A non-linear Eulerian approach for assessment of health-cost externalities of air pollution

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by
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Abstract
Integrated assessment models which are used in Europe to account for the external costs of air pollution as a support for policy-making and cost-benefit analysis have in order to cope with complexity resorted to simplifications of the non-linear dynamics of atmospheric sciences. In this paper we explore the possible significance of such simplifications by reviewing the improvements that result from applying a state-of-the-art atmospheric model for regional transport and non-linear chemical transformations of air pollutants to the impact-pathway approach of the ExternE-method.

The analysis shows that while external cost estimates are relatively robust for primary particles and SO_2*, the simplified linear approach fails adequately to capture the complex air pollution transport and chemistry related to nitrogen compounds and ozone formation, which is more suitably accounted for with an Eulerian module for regional dispersion and transformation. The simplified linear approach furthermore may amplify numerical noise in the atmospheric modelling. The significance is likely to be site- and source-specific but in the present analysis of three high-stack point sources in the Copenhagen area (Denmark) the more rigorous approach results in lower damage costs per unit of* NO_x *and allows for an improved assessment of ozone formation.*

1. Introduction.

According to Spash and Vatn (2006) the reliability of natural science data generally remains unquestioned in economic analysis of environmental change. In this paper, the issue of air pollution and its impacts on public health are addressed in order to demonstrate the usefulness and potential significance of a better ecologically informed approach to account for external costs in environmental economics.

Although some economists have been uneasy with the hypothetical answers obtained in response to hypothetical questions (cf. Diamond and Hausman, 1994) contingent valuation surveys (CV) and discrete experiments are practised comprehensively as a procedure for uncovering the willingness-to-pay (WTP) for changes in environmental goods. In the simple version of a CV survey respondents are questioned “top-down” about their WTP for a reduction in air pollution with only a crude specification of the health and environmental implications of that level of air pollution (e.g. Wang and Mullahy, 2006). More sophisticated “bottom-up” methods which were pioneered by the US EPA have been profoundly improved in the context of the European ExternE project, where valuation has been linked with modelling of specific health effect endpoints, i.e. bronchitis or asthma attacks, in relation to quantified changes in air quality as captured by means of simple atmospheric models (Krewitt, 2002; European Commission, 2002). The approach is known as the *impact pathway approach* as it relies on careful mapping of the impacts of changes in environmental quality in relation to various endpoints, including not only human health but also other relevant air pollution receptors such as buildings and vegetation (Holland, 1995). It does not take legislative thresholds (e.g. critical loads) into account, but ascribes effects even to low exposures where such effects are documented in the scientific literature. The purpose of the impact pathway approach is to identify the specific pathways along which the changes in environmental quality will have impacts and consequences to which human-beings can express their preferences in monetary terms. This procedure allows for a more detailed and realistic specification of impacts and their elicitation in CV surveys.

While the impact pathway approach by now is fairly established in policy support for estimating external costs from air pollution and also has triggered innovations of valuation research, the specific air pollution modelling employed for detailing the environmental consequences has been subject to less attention and deserves more scrutiny. In this paper, we explore the implications of applying a full-scale Eulerian air pollution model for the regional transport and atmospheric chemistry when accounting for the external costs of air pollution. This is accomplished by contrasting the methods and results of the ExternE Ecosense model with results from a new integrated model, EVA (Economic Valuation of Air pollution), based on the Danish Eulerian Hemispheric Model (DEHM). Before presenting the technical results of the analysis the paper provides some observations on state-of-the-art in externality assessments.

2. Recent advances in externality assessments.

Pigou (1920:184), originator of the concept of externalities, observed that air pollution caused substantial annual economic losses for "extra laundry costs, artificial light and damage to buildings". He did not include health costs, although he observed that at the turn of the century "in London owing to the smoke, there is only 12 percent as much sunlight as astronomically possible". More than 75 years elapsed between Pigou's observations and the first comprehensive assessment of air pollution externalities based on the impact pathway approach (European Commission, 1995; Krewitt, 2002). Subsequent assessments have refined and improved the basic 1995 assessment (Holland et. al., 1999; Friedrich and Bickel, 2001; European Commission, 2002; Holland and Watkiss, 2002; Friedrich et. al, 2004, Bickel and Friedrich, 2005).

While the assessments produced are similar in that health costs dominate most other cost categories, numerous methodological and empirical uncertainties have continued to underpin the attempts to account for the external costs of air pollution (Schleisner, 2000; Rabl et. al., 2004). These uncertainties have remained under discussion for several years but have gradually been narrowed as scientific evidence for the causal relationships has improved and as theoretical clarification in economics has progressed. In the following, three key uncertainties are summarized, together with how they have been dealt with in the ongoing work on air pollution externalities. For a more extensive review of the impact pathway methodology and its implementation we refer to the ExternE methodology volume (Holland et. al., 1999; Bickel and Friedrich, 2005; Andersen et. al., 2004), while more in-depth treatments of the particular issues are referenced below.

Exposure-response relations for particulate matter

The publication of the first externality assessment coincided with the publication of the findings relating to the relationship between mortality and levels of ambient concentrations of particulate matter of Pope and associates, made on the basis of the

American Cancer Society cohort (Pope et. al., 1995). The study, which was based on a comprehensive cohort of about 500.000 individuals, who were followed for more than 20 years and for which death certificates were obtained where relevant, found a statistically significant relationship between certain types of air pollution related mortalities and atmospheric levels of particulate matter. As such, it confirmed previous time-series studies on mortality effects from air pollution but was able to quantify the relationship on the basis of the more comprehensive data, including control for a range of intervening variables.

In the ExternE project the results of the Pope study were used to derive an exposure-response function for the relationship between mortality and levels of ambient air quality. The rather clear relationship, which indicated an additional early mortality of 0.4 percent for each microgram increase in particulate matter (PM10), was soon called into question. While the 1995 ExternE assessment applied the full exposure-response function, subsequent externality assessments chose to scale the function down to 1/3 of the published figure (e.g. European Commission, 1999; Friedrich and Bickel, 2001; Holland and Watkiss, 2002; Bickel and Friedrich, 2005). The Krewski reanalysis (2000), at the request of the US EPA, confirmed the findings of the original study, however, and the externality assessments reverted to using the original exposure-response function (for an overview of the health effects debate, see Pope and Dockery, 2006). Subsequent reviews by WHO committees for the European Commission have confirmed this basic approach.

Disentangling endpoints and avoiding double-counting

A more general concern with the impact pathway methodology, from an economic point of view, has been whether the splitting of air pollution effects into numerous, smaller mortality and morbidity effect endpoints related to individual pollutants would lead to double-counting of effects, and hence to exaggerated estimates for external costs. Table 1 provides an overview of the particular effect endpoints involved in the accounting for air pollution externalities. Many of the endpoints are similar for NO_x, SO_2 and PM, respectively. However, one needs to understand the atmospheric chemistry at play and the way in which SO_2 and NO_x translate into health effect endpoints. Because both SO_2 and NO_x form secondary particulates after transport and chemical transformation, it is really the effect of the two types of particulates, sulphate (SO_4^{--}) and nitrate (NO_3^{-}), that is at play rather than the primary emissions of SO_2 and NO_x per se. There are relatively few individual effects for the primary emissions of SO_2 and NO_x. Furthermore, one may note from Table 1 that the effects are frequently adjacent, so that bronchodilator use, restricted activity days and mortalities, while linked to the same pollutants, are mutually exclusive in their physical character. Where this is not the case, e.g. for restricted activity days, days with hospital admissions are deducted. While the basic relationship between particulates and mortality was established by a research team led by an environmental economist, i.e. C.A. Pope, it is appropriate to involve medical expertise for judgement on state-of-the-art in epidemiology. In the context of the EU's Clean Air for Europe programme, committees in the World Health Organization (WHO) were asked to provide peer review of the exposure-response functions applied in the externality assessments. This involvement has only led to minor revisions of the ExternE approach. The main issue that may need further attention is whether acute and chronic deaths have been sufficiently disentangled (chronic deaths are deaths that follow after a longer period of exposure).

