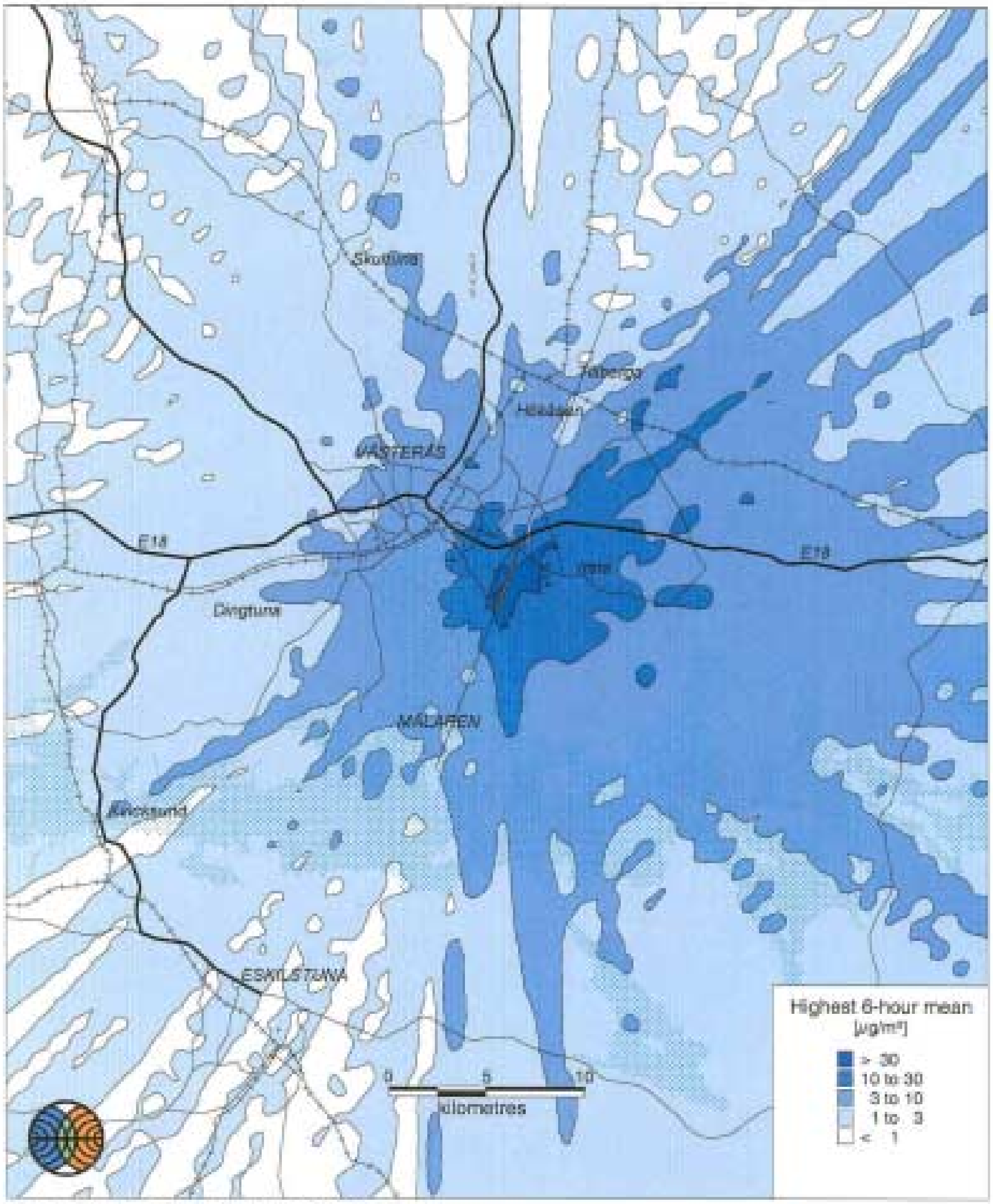
VÄSTERÅS AIRPORT



Finnish Meteorological Institute 2002

Figure 2. The annual mean NO_2 concentration (ng/m³).

