

EVA – a non-linear Eulerian approach for assessment of health-cost externalities of air pollution

by

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Abstract

Integrated models which are used to account for the external costs of air pollution have to a considerable extent ignored the non-linear dynamics of atmospheric science. In order to bridge the gap between economic analysis and environmental modelling an integrated model EVA, based on a Eulerian atmospheric module for regional transport and chemical transformation of air pollutants, has been developed. The EVA model follows the impact-pathway approach of the ExternE-project, but provides damage estimates which are more consistent with the laws of physics and chemistry.

In this paper the significance for the final external cost estimates of the Eulerian approach is explored. Uncertainties in the health costs estimates are endemic in particular for mortality, but in order to achieve a common baseline the approach recommended by the OECD has been employed for the valuation part. This approach implies the use of life-years lost as the basis for the valuation of chronic mortality.

The comparison shows that external cost estimates from the approach normally used as a basis for cost-benefit analysis do not provide consistent figures as they fail adequately to capture the non-linear source-receptor relations of the emissions. External cost estimates based on the Eulerian approach, on the other hand, are in mutual conformity. The existence of non-linear dynamics and possible thresholds, both in the atmospheric modelling and in the dose-response functions for health effects, need further attention and should not be neglected when interpreting estimates for external effects.

EVA is an abbreviation of Economic Valuation of Air Pollution.

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 more ecologically informed approach to accounting for external effects in environmental economics. 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. Many economists have been uneasy with the hypothetical answers obtained in response to such hypothetical questions (Diamond and Hausman, 1994). The description of the environmental goods in the questionnaires applied has become subject to substantial criticism, and it has been argued that preferences do not exist a priori, but rather are formed in discursive processes (Sagoff, 1998). The focus here will be on a related issue of concern, so far subject to less attention in the literature, namely approaches employed for providing descriptions and quantifications of the environmental consequences in stated preference surveys. If preferences, rather, are formed in discursive processes, it would seem important to be able to uncover preferences for non-market goods on the basis of descriptions of environmental consequences that include non-linearity and threshold effects.

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 have been developed in the context of the European ExternE project, where valuation has been linked with specific health effect endpoints, i.e. bronchitis or asthma attacks, and the exact number of such health-effect endpoints in relation to quantified changes in air quality as captured by means of simple atmospheric models (Krewitt, 2002; European Commission, 2002). The latter approach is known as the impact pathway approach for accounting for external costs, 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, and perhaps more importantly, it allows subsequently for a dynamic process of aggregation on the basis of integrated atmospheric-economic modelling.

While the impact pathway approach by now is fairly established in research on externalities from air pollution and in fact has caused a new direction in valuation research to develop, the specific air pollution modelling employed in economic analysis for detailing the environmental consequences of changes in polluting emissions has been subject to less attention and does in fact not compare favourably to state-of-the-art in environmental modelling. The air pollution modelling in ExternE’s Ecosense-model assumes that air mass trajectories are linear and does not account fully for the complex chemical interactions that take place as air pollution is transported and mixes with other pollutants present in the background concentrations. Meteorology is non-linear and there are thresholds in ozone formation which need to be accounted for properly before reliable estimates for external effects can be reached. In this article, we explore the implications of applying full-scale Eulerian air pollution models for the regional transport and atmospheric chemistry when accounting for the external costs of air pollution. This is carried out by analysing and contrasting the methods and results of the ExternE Ecosense model with results from a new model, EVA (Economic Valuation of Air pollution), based on the Danish Eulerian Hemispheric Model (DEHM) approach to atmospheric modelling. Before presenting the technical results of the analysis the paper provides some observations on externality assessments.

2. Recent advances in externality assessments.

Pigou (1920), 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).

While the assessments made 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). 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) as well as to the Danish review (Andersen et. al., 2004), while more in-depth treatments of the particular issues are referenced below.