Valuation of statistical lives and life-years

In ExternE the mortality effect dominates all other effects, including most other health effects. This is because of the valuation of human lives relative to all other goods in question, including morbidity effects. In theoretical terms, valuation does not address human lives per se but the value of preventing a statistical fatality; hence it is a change in risk that is subject to valuation (Nielsen, 2006). Conventionally WTP has been elicited for changes in risks that would save additional human lives, hence leading to derived figures for Value of a Statistical Life (VSL). With regard to air pollution fatalities, most of the victims are believed to be elderly; although this has been shown only in time-series studies and not yet in cohort studies, there appears to be consensus about this assumption, due to the specific mechanisms of air pollution death. Whether one should apply the standard procedure in transport economics and account for statistical lives according to the VSL-valuation tradition, or whether one should rather count the lost life-years and hence value according to the more recently developed VOLY-approach remains a significant methodological issue (VOLY abbreviates *value of life year*). While the initial ExternE study applied the VSL-approach, the VOLY-approach has since 1999 been implemented in ExternE (Rabl, 2006 provides the theoretical rationale for the estimation of life years lost instead of premature lives lost). OECD guidelines recommend that the two approaches are combined, so that VSL is applied for acute mortality while VOLY is used for chronic mortality (Pearce et. al., 2006). We here adopt the OECD recommendation and apply the life year value identified in the NewExt study (Friedrich et. al., 2004) as subsequently published by Alberini et al. (2006).

Uncertainties in the estimates of external costs are endemic, but following the extensive research and subsequent review process a certain degree of consensus has emerged in the literature as to the accounting for external effects of air pollution. Ambient concentrations of particulate matter are considered to affect public health, and there appears to be consensus that the most authoritative estimate of exposure-response functions for the purpose of externality assessments is provided by the research of Pope et al. (1995, 2002, cf. also Krewski et al., 2000). The range of morbidity effects involved have been identified with a view to avoid double-counting, and although an improved statistical basis for individual morbidity effects would appear desirable, the aggregated externality figures are not believed to be particularly sensitive to changes here. We here apply the basic methodology of ExternE, as documented in Holland et al. (1999), adapted to Danish circumstances and price levels, cf. Andersen et al. (2004). This represents a complex and profound aggregation of the knowledge base relating to the health effects from air pollution and readers not familiar with the basic methodology are referred to the above cited publications. Here, we focus particularly on the specification of the environmental consequences of emissions and explore the implications of the approach adopted for the atmospheric modelling.

Health effect endpoint	Exposure-response function per microgram/m^3/year	Valuation Euros (2004-prices)
MORBIDITY_PM		
Chronic bronchitis	8.2×10^{-5} (adults)	50360 per case
Restricted activity days	8.4×10^{-4} ÷ hosp. adm. (adults)	116 per day
Hospital admissions		
- respiratory	3.46×10^{-6}	7409 per case
- cerebrovascular	8.42×10^{-6}	9387 per case
- congestive heart failure	3.09×10^{-5} (>65years)	15450 per case
- lung cancer	1.26×10^{-5} (adults)	20150 per case
Asthma children (7,6%<15years)		
- bronchodilator use	1.29×10^{-1}	20 per case
- cough	4.46×10^{-1}	54 per case
- lower resp. symptoms	1.72×10^{-1}	14 per case
Asthma adults (5,9%>15years)		
- bronchodilator use	2.72×10^{-1}	20 per case
- cough	2.8×10^{-1}	54 per case
- lower resp. symptoms	1.01×10^{-1}	14 per case
IQ lead (Pb) (<1 year)	1.3	23715 per point
mercury (Hg) (foetus)	0.33	
MORTALITY		1941000 per case
Acute mortality_SO2	7.85×10^{-6}	71000 per yoll
Chronic mortality_PM	1.138×10^{-3} (>30 years)	2912000 per case
Infant mortality_PM	4.68×10^{-5} (<9 months)	1941000 per case
Acute mortality_O3	3.27×10^{-6} *SOMO35[1]	

Table 1. Exposure-response functions and unit values applied for assessment of the damage costs of air pollution with EVA and Ecosense respectively. Exposure-response functions are in accordance with Pope (2002) and Holland (1999) but without scaling and adapted to age distribution and mortality rate of the Danish population. For morbidity effects the monetary values follow Andersen et. al (2004); for mortality effects Pearce et. al. (2006) and Alberini et. al. (2006), cf. text (yoll is years of life lost). IQ-effects are based on Schwartz (1994), Budtz-Jørgensen et. al. (2004) and Salkever (1995)

3. Linearity and non-linearity in air pollution modelling for externality assessments

The Ecosense approach

The ExternE project has resulted in the computer software programme Ecosense, which integrates air pollution modelling with effects on human health and economic valuation (European Commission, 1999). The modelling of emissions and atmospheric transport and chemistry in Ecosense is based on local air pollution modelling as well as regional air pollution modelling, and Ecosense integrates the results from the two separate model complexes into delta concentrations in a standard EMEP grid comprising most of Europe. The delta concentrations express the marginal

[1] SOMO35 denotes the number of days where the diurnal max. 8-hour mean of 35 ppb is exceeded

change in air pollutant concentrations across Europe as a result of the particular point source for which the emissions are modelled in Ecosense. As such, the damage costs are highly site-specific. The Ecosense model has been made operational in a desk-top software program and has been distributed across Europe for the purpose of providing input of benefit estimates to cost-benefit analysis. For the analysis here we use version 4.0.

For local scale modelling in Ecosense the ISC-model (Industrial Source Complex Short Term Model, cf. Brode and Wang, 1992) is used. The model is a conventional Gaussian smoke plume model, which can provide annual mean concentrations of SO_2, NO_x and particulate matter as a result of point source emissions. There is no atmospheric chemistry included in the model formulation.

For regional scale modelling in Ecosense the WTM-model (Windrose Trajectory Model, cf. Derwent et al., 1988) is used. The WTM was developed more than 20 years ago for nitrate and sulphate air components. It is assumed in the model that the atmospheric transport takes place along straight lines up to several thousands of km from the receptor points. For each grid cell, it applies an average wind speed for each wind sector according to a frequency distribution of the wind directions (the windrose). As a characterisation of regional transport of air pollutants, a simplification on a substantial scale has been introduced by not referring to realistic atmospheric transport. As seen from inspection of Figure 1 trajectories of air pollutants do not run along straight lines but are in reality a result of the complex meteorological forces. The laws of physics even allow for the possibility that the wind takes one direction near the surface but a different, even opposite, direction higher up in the atmosphere; a feature of the earth's rotation known as the Ekman spiral, that together with the vertical mixing or the air pollutants significantly impacts the real dispersion and transformation of air pollutants.

The atmospheric chemistry in the regional WTM-model is relatively simplified too. Crucially the chemical reactions between nitrate precursors and the formation of ozone are not captured directly in the model. A separate SROM-model (Source-Receptor Ozone Model) based on statistical relations between sources and receptors has been used to derive ozone concentrations, but the feed-back mechanism on NO_x or VOC (Volatile Organic Compounds) concentrations via the photochemistry has not been accounted for. As a result, the resulting concentrations can be either overestimated or underestimated.