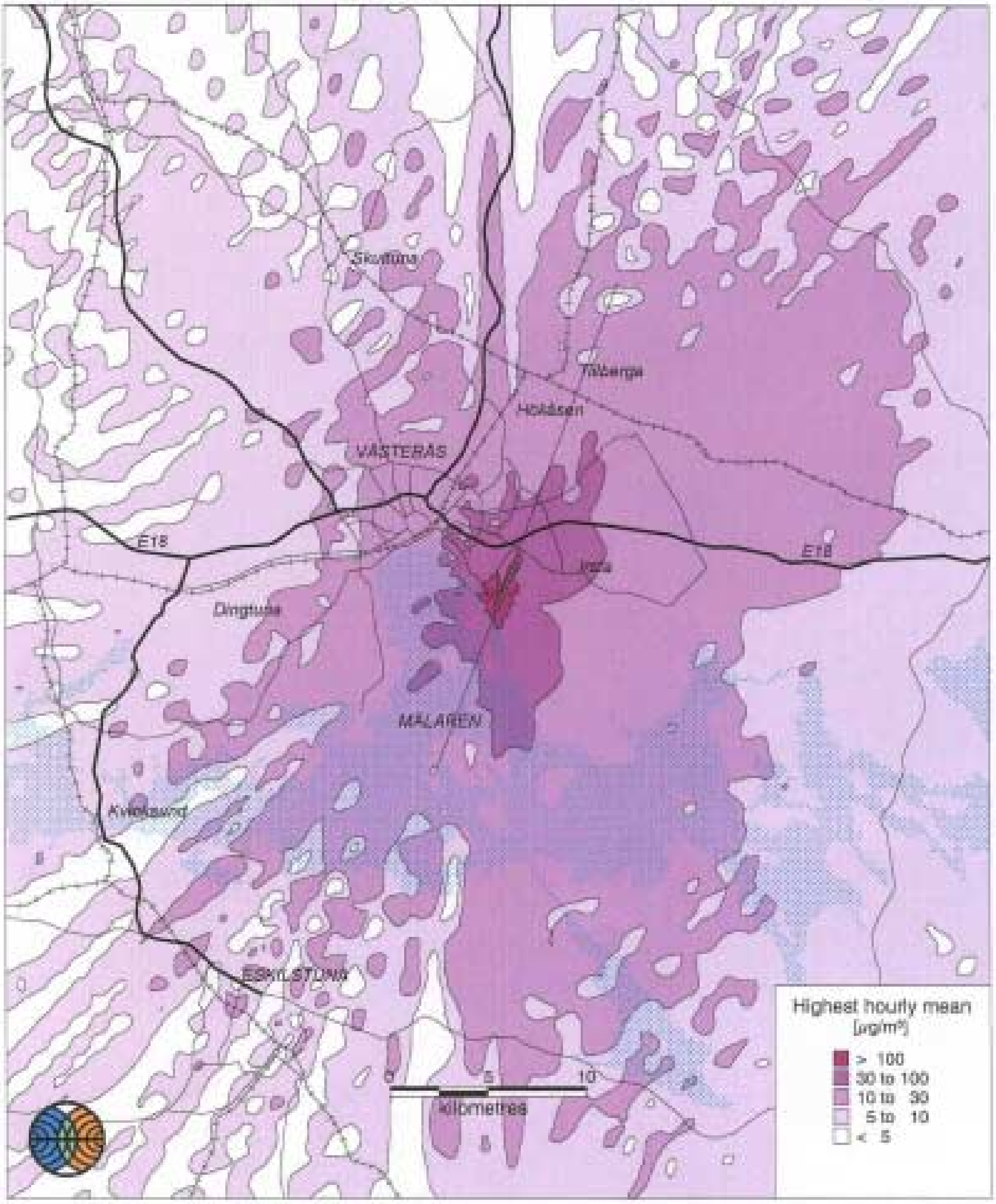
VÄSTERÅS AIRPORT



Finnish Meteorological Institute 2002

Figure 3. The highest 6-hour mean O_3 depletion ($\mu g/m^3$) caused by chemical transformation of nitrogen oxides.