Dose-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 cohorte (Pope et. al., 1995). The study, which was based on a comprehensive cohorte of about 500 000 individuals, who were followed for in the region of 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 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 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 microgramme 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 function down to 1/3 of the published figure (e.g. European Commission, 1999; Friedrich and Bickel, 2001; Holland and Watkiss, 2002). The Krewski reanalysis (2001), 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 without adjustments (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 various environmental economists, 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, SO₂ and PM, respectively. However, one needs to understand the atmospheric chemistry at play and the way in which SO₂ and NO_x translate into health effect endpoints. Because both SO₂ and NO_x form secondary particulates after transport and chemical transformation, it is really the effect of the two types of particulates, sulphates (SO₄²⁻) and nitrates (NO₃⁻) that is at play rather than the primary emissions of SO₂ and NO_x per se. There are relatively few individual effects for the primary emissions of SO₂ 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 nevertheless more appropriate that medical expertise is involved 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 has remained 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 Pearce approach 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. That

ambient concentrations of particulate matter affect public health has become established, and there appears to be consensus that the most authoritative estimate of dose-response functions for the purpose of externality assessments is provided by the research of Pope et al. (1995, 2002, cf. also Krewski et al., 2001). The range of morbidity effects involved have been selected to avoid double-counting, and although a better statistical basis for individual effects is still desirable, the aggregated externality figures are not particularly sensitive to changes in these. 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 these 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 microgramme/m ³ /year | Valuation Euros (2004-prices) |
|--------------------------------|--|-------------------------------------|
| MORBIDITY_PM | | |
| Chronic bronchitis | 8,2E-5 (adults) | 50.360 per case |
| Restricted activity days | 8,4E-4 ÷ hosp. adm. (adults) | 116 per day |
| Hospital admissions | | |
| - respiratory | 3,46E-6 | 7.409 per case |
| - cerebrovascular | 8,42E-6 | 9.387 per case |
| - congestive heart failure | 3,09E-5(>65years) | 15.450 per case |
| - lung cancer | 1,26E-5 (adults) | 20.150 per case |
| Asthma children (7,6%<15years) | | |
| - bronchodilator use | 1,29E-1 | 20 per case |
| - cough | 4,46E-1 | 54 per case |
| - lower resp. symptoms | 1,72E-1 | 14 per case |
| Asthma adults (5,9%>15years) | | |
| - bronchodilator use | 2,72E-1 | 20 per case |
| - cough | 2,8E-1 | 54 per case |
| - lower resp. symptoms | 1,01E-1 | 14 per case |
| IQ lead (Pb) (<1 year) | | |
| mercury (Hg) (fosters) | 1,3 0,33 | 23.715 per point |
| MORTALITY | | |
| Acute mortality_SO2 | 7,85E-6 | 1.941.134 per case |
| Chronic mortality_PM | 1,138E-3 (>30 years) | 194.957 per yoll |
| Infant mortality_PM | 4,68E-5 (<9 months) | 2.911.700 per case |
| Acute mortality_O3 | 3,27E-6*SOMO35 ¹ | 1.941.134 per case |

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 Holland et. al. (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)

¹ SOMO35 denotes the number of days where max. 8-hour mean of 35 ppb/m³ is exceeded

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 and economic valuation (European Commission, 1999). The modelling of emissions transport and atmospheric 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 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 arrived at are highly site-specific. The Ecosense model has been made operational in a desk-top programme 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 SO₂, NO_x and particulate matter as a result of point source emissions. There is no atmospheric chemistry included in the model formulation, so in particular NO_x - concentrations are not likely to be adequately captured, especially in the periphery of the source, in this type of model.

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. The model assumes a constant average windspeed (of 7.5 metres per second) and the trajectories of emission transport are assumed to run along straight lines. Despite their regional coverage they are weighted according to local meteorology. The model extends to more than 2 500 km from the receptor points. By applying an average wind speed from the receptor points' local meteorology, as a characterisation of regional transport of air pollutants, a simplification on a substantial scale has been introduced. As seen from inspection of Figure 1, trajectories of air pollutants do not in reality run along straight lines but are nested as a result of the meteorological forces. The laws of physics 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 laws of nature known as the Ekman spiral, that may significantly impact the real dispersion of air pollutants.

Also the atmospheric chemistry in the regional WTM-model is relatively simplified. 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 - concentrations via the photochemistry has not been accounted for. As a result, the resulting concentrations can be either overestimated or underestimated.