A further problem with the WTM-model is that it does not make full use of available meteorological data but applies an annual average of the meteorology. In this way local and regional variability in the transports of pollutants due to the ever changing weather patterns is not accounted for. If the prevailing wind is from the west, pollutants will only be expected to follow this direction. More accurate atmospheric chemistry-transport modelling that takes into account hour-to-hour changes in the weather (as e.g. wind speed and direction) requires much more computer capacity and could not be run on a desk-top PC.

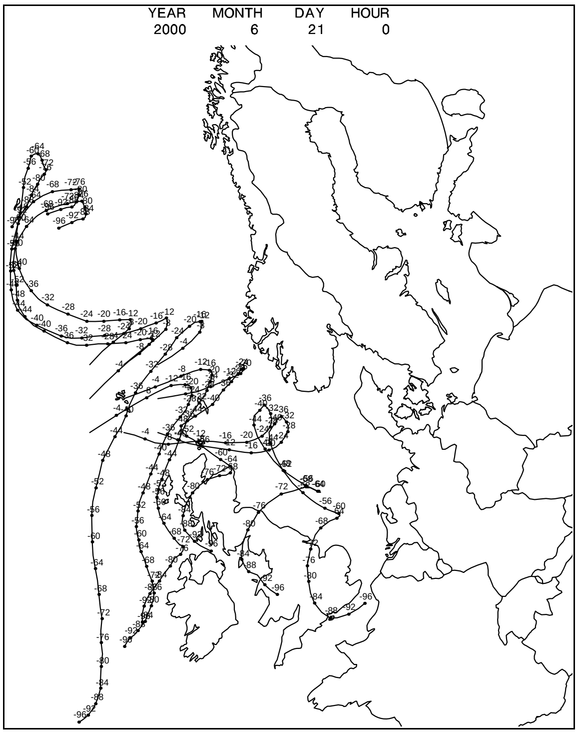

Figure 1 Examples of 96 hours trajectories made on basis of meteorological forecasting model (Ambelas Skjøth, 2002). The nine trajectories are started at the same time, but at slightly different starting positions. It can be seen in the figure that a relatively small change in the starting location can give very different trajectories. It illustrates the complex characteristics of atmospheric transport, and that even small uncertainties can give rise to very different results – e.g. very different contributions from different source areas.

A non-linear Eulerian approach

In order to explore the significance of the atmospheric modelling as a basis for accounting for external costs, a state-of-the-art non-linear Eulerian regional-scale air pollution model has been applied. The application is performed via the integrated model EVA (Economic Valuation of Air Pollution), which for the air pollution modules comprise a standard local Gaussian plume model OML (Operational Meteorological Air quality model, cf. Olesen et. al., 1992) and the regional Eulerian model DEHM (Danish Eulerian Hemispheric Model, cf. Christensen, 1997, Frohn et al., 2001; Frohn, 2004). The remaining modules of the impact pathway chain which include exposure-response functions and valuations are for consistency purposes held constant, so that the monetary damage cost output of Ecosense is compared directly with the monetary output of EVA. In the application, use is made of exposure-response functions adapted from the European CAFÉ-assessment, which have gone through a separate health expert assessment in the EU in consultation with WHO. The value estimates for health costs are adapted to Danish relative prices and preferences, based on the CAFÉ methodology.

Considering only mathematical models based on the fundamental description of atmospheric physical and chemical processes, there are basically two approaches available for long-range air pollution modelling; Lagrangian models and Eulerian models (cf. Peters et al., 1995). Models designed to study the change in chemical composition of air as it is moved with the flow over areas with differing emission sources are called Lagrangian models. Models constructed using a stationary grid in which the chemical composition changes in response to the air flowing in or out of the individual grid cells are called Eulerian models.

In the model domain of an Eulerian model, which is divided into grid cells in which the spatial and temporal concentration distribution is described for every time step, the change in concentration in each grid cell is calculated by taking into account all the sinks and sources of each chemical component in the model. In order to perform this calculation information is needed concerning the amount of air blowing in and out of every single grid cell (meteorological data), the amount of chemical components emitted from the surface (emission data), the amount of chemical components deposited on the surface (determined through a calculation of dry and wet deposition in the model) and the amount of chemical components which are transformed into other components (determined through a calculation of chemical reactions and rates). Within every grid cell the concentration distribution is assumed to be homogenous.

The results from an Eulerian model include the concentration distributions for the entire period of the model run, and over the entire domain of the model. The emphasis on capturing the non-linear chemical transformation of air pollutants implies that ozone formation is accounted for in a calculation procedure, which is integrated with the other emissions, and hence that the final concentrations arrived at are mutually adjusted to allow for feedback effects. The approach, as such, takes into account complex physical and chemical processes, and the resulting dispersion patterns for air pollution emissions are highly non-linear. Non-linearity is of particular relevance for certain emissions. While source-receptor relations of SO_2 and primary particulates in the main can be assumed to be approximately linear, this is not the case for NO_x, NMVOC and NH_3, due to the atmospheric chemistry at play.

In EVA, results from the local scale model and the regional scale model are obtained separately. The local scale model (OML) treats an area of 50 km x 50 km around the point source and the regional scale model (DEHM) treats the remaining area with grid cells of resolution 16.67 km x 16.67 km. Effectively, this means that the results from the local scale model substitutes the results from nine grid cells in the regional scale modelling around the source. The local scale model further decomposes the local area into 1 km x 1 km to allow for high-resolution modelling close to the source.

In the following the monetary output results from ExternE’s Ecosense model are compared with the monetary results from the integrated EVA-model. For consistency purposes the same exposure-response functions and monetary unit values have been applied in the two models, so that variations in results can be traced back to the atmospheric modelling. The specific results are sensitive to the approach applied for valuation of mortality; however valuation of mortality is in accordance with the recently published estimates from the NewExt project (Alberini et. al., 2006). We are here mainly interested in the differences between the two models and in validation.

4. Modes of electricity generation explored

As the ExternE method is site-specific and bottom-up oriented, the external effects are calculated for specific emitters. In the following, we apply the convention of the Danish ExternE-study (Schleisner and Nielsen, 1997) and model the external effects from the emissions of a modern fossil fuel-based combined heat and power (CHP) plant, Fynsværket (FV). In addition, we include results for two other plants in order to test how sensitive the external cost estimates are to site specificity. While Fynsværket has a rural location, the two other plants are located in a suburban area and the city centre of Copenhagen, respectively.

Fynsværket produces electricity and district heating from two units, from 1974 and 1991, respectively. Both units are primarily based on coal, but fuel oil is used to start generation. The older unit periodically substitutes coal with natural gas. For the scenario modelled here natural gas makes up 40 percent of the total fuel supply and coal 60 percent. The fuel mix is hence a reasonable match in relation to the general balance between coal and gas in the fossil fuel based part of the Danish power and heat generation system. Both units at Fynsværket are equipped with a filter that captures 99.5 percent of the fly ash. The younger unit in addition has a desulphurisation unit, but the plant has no de-NO_x unit for the investigated emission scenario which corresponds to the year 2000. As a result, emissions in the modelled scenario amount to 499 tonnes SO_2, 4403 tonnes NO_x and 147 tonnes primary particulates ($PM_{2.5}$) per year (ELSAM, 2000, 2003, 2004).