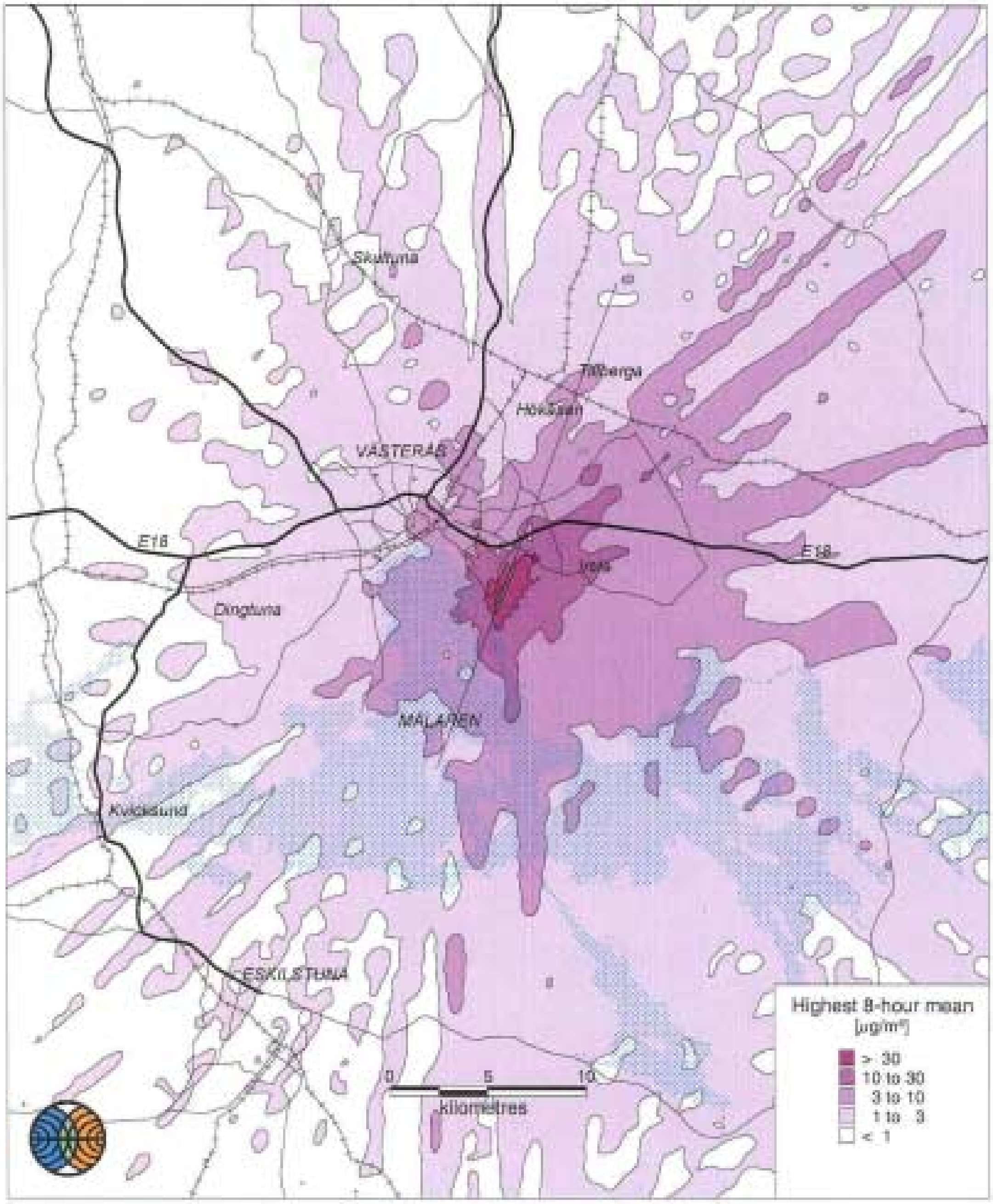
VÄSTERÅS AIRPORT



Finnish Meteorological Institute 2002

Figure 4. The highest hourly mean CO concentration ($\mu g/m^3$).

VÄSTERÅS AIRPORT



Finnish Meteorological Institute 2002

Figure 5. The highest 8-hour mean CO concentration ($\mu g/m^3$).