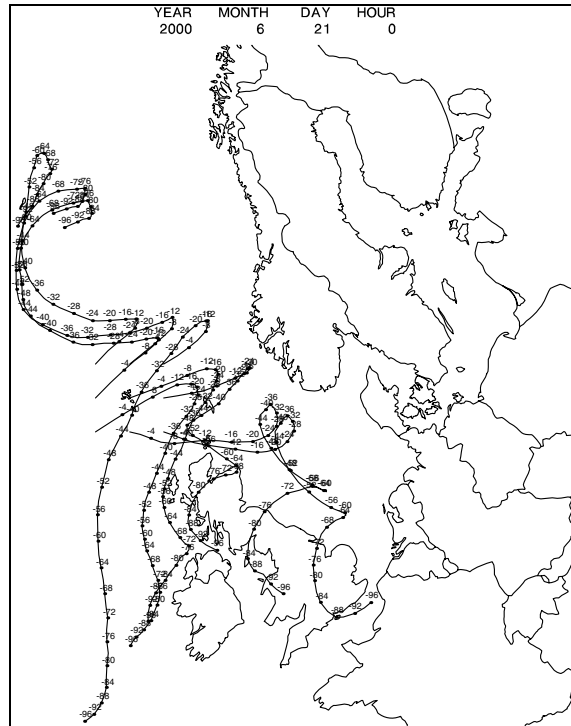


Figure 1 Trajectories on basis of meteorology forecasting model (Ambelas Skjøth, 2002).

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 instability in 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 modelling that takes into account hour-to-hour changes in wind speed and directions requires much more computer capacity and could not be run on a desk-top PC.

It is difficult to check the model output of Ecosense's atmospheric modules against more state-of-the-art air pollution models, as Ecosense in the user interface does not generate separate outputs for the atmospheric calculations but only an integrated monetary output. The absence of maps in the Ecosense software further complicates the identification of grid cells so as to allow for direct comparison with results from other models.

A non-linear Eulerian approach

In order to improve the atmospheric basis for accounting for external costs, non-linear Eulerian air pollution models have been applied in the following. Such models also allow us to investigate the robustness of Ecosense outputs. 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 Airqualitymodel, 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 held constant, so that the monetary damage cost output of Ecosense is compared directly with the monetary output of EVA.

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 mesh in which the chemical composition changes in response to the air flowing in or out are called Eulerian models.

In the model domain of a Eulerian model, which is divided into grid cells in which the spatial and temporal concentration distribution is described for every time interval, 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) and the amount of chemical components which are transformed into other components (determined through a calculation of chemical rates and reactions). Within every grid cell the concentration distribution is assumed to be homogenous.