Amagerværket (AV), located in the centre of Copenhagen, produces electricity and district heating in three units, of which only the most modern unit from 1989 runs continuously. The two others are reserve units, built in 1971, and one of these is based on biowaste fuel only. In the modelled emission scenario fossil fuels account for nearly 95 percent of total fuel input, so Amagerværket represents a coal-based unit typical for a conventional Danish power plant. In terms of treatment Amagerværket has a desulphurisation and a de-NO_x unit at its main unit, while the two other units are equipped with low-NO_x burners. Emissions in the modelled scenario, which corresponds to the year 2004 amount to 750 tonnes SO_2, 1347 tonnes NO_x and 17 tonnes primary particulates ($PM_{2.5}$) per year (Energi E2, 2002, 2003, 2005).

Fynsværket and Amagerværket are among the eleven large fossil fuel based CHP plants in Denmark.Among renewables waste incineration is in Denmark more significant than wind energy (Energistyrelsen, 2005: 9. Hence, in order further to explore the ability of the models to capture the particular features of point source emissions in urban areas, a decentralised CHP unit, Vestforbrændingen (VF), was included in the study. Although a decentral unit in the energy supply system, Vestforbrændingen is in fact the largest municipal waste incinerator in the Nordic countries. It is based primarily on the use of municipal waste as fuel (500 000 tonnes annually) and produces heat and electricity. Vestforbrændingen, which is located in a suburb of Copenhagen, emitted 312 tonnes of SO_2, 787 tonnes of NO_x and 6.3 tonnes of primary particulates ($PM_{2.5}$) per year in the modelled scenario. It also emits heavy metals. In the analysis year (2000-emissions) Vestforbrændingen has de-NO_x on one of its units (Vestforbrændingen, 2000, 2005).

5. Transport, atmospheric chemistry and delta-concentrations.

The basic framework for atmospheric modelling in EVA is provided by the Danish Eulerian Hemispheric Model (Christensen, 1997; Frohn et al., 2001; Frohn, 2004).

According to the impact-pathway approach one needs to model the change in annual concentrations of air pollutants from a particular source in order to arrive at marginal damage costs. The marginal air pollution concentrations are here abbreviated as delta-concentrations. The delta-concentrations are required for the subsequent assessment of exposure and as a starting point for use of the exposure-response functions and the unit damage costs.

To arrive at robust delta-concentrations is an exercise that requires great care, as the delta-concentrations result from a baseline scenario without the source and an emission scenario with the source. In both scenarios there is a considerable level of background pollution that needs to be adequately captured by the atmospheric model. As the delta-concentrations are multipliers for exposure and damage costs the question arises as to how far one can trace the impact of a particular source on annual concentration values as the values becomes very small far from the source – i.e. what is the geographical distribution of the signal to noise ratio of the delta-concentrations.

As indicated above Ecosense extends its modelling to European regional scale, i.e. up to 2000 km from the source. In its linear modelling approach with transport along straight lines, and where only the concentrations arising from the source in investigation are considered, the small increments of delta-concentrations at each receptor point materialise over the entire European map. The simple approach for calculating atmospheric transport does not introduce any numerical difficulties with the very small numbers far from the source. However, when multiplied by large population numbers they may add considerably to the externality figures.

When calculating the delta-concentrations in the Eulerian model framework, two different calculations with the regional chemistry-transport model are in principle performed with two different emission scenarios (with and without the point source) and the resulting fields of annual mean values are then subtracted to give the geographical distribution of the delta-concentrations originated from the point source.

Most state-of-the-art Eulerian models use higher order numerical methods for solving the atmospheric transport due to the wind (also called the advection part). These higher order algorithms are relatively accurate, which is desirable since errors in atmospheric transport add up. However, such methods also have the problem of introducing unwanted numerical oscillations (numerical noise), which constitutes a major problem when applying the above mentioned methods of subtracting two different model results consisting of relatively large values to obtain relatively small values – i.e. the delta concentrations. In our first experiments, it was clear that the unwanted oscillations (the noise) were very large compared to the resulting delta-concentrations (the signal).

To avoid this problem a method known as “tagging” was introduced in the model. In this method, the atmospheric transport and transformation of the emissions from the point source in consideration is split from the “background” transport and transformation from all other sources in the model, but still calculated simultaneously together with the “background”. In practise, an additional field containing the concentrations arising from the source (the tag) is advected separately; in this way the delta-concentrations can be calculated individually and continuously and thereby the numerical noise can be minimised. For every time step in the model where the non-linear processes are treated (chemistry and wet deposition of all the species), the background field and the tag are added and the total changes in concentrations arising from the non-linear processes are calculated. Within the same time step, the resulting changes in concentrations from the non-linear processes are calculated for the background field alone. The additional impacts from the source on the concentrations (the tag or the delta-concentrations) can then be calculated by subtracting the two concentration fields from each other after e.g. the chemistry step in the model, and the two fields are again advected separately. In this way the non-linear effects from e.g. the background chemistry can be taken into account in the delta-concentrations without loosing trace of the part of the concentrations that arises from the source.

Figures 2, 3 and 4 show the resulting regional scale modelling outputs for each of the air pollutants, NO_x, SO_4^{--} and ozone (O_3); in contrast to the Ecosense model statistically significant impacts from the sources on grid cell concentrations can be identified only within a range of some hundred kilometres.

As a further observation and caveat for interpretation of the resulting health damage costs Copenhagen is located on an island where the Baltic Sea and the North Sea meet, and a considerable portion of the pollutants affect annual concentrations over sea territory only. Although the model captures these changes it should be emphasized that since no population is assumed to be exposed on sea territory these delta-values do not affect the subsequent damage estimates. Only delta-concentrations in grid cells with land territory cause human exposure, and hence external costs in this model which is confined to human health effects. Dispersal of emissions in a way that affects mainly the annual average concentrations over unpopulated sea territory is also a feature of local pollutants; in fact, for the majority of plants, the prevailingly western winds over Denmark cause the most significant changes in delta-concentrations to take place over the Baltic Sea. Power plants were located to take advantage of this situation. Similar emissions in a location in continental Europe, with greater population exposures and no sea territory over which to dilute, would be appreciated differently in terms of damage costs.