VÄSTERÅS AIRPORT

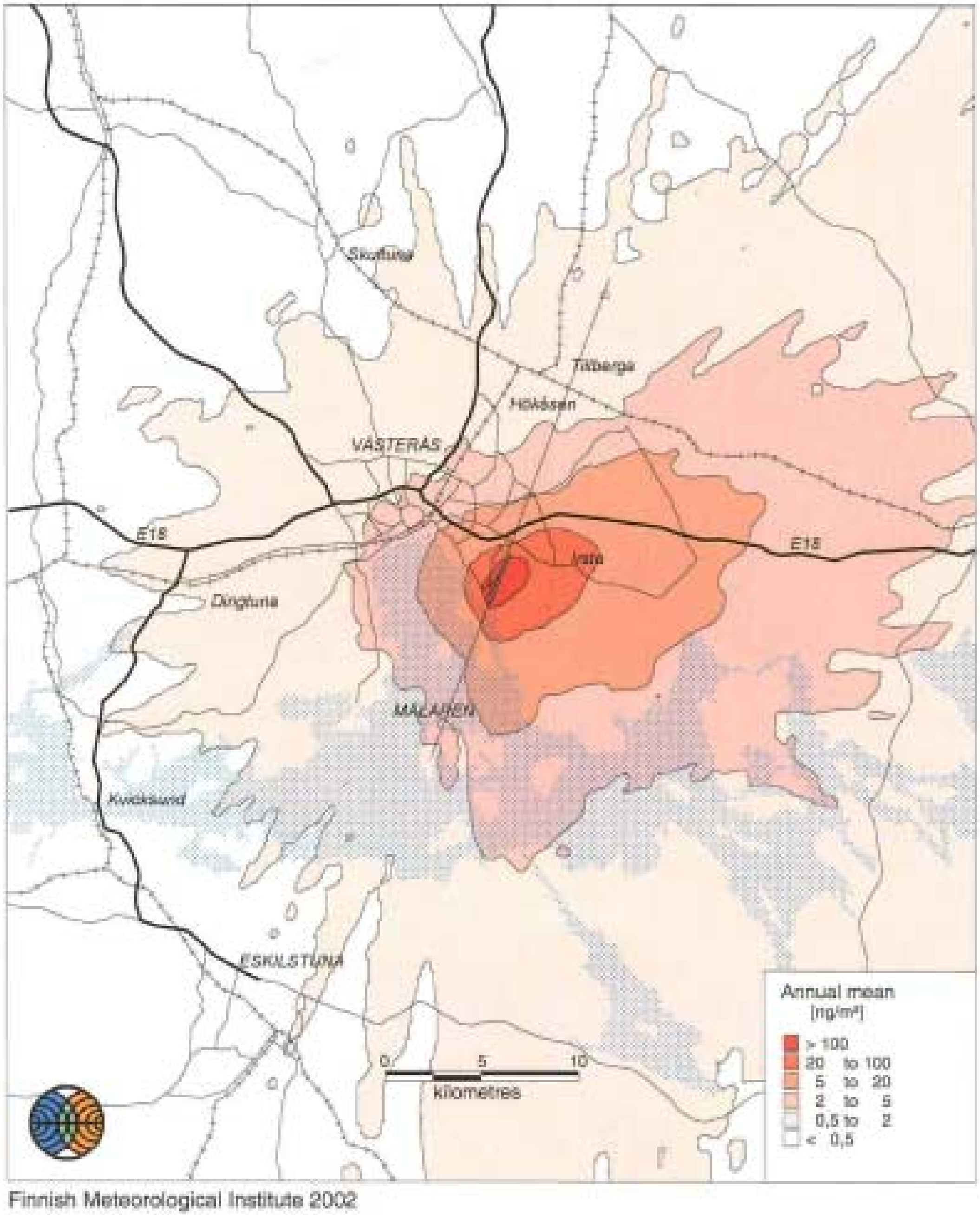
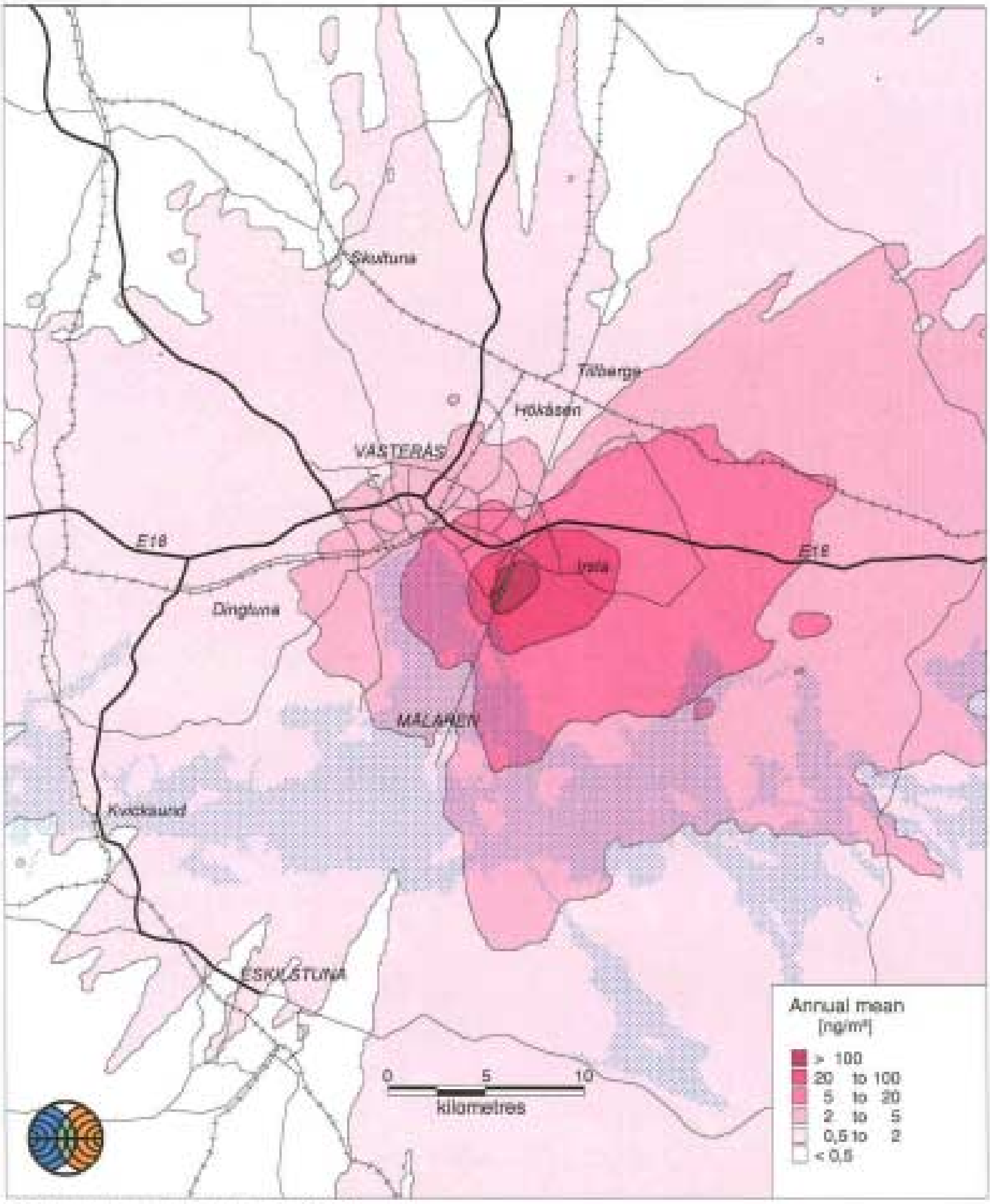