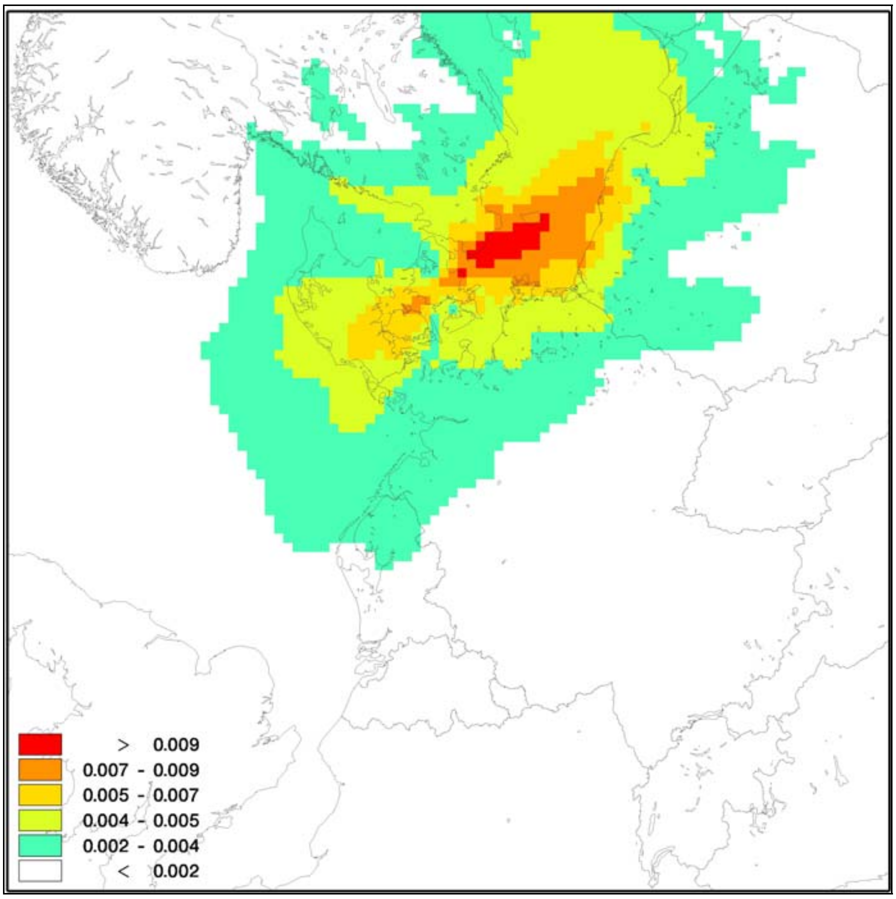

Figure 3: Changes in annual delta-concentrations of nitrate from NO_x emissions of Fynsværket according to the regional-scale model of EVA.

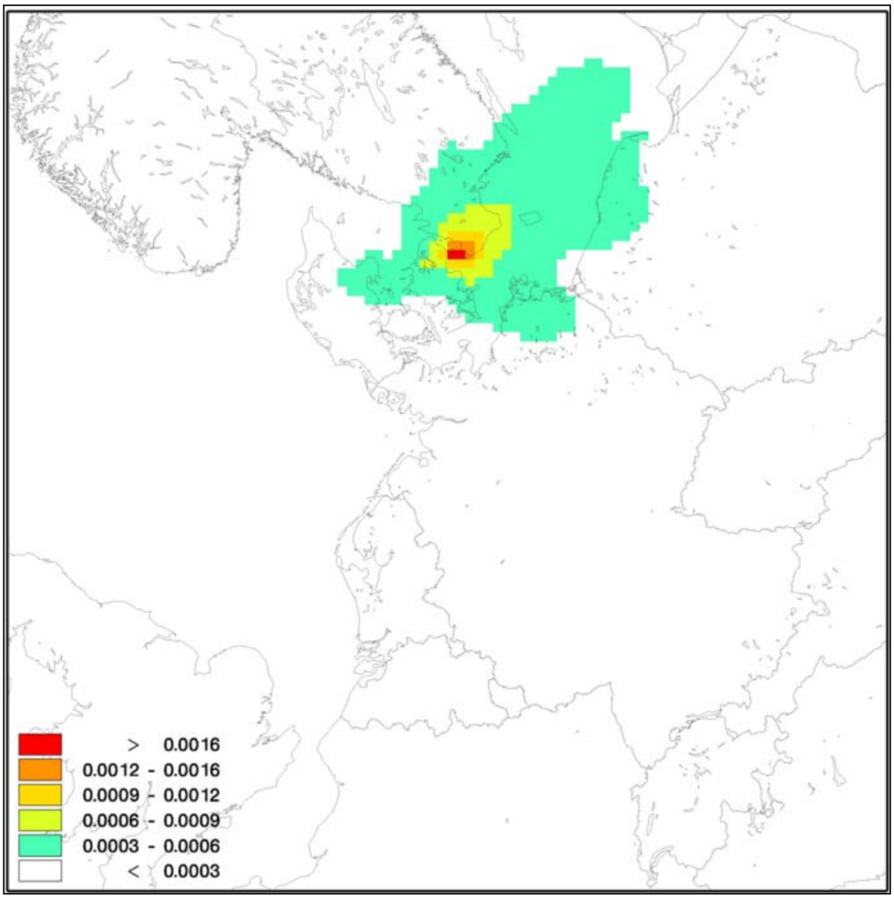

Figure 4: Changes in annual delta-concentrations of sulphate from SO_2 emissions of Amagerværket according to the regional-scale model of EVA.

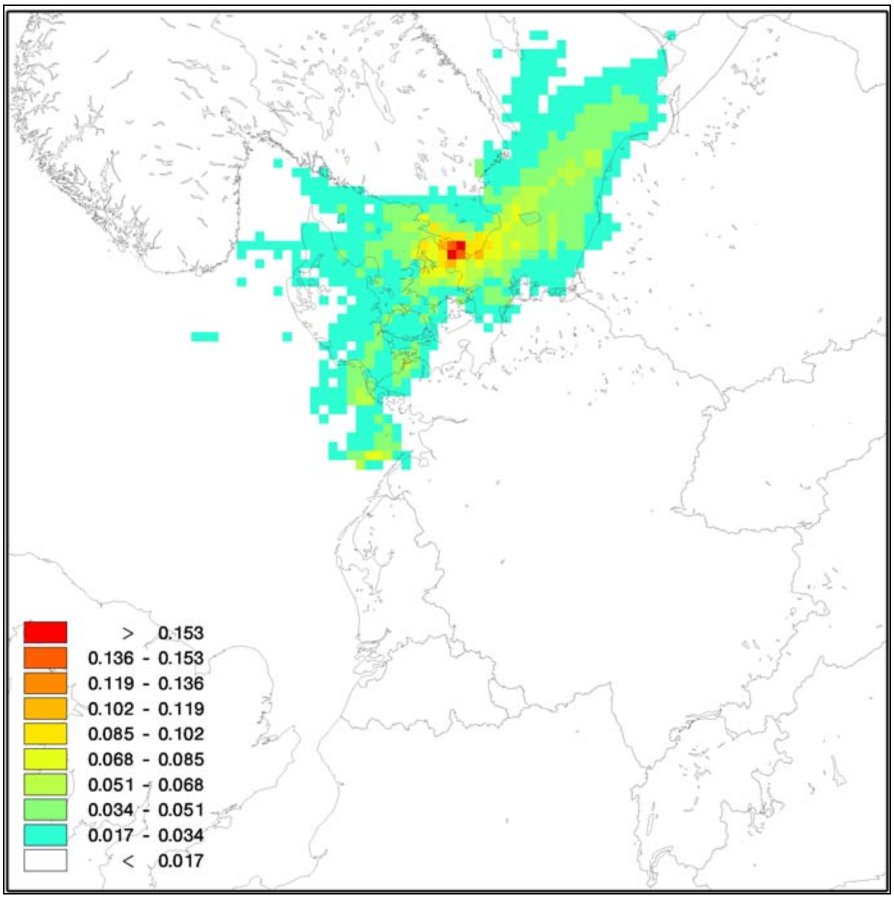

Figure 5: Changes in annual delta-concentrations of O_3 from emissions of Amagerværket according to the regional-scale model of EVA.

6. Comparing damage estimates from EVA and Ecosense.

NO_3^- and SO_4^{--} are secondary particulates which tend to be transported over longer distances, while $PM_{2.5}$ and SO_2 are emissions that produce damage predominantly in the local scale area (within the 50 km x 50 km area). Tables 2-4 provide an overview of the results from the two models in a format that, for each of the three plants, allows for direct comparison of external costs per kilogramme, dependent on the meteorological years.