Figure 6. The annual mean HC concentration (ng/m³).

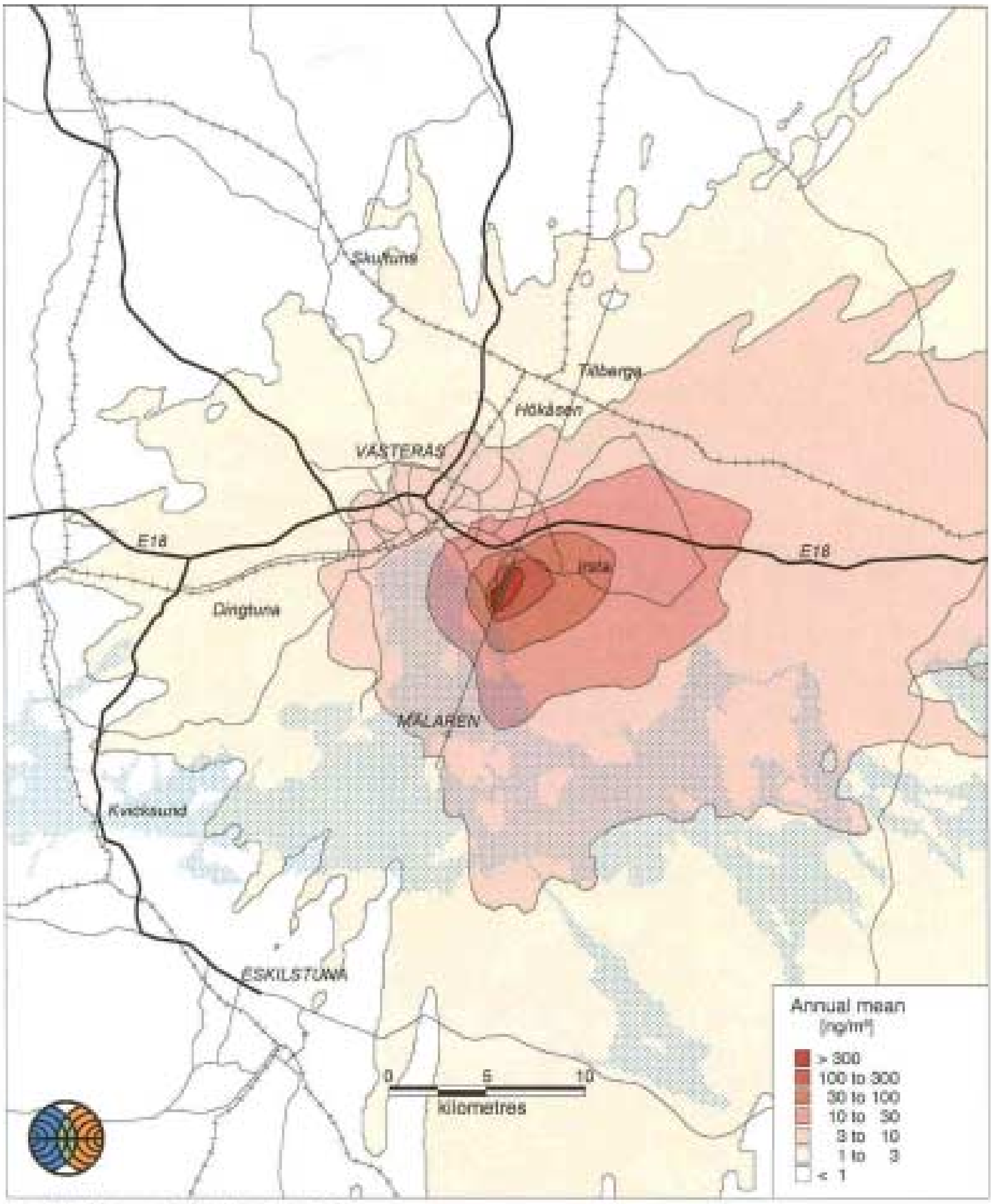
VÄSTERÅS AIRPORT



Finnish Meteorological Institute 2002

Figure 7. The annual mean PM concentration (ng/m³).

VÄSTERÅS AIRPORT



Finnish Meteorological Institute 2002

Figure 8. The annual mean SO_2 concentration (ng/m³).

Appendix 2 Role of ground level ozone

The assessment of the effect of emission reductions on local ozone concentrations represents the one most complex problem within ExternE methodology as well as in general in air quality management. The difficulties are caused by the strong non-linearity associated with ozone formation. Urban NO_x emissions usually cause a decrease of ozone concentrations due to the following chemical reaction: $NO + O_3 = NO_2$.

Effect is usually evident fairly far downwind in suburban and rural areas. In some photo-chemically very active cities such as Milan, Athens or Los Angeles chemical reactions may increase ozone levels closer to emission sources. Far downwind, in rural areas, NO_x emissions do increase ozone levels in most parts of Europe. Elevated ground level ozone concentrations are therefore clearly a regional problem especially in Scandinavia.

Practical example of atmospheric chemistry can usually be seen for instance in Stockholm where urban ozone concentrations in Södermalm are lower in comparison with suburban/rural concentrations of Aspvreten.[7] A similar effect is very evident in Helsinki, and several studies have been published on this matter over the course of years (e.g. Hämekoski & Lahdes 1990). The effects of VOC emissions further complicate the ozone chemistry. See for example Seinfeld (1986) for detailed explanation of ozone chemistry.

Due to the complex and “reverse” behaviour of ozone, possible impacts of ozone cannot be reliably assessed on the local scale.

[7] http://www.slb.mf.stockholm.se/slb/r2_databas_air.htm

Appendix 3 Damage cost estimates

In the UNITE (2000) research project, IER (at the University of Stuttgart) has produced a set of country specific unit values for assessing the damage costs of airborne pollutants (Bickel & Schmid, 2002; Tervonen *et al.*, 2002). These country specific values have their origin in Friedrich and Bickel (2001), but they have been averaged for the Member States of the European Union, and then transferred to Finland by purchasing power parity rules for implementing case studies within UNITE.

In Table 1, the European average values on local health impacts and the values transferred to Finland are presented at the price level of 1998. In Table 2, the same set of values is presented for Finland at the price level of the year 2000. In Table 3, the monetary values used in UNITE for calculating regional damage costs in southern Finland (a result of a dispersion modelling exercise by IER) are presented by tonne of pollutant, at the price level of the year 2000. The sets of values in Tables 2 and 3 have been transferred to Sweden according to a purchasing power parity ratio by OECD (http://www1.oecd.org/std/gdpperca.htm). The exchange rate EUR => SEK used is 8.6883 (November 2000; Nellthorp et al., 2001).

Table 1. Monetary values for health impacts in UNITE, (factor cost without taxes, $€_{1998}$).

Impact	European average	For	Finland
Year of life lost (chronic effects)	74 700	76 500	€ per YOLL
Year of life lost (acute effects)	128 500	131 600	€ per YOLL
Chronic bronchitis	137 600	140 900	€ per new case
Cerebrovascular hospital admission	13 900	14 230	€ per case
Respiratory hospital admission	3 610	3 700	€ per case
Congestive heart failure	2 730	2 800	€ per case
Chronic cough in children	200	200	€ per episode
Restricted activity day	100	100	€ per day
Asthma attack	69	71	€ per day
Cough	34	35	€ per day
Minor restricted activity day	34	35	€ per day
Symptom day	34	35	€ per day
Bronchodilator usage	32	33	€ per day
Lower respiratory symptoms	7	7	€ per day
Source: Own calculations based on Friedrich and Bickel (2001) and Nellthorp et al. (2001).			

Table 2. Country specific values for health effects in Finland according to UNITE at the price level of 2000 (Bickel & Schmid, 2002).

End point	€ at factor cost without taxes
Year of life lost (chronic effects), 3% DR	78468
Year of life lost (acute effects), 3% DR	134991
Chronic bronchitis	144543
Cerebrovascular hospital admission	14600
Respiratory hospital admission	3796
Congestive heart failure	2873
Chronic cough in children	205
Restricted activity day	103
Asthma attack	73
Cough	36
Minor restricted activity day	36
Symptom day	36
Bronchodilator usage	34
Lower respiratory symptom	7
Lung cancer risk	1179893

Table 3. Monetary values of damages caused by regional pollutants in southern Finland by the UNITE convention, presented at the price level of 2000 (Tervonen *et al.*, 2002).

NO_x

€ per tonne	via nitrates	via ozone	Total
crops		129	129
material			
morbidity	382	115	497
mortality	879	78	957
health	1 260	193	1 453
Total	**1 260**	**322**	**1 582**

$PM_{2.5}$

€ per tonne	$PM_{2.5}$
morbidity	870
mortality	2 003
Total	**2 873**

SO_2

€ per tonne	via SO_2 and sulfates
crops	-8
material	71
morbidity	218
mortality	554
health	772
Total	**834**

NMVOC

€ per tonne	via ozone
crops	93
material	0
morbidity	89
mortality	60
health	149
Total	**242**