The external costs as modelled in EVA are an average of three meteorological years, whereas in Ecosense, where the atmospheric long-transport module is rather static, there is little need for multiple meteorological years – only the local scale assessment depends on a meteorological dataset. For both models the background meteorology is the 1998 EMEP dataset. In EVA the three meteorological years are 2000-2002. As the models are used to provide for a statistical prediction of the external costs, the averaging of results arising from different meteorological conditions (three different years) is applied in order to achieve greater representativity; it does not restrict the results to these specific years. However, there is a significant difference as to whether meteorological conditions are averaged a priori, as in the Ecosense model, or whether it is done only after the richness of variation in the meteorological conditions have been explored, as in EVA.

Ozone health effects are included in EVA for days where the 8-hour maximum average exceeds 35 ppb/m^3. Ground level ozone is created mainly in more southern areas of Europe, and transported regionally to affect Denmark. There is a positive externality as ozone is removed in the immediate vicinity of the smoke gas plume due to reactions with NO_x. However, in the summer period there will be photochemical reactions which lead to formation of ozone along the smoke plume transport. The external costs will depend on the net effect of these two opposites. According to the EVA-modelling the balance for ozone is in urban areas predominantly a net positive effect; in Tables 2-4 these figures have negative signs as the effect must be deducted from the negative externalities. A caveat is that the health effects of NO_2 have not been included. In the chemistry, ozone reacts with NO to form NO_2. Although it is acknowledged that NO_2 has negative health effects, a separate exposure-response function that disentangles the specific NO_2-impacts on mortality and morbidity can not be specified.

External costs of heavy metals have been assessed in the EVA local scale model only. The damage figures relate to immediate externalities of impacts on *Intelligence Quotient* (IQ); effects from accumulation in the environment have not been accounted for.[2] Ecosense does include local scale modelling of heavy metals, however this feature has not been applied in the present study.

[2] Taking the relation between inhalation and ingestion in Spadaro and Rabl (2004) indicates a total external cost of 1279 €/kgPb and 234 €/kgHg for Copenhagen.

FV (rural)	EVA (m_2000)	EVA (m_2001)	EVA (m_2002)	EVA Average	Ecosense[3]
$PM_{2.5}$-primary	9,95	13,42	11,45	11,61	10,43
SO_2SO_2	1,11	1,46	1,12	1,23	2,76
SO_4SO_2	8,84	8,84	8,86	8,85	5,97
SO_2-total	9,95	10,30	9,97	10,07	8,73
NO_3/NO_x	5,91	7,45	7,10		
O_3/NO_x	0,02	0,03	0,05		
NO_x-total	5,92	7,48	7,15	6,85	11,37
Pb	2,17	1,94	2,00	2,04	-
Hg	0,68	0,61	0,63	0,64	

Table 2. Fynsværket/Coal&GasCHP: External costs of air pollution emissions in Euro per kilogramme (2004-prices) for three consecutive meteorological years.

AV (urban)	EVA (m_2000)	EVA (m_2001)	EVA (m_2002)	EVA Average	Ecosense
$PM_{2.5}$-primary	15,51	20,18	17,82	17,84	26,32
SO_2SO_2	2,22	2,71	2,28	2,40	6,16
SO_4SO_2	12,12	13,77	13,33	13,07	6,37
SO_2-total	14,34	16,48	15,61	15,48	12,55
NO_3/NO_x	5,41	6,87	6,51		
O_3/NO_x	-0,11	-0,11	-0,09		
NO_x-total	5,30	6,76	6,42	6,16	9,79
Pb	36,51	42,34	37,33	38,73	
Hg	11,62	13,47	11,89	12,33	-

Table 3. Amagerværket/CoalCHP: External costs of air pollution emissions in Euro per kilogramme (2004-prices) for three consecutive meteorological years.

VF (urban)	EVA (m_2000)	EVA (m_2001)	EVA (m_2002)	EVA Average	Ecosense
$PM_{2.5}$-primary	19,94	21,11	17,15	19,40	30,19
SO_2SO_2	3,22	2,80	2,03	2,68	7,03
SO_4SO_2	12,00	13,55	13,17	12,91	6,35
SO_2-total	15,23	16,35	15,21	15,60	13,39
NO_3/NO_x	5,42	6,93	6,49		
O_3/NO_x	-0,13	-0,13	-0,11		
NO_x-total	5,30	6,80	6,39	6,16	9,89
Pb	64,85	46,64	32,18	47,89	
Hg	20,65	14,86	10,24	15,25	-

Table 4. Vestforbrændingen/WasteCHP: External costs of air pollution emissions in Euro per kilogramme (2004-prices) for three consecutive meteorological years.

[3] Results for Ecosense deviate from figures reported in a recent ExternE methodology volume (Bickel and Friedrich, 2005). Differences are due to the use of relative Danish prices, as well as exposure-response functions as reported in Pope et. al. (2002). Bickel and Friedrich downscale certain ER-functions, e.g. for nitrate particles by about 85% (Bickel and Friedrich, 2005: 85).

Comparing the three tables it can be noted that EVA damage estimates are relatively consistent between the three power plants, taking into account the expected differences between urban and rural locations. Although physical NO_x emissions are substantially lower at the Amagerværket (due to low NO_x burners and de-NO_x equipment), the *per kilogramme* damage estimate is similar to that of Fynsværket, with a much higher absolute emission, as well as of Vestforbrændingen. This is not entirely unexpected for a regional pollutant as NO_x, when the sources are within short distances of each other and affecting the same population base, while transport and atmospheric chemistry are considered at both local and regional scale. There is consistency too for for SO_2 and $PM_{2.5}$ for the two units within the urban area, while Fynsværket as a unit in a rural area, as expected, records lower SO_2/PM damage costs due to lower population densities in the proximity of the plant.

The results from Ecosense provide a similar overall pattern for urban and rural sources respectively, as well as for the regional-scale pollutant NOx. Ecosense, despite shortcomings of atmospheric modules, produce reasonable approximations of damage costs for PM and SO_2 as compared to EVA. The NO_x damage costs per kg vary from about 9,79 € and up to 11,37 € and appear at first impression consistent. However, in comparison with EVA-results, the damages for NO_x are in Ecosense estimated to be more than 50 per cent higher. We have carefully checked all aspects of the calculations and regard the high Ecosense NO_x-damage figure arrived at as an implication of the linearised approach to regional-scale air pollution modelling and the absence in Ecosense of procedures to tag delta-concentrations and cut off numerical noise. Once minute delta-concentrations are multiplied with population figures across Europe, numerical noise inflates the external costs of NO_x. For PM and SO_2, on the other hand, the absence of atmospheric chemistry allows linearised approaches to produce reasonable approximations.

While Ecosense was created to allow comparably easy assessments for sites all over Europe, EVA was developed with precision for the situation in Denmark in mind. However, to the extent that population data is available EVA can be applied across all of Europe, and indeed, the northern hemisphere as such, as it refers to EMEP background data at this scale.

6. External costs per kWh.

In order to explore the external costs relative to the energy output of the three CHP plants, Table 5 provides an overview of the damages per kWh electricity. Such figures can be used when comparing energy production based on fossil fuels with energy production based on renewable energy (notably to justify the rate for the wind energy subsidy); the external effects can be included in such analysis to allow for a comparison of the relative efficiency. In economic analysis of climate policies the external costs are known as ancillary benefits of substituting carbon fuels with renewables or energy efficiency.

As the damage estimates per kilogramme were noted to be relatively robust in EVA, once several meteorological years had been averaged, we take these as the starting point for the assessment. However, it is the site-specific (plant-specific, cf. Tables 2-4 above) damage costs per kilogramme which are used. The years above are meteorological years; however, in the following we assess the external costs for the more recent years 2003-2005 on the basis of the available information on energy production. (In Denmark data on emissions and production is published in obligatory "green accounts" by power plant operators and other large emitters).

2003	**Emissions in 2003 (tonnes)**			**External costs 2003 (million €)**			**External costs 2003 (eurocents/kWh)**		
CHP	FV	AV	VF	FV	AV	VF	FV	AV	VF
$PM_{2.5}$	115	26	2	1,3	0,5	0,0	0,07	0,02	0,04
SO_2	917	2028	36	9,2	31,9	0,6	0,47	1,58	0,48
NO_x	4717	2515	529	32,3	15,8	3,3	1,63	0,78	2,81
Sum				42,9	48,2	3,9	2,16	3,37	3,33
2004	**Emissions in 2004 (tonnes)**			**External costs 2004 (million €)**			**External costs 2004 (eurocents/kWh)**		
CHP	FV	AV	VF	FV	AV	VF	FV	AV	VF
$PM_{2.5}$	21	17	3	0,2	0,3	0,1	0,01	0,02	0,07
SO_2	971	750	19	9,8	11,7	0,3	0,49	0,72	0,37
NO_x	5378	1347	294	36,9	8,4	1,8	1,85	0,52	2,24
Sum				46,9	20,4	2,2	2,35	1,26	2,68
2005	**Emissions in 2005 (tonnes)**			**External costs 2005 (million €)**			**External costs 2005 (eurocents/kWh)**		
CHP	FV	AV	VF	FV	AV	VF	FV	AV	VF
$PM_{2.5}$	20	18	3	0,2	0,3	0,1	0,01	0,02	0,08
SO_2	669	89	21	6,7	1,4	0,3	0,37	0,10	0,39
NO_x	4541	448	519	31,1	2,9	3,2	1,72	0,20	3,84
Sum				38,1	4,6	3,6	2,11	0,32	4,31

Table 5 Emissions and external costs (health) of CHP air pollution for two central and one decentral unit for the years 2003-5 (2004 prices).

There are significant variations in the figures, with one plant, Amagerværket, being recorded in the final year for significantly lower costs per kWh as compared to the other two. The main reason for this difference appears to be that Amagerværket has a de-NO_x unit, so that NO_x emissions are more than a factor 10 lower at Amagerværket The highest external costs are found at the waste incinerator, Vestforbrændingen, in the range of 3-4 eurocents per kWh (without inclusion of the external effects from other pollutants (heavy metals etc.)).

It can to some extent be misleading to assess the external costs against electricity production only, when in fact the three CHPs produce both electricity and heat. It depends on whether one regards the heat production as purely additional, or whether one rather regards the two as complementary. There are different methods available for splitting electricity and heat production; while the ExternE project uses the so-called Exergy method, we use here the Danish 200 percent model. The 200 percent model was developed after consideration of a range of splitting techniques, but its results are in fact rather close to what would be obtained by the Exergy model (Energistyrelsen, 2002). The 200 percent model regards heat production as marginal and accords the greater share of emissions to electricity. Table 6 provides for the three CHP plants an overview of external costs per kWh and per heat unit.

eurocents per kWh*	**External costs/kWh Fynsværket**			**External costs/kWh Amagerværket**			**External costs/kWh Vestforbrændingen**		
.	2003	2004	2005	2003	2004	2005	2003	2004	2005
Electricity	1,75	1,87	1,65	2,02	1,13	0,42	2,17	1,73	2,79
Heat	0,42	0,48	0,45	1,35	0,86	0,24	1,16	0,95	1,52
Sum	2,16	2,35	2,11	3,37	1,99	0,66	3,33	2,68	4,31

*Table 6. External costs (health) per kWh split on electricity and heat according to the Danish 200 per cent model (2004-prices). Three CHP plants in the greater Copenhagen area. Excluding micro-pollutants such as heavy metals (*For heat 1 kWh is 3600 kJ)*

Table 6 indicate that external costs for electricity range from as low as 0.40 eurocents/kWh and up to about 2.80 eurocents/kWh depending on fuels, flue gas cleaning and location of the power plant in relation to populated areas.

- the highest external costs are present in the case of the municipal waste incinerator located in a suburban area. There is relatively high exposure and no immediate advantage of the prevailingly western winds as the emissions are carried over large residential areas.

- the lowest external costs (0.42 eurocents/kWh) are identified for the most recent year of operation at Amagerværket, a central coal-based unit with advanced flue gas treatment with desulphurisation and de-NO_x. Despite its location in the centre of Copenhagen the external costs are modest; the high-stacks and prevailingly western winds go some of the way to explain this result; however, for 2005 there is an anomaly in that the more polluting reserve unit was not in operation, and the 2004-figure of 1.13 eurocent/kWh is likely more representative for operation with the phasing in and out of reserve units, than the 0.42 eurocents/kWh for continuous operation only.

- for Fynsværket the external costs of about 2.1 eurocents/kWh are fairly stable over the years. Whereas the plant in the modelled year uses a mix of coal and gas, the subsequent years were based on coal as the primary fuel. Once fuels have been converted to emissions it is these which determine the external costs. With its location in a rural area this plant has often been used as the reference for the average external costs related to fossil fuel electricity generation in Denmark.

The figures suggest that the advantage of renewable energies without air pollutants under the present circumstances is in the range from 1-3 eurocents/kWh depending on site-specificity and which type of electricity that is substituted. The advantage would be reduced if more flue-gas treatment is introduced, but it seems that it would require an exceptional effort to reduce it to less than 1 eurocent/kWh. In a liberalised electricity sector with limited means for investments the question is, furthermore, whether best use is made of the resources available by retro-fitting fossil fuel units, or by investing in new and cleaner modes of power generation. It appears that it would be particularly beneficial to substitute reserve units rather than base load units; this observation raises further issues about the ability of renewable energy to generate stable base load supply. These issues call for a more extensive analysis of the energy supply system and its external effects from which we abstain here (see an initial attempt in Andersen et al., 2007).

7. Conclusions

The damage estimates of the type presented here are often used in cost-benefit analysis to quantify the monetary values of reducing air pollution. Although historical data for emissions and meteorology has been applied it is important to stress that the damage estimates represent statistical *predictions* of damages, based on current knowledge regarding the relationships between exposure and health effects and the related costs.

It could be argued that damage estimates which involve a weather forecast are speculative in nature. The contribution of the EVA-modelling system to the impact pathway approach for externality assessment is, however, that the methodology is based on a carefully modelled average of the meteorological conditions; taking into account the physical complexities of the weather system and of the subsequent transport and chemical transformation of emissions in the atmosphere, including the relevant non-linearities. An extensive effort to capture the meteorology and the atmospheric transport and transformation adequately in the predictions of the health damages related to air pollution provides a method which avoids the numerical noise from atmospheric models, when assessing the external effects relating to the individual emissions. Further extension of the EVA modelling framework seems to be especially relevant for damagae cost estimates for other emissions where the non-linearities of atmospheric chemistry are at play, such as NMVOC and NH_3.

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References:

Alberini A, Hunt A, Markandya A, 2006. Willingness to pay to reduce mortality risks: evidence from a three-country contingent valuation study. Environmental & Resource Economics 33:251-264.

Ambelas Skjøth C, Hertel O, Ellermann T, 2002. Use of the ACDEP trajectory model in the Danish Nation-wide Background Monitoring Programme. Physics and Chemistry of the Earth 27: 1469-1477.

Andersen MS, Frohn LM, Jensen SS, Nielsen JS, Sørensen PB, Hertel O, Brandt J, Christensen J. 2004. Sundhedseffekter af luftforurening – beregningspriser [in Danish]. DMU Faglig rapport 507. Roskilde: National Environmental Research Institute , 85p.

Andersen MS, Frohn LM, Brandt J, Jensen SS. 2007. External effects from power production and the treatment of wind energy (and other renewables) in the Danish energy taxation system. In: Critical Issues in Environmental Taxation: International and Comparative Perspectives Volume IV (Deketelaere K, Milne JE, Kreiser LA, Ashiabor H, eds).. Oxford University Press, 319-336.

Bickel P, Friedrich R, eds. 2005. ExternE Methodology 2005 update. Luxembourg:European Commission.

Brode RW, Wang JF. 1992. User's Guide for the Industrial Source Complex (ISC2) Dispersion Models Volumes I-III. EPA-450/4-92-008a, EPA-450/4-92-008b, EPA-450/4-92-008c. Research Triangle Park, NC: . US Environmental Protection Agency.

Budtz-Jørgensen E, Debes F, Weihe P, Grandjean P. 2004. Adverse mercury effects in 7-year old children expressed as loss in IQ. Available: http://www.chef-project.dk/ [accessed 15 January 2008].

Christensen JH. 1997. The Danish Eulerian Hemispheric Model: A three-dimensional air pollution model used for the Arctic. Atmospheric Environment, 31:24, 4169-4191.

Derwent RG, Dollard GJ, Metcalfe SE. 1988. On the Nitrogen Budget for the United Kingdom and North-west Europe. Q.J.R. Meteorol. Soc. 114:1127-1152.

Diamond PA, Hausman JA. 1994. Contingent valuation – is some number better than no number. Journal of Economic Perspectives 8:4, 45-64.

European Commission. 1995. ExternE – Externalities of Energy, vol. 1-6. Bruxelles.

European Commission. 1999. ExternE – Externalities of Energy*:* Methodology annexes. Appendix I: The Ecosense model. Bruxelles. Available: http://externe.jrc.es/append.pdf [accessed 15 January 2008].

European Commission. 2002. External costs – research results on socio-environmental damages due to electricity and transport. EUR 20198. Bruxelles.

European Commission. 2005. ExternE – Externalities of Energy*:* Methodology 2005 Update. Bruxelles. EUR 21951.

ELSAM. 2000, 2003, 2004. Fynsværket – grønt regnskab [in Danish], Odense.

Energi E2. 2002, 2003, 2005. Amagerværket – grønt regnskab [in Danish], København.

Energistyrelsen. 2002. Model for fordeling af emission i forbindelse med kombineret produktion af el og varme [in Danish] 5.12., j.nr. 862-0014.

Energistyrelsen. 2005. *Energi-statistik 2004* [in Danish]. København.

Friedrich R, Bickel P. 2001. Environmental External Costs of Transport. München: Springer.

Friedrich R. (ed). 2004. NewExt Final report. Stuttgart: IER, 333p.

Frohn LM. 2004. A study of long-term high-resolution air pollution modelling [PhD thesis]. University of Copenhagen, Niels Bohr Institute of Astronomy, Geophysics and Physics. 482p.

Frohn LM, Christensen JH, Brandt J, Hertel O. 2001. Development of a high resolution integrated nested model for studying air pollution in Denmark. Physics and Chemistry of the Earth 26:10, 769-774.

Holland M. 1995. Assessment of the economic costs of damage caused by air pollution. Water and air and soil pollution 85:4, 2583-2588.

Holland M, Berry J, Forster D. eds. 1999. ExternE Externalities of Energy Volume 7. Methodology 1998 Update. Bruxelles: European Commission DG Research.

Holland M, Watkiss P. 2002. BeTa: Benefits Table database. Estimates of marginal external costs of air pollution in Europe. Version E1.02a. Available at: webref [accessed 15 January 2008].

Krewitt W. 2002. External costs of energy – do the answers match the questions? Looking back at 10 years of ExternE. Energy Policy 30, 839-848.

Krewski D, Burnett RT, Goldberg MS, Hoover K, Siematycki J, Jerrett M, Abrahamowicz M and White W. 2000. Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality. Special Report. Boston: Health Effects Institute.

Nielsen JS. 2006. Valuing changes in mortality risk. Health Economics Papers 2006(3): 137 pp. Odense: University of Southern Denmark.

OECD. 2006. Purchasing power parities. Main Economic Indicators. Paris.

Olesen HR, Løfstrøm P, Berkowicz R and Jensen AB. 1992. An Improved Dispersion Model for Regulatory Use - The OML Model in; Air Pollution Modelling and Its Applications IX (Dop, H. van and Kallos, G). Plenum Press, 29-38.

Pearce D, Mourato S, Atkinson G. 2006. Cost-benefit analysis and the environment. Paris: OECD

Peters LK, Berkowitz CM, Carmichael GR, Easter RC, Fairweather G, Ghan SJ, Hales JM, Leung LR, Pennell WR, Potra FA, Saylor RD, Tsang TT. 1995. The current state and future direction of Eulerian models in simulating the tropospheric chemistry and transport of trace species: A review. Atmospheric Environment 29:2,189-222.

Pigou AC. 1920. The Economics of Welfare. London: Macmillan.

Pope CA, Thun MJ, Namboodiri MM, Dockery DW, Evans JS, Speizer FE, Heath Jr CW. 1995. Particulate air pollution as a predictor of mortality in a prospective study of US adults. Am J Respir Crit Care Med 151: 669-674.

Pope CA, Burnett RT, Thun MJ, Calle EE, Krewski D, Ito K, Thurston GD. 2002. Lung cancer, cardiopulmonary mortality and long-term exposure to fine particulate air pollution. Journal of American Medical Association 287:9, 1132-1141.

Pope CA, Dockery DW. 2006. Health effects of fine particulate air pollution: lines that connect. Journal of Air and Waste Management Association 56: 709-742.

Rabl A. 2006. Analysis of air pollution mortality in terms of life expectancy changes: relation between time series, intervention and cohort studies. Environmental Health 5:1

Rabl A, Spadaro J, van der Zwaan B. 2004. Uncertainty of air pollution cost estimates: to what extent does it matter ? Environmental Science and Technology 39:2, 399-408.

Salkever, DS. 1995. Updated estimates of earnings benefits from reduced exposure of children to environmental lead. Environmental Research 70:1-6.

Schleisner L, Nielsen PS. 1997. ExternE national implementation Denmark. Risø-R-1033. Roskilde: Forskningscenter Risø.

Schleisner L. 2000. Comparison of methodologies for externality assessment. Energy Policy 28:15, 1127-1136.

Schreyer P, Koechlin F. 2002. Purchasing power parities – measurement and uses. Statistics Brief 3. Paris: OECD.

Schwartz J. 1994. Low-level lead-exposure and childrens IQ – a meta-analysis and search for a threshold. Environmental Research 65:1, 42-55.

Spadaro J, Rabl A. 2004. Pathway analysis for population-total health impacts of toxic metal emissions. Risk Analysis 24:5, 1121-1141.

Spash C., Vatn A. 2006. Transferring environmental value estimates: issues and alternatives. Ecological Economics 60:2, 379-388..

Vestforbrændingen. 2000, 2005. Grønt regnskab [in Danish]. Glostrup.

Wang H, Mullahy J. 2006. Willingness to pay for reducing fatal risk by improving air quality: A contingent valuation study in Chongqing China. Science of the Total Environment 367:1, 50-57.