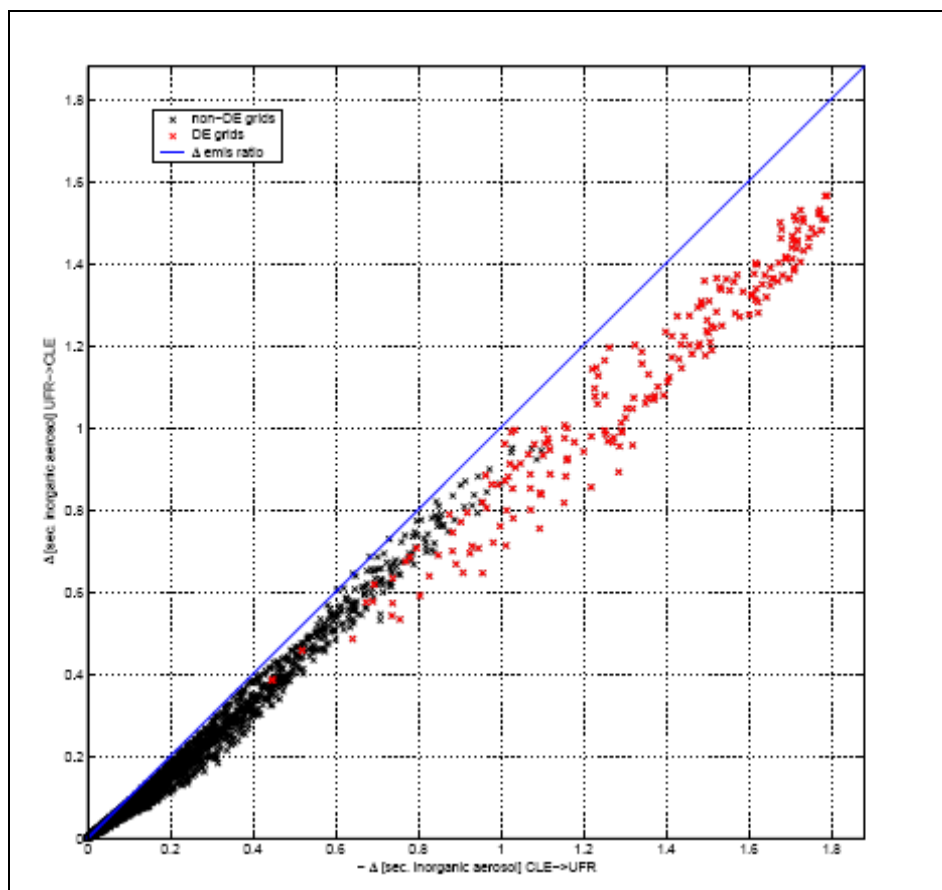


Figure 2: Non-linear source-receptor relations for delta-concentrations in an Eulerian model (Source: Amann et al., 2004)

The results from a Eulerian model present the concentration distributions for the entire period of the model run, and over the entire domain of the model. The emphasis on capturing the 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 account of 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 SO₂ and primary particulates in the main can be assumed to be approximately linear, this is not the case for NO_x, VOC and NH₃, due to the atmospheric chemistry at play. Figure 2 shows non-linear relations of NO_x – reductions to background concentrations as calculated with a Eulerian model.

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 x 50 km around the point source and the regional scale model (DEHM) treats the remaining area. Effectively this means that the results from the local scale model substitutes for nine grid cells in the regional scale modelling. The local scale model decomposes the local area into 1 x 1 km to allow for high-resolution modelling of certain emissions and their transport and dispersion and to avoid difficulties in the periphery of the local scale modelling.

In the following, the significance for the external costs of applying a state-of-the-art atmospheric chemistry transport model, based on the best available knowledge regarding physical and chemical processes in the atmosphere, is explored. This is carried out by comparing the monetary output results from ExternE's Ecosense model 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 recently published estimates in the international scientific literature (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 electricity system. Although renewable energy (primarily wind) contributes 29 percent of the electricity production in Denmark, the remaining share comes from fossil fuels (Energistyrelsen, 2005: 9).

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 yet. As a result, emissions in the modelled scenario amount to 499 tonnes SO₂, 4403 tonnes NO_x and 147 tonnes primary particulates (PM_{2.5}²) per year.

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 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 amount to 587 tonnes SO₂, 1192 tonnes NO_x and 14 tonnes primary particulates (PM_{2.5}) per year.

Fynsværket and Amagerværket are among the eleven large fossil fuel based CHP plants in Denmark. However, among the alternatives to fossil fuels, waste incineration is a significant source of electricity in Denmark, and is second to wind energy, overall. 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 both electricity and heat.

Vestforbrændingen, which is located in a suburb of Copenhagen, emitted 312 tonnes of SO₂, 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, which are analysed in more detail in a separate paper. In the analysis year (2000-emissions) Vestforbrændingen has no de- NO_x unit but an older desulphurisation unit. (However, in 2006 flue gas scrubbers have been installed causing the results to be of less practical relevance for the present circumstances.)

The three units analysed are modelled for historical emissions; below, in Section 6, these are used for an assessment of external costs in more recent years.

² PM in this paper denotes only primary emissions of PM_{2.5} unless otherwise indicated.

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. In Ecosense two calculations are performed with two emission scenarios (with and without the point source) and the annual mean values are then subtracted to give the delta concentrations, but it is not clear what methodology has been applied in the Ecosense software to distinguish increments in delta-concentrations from numerical noise.

For the purpose of the EVA model a tagging approach was developed. In the EVA model only one calculation is performed with the model where an additional concentration field containing the concentration arising from the point source emission (tags) has been included in the model and is advected separately. In this way the delta concentrations can be calculated continuously thereby minimising the numerical noise.

Figure 3 shows the resulting delta concentrations for SO₂-emissions from the CHP Amagerværket near the centre of Copenhagen. As with SO₂, PM_{2.5} also has impacts on delta-concentrations mainly at the local scale, i.e. within a 50 km range of the source.

Figures 4, 5 and 6 show regional scale maps for each of the air pollutants, NO_x, SO₄²⁻ and ozone (O₃); these are pollutants with impacts on delta-concentrations at a much larger regional scale. The maps show the source in question and the resulting delta-concentrations of the pollutants in the Greater Copenhagen area.

As Copenhagen is located on an island where the Baltic Sea and the North Sea meet, a considerable portion of the pollutants affects 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 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 prevailing 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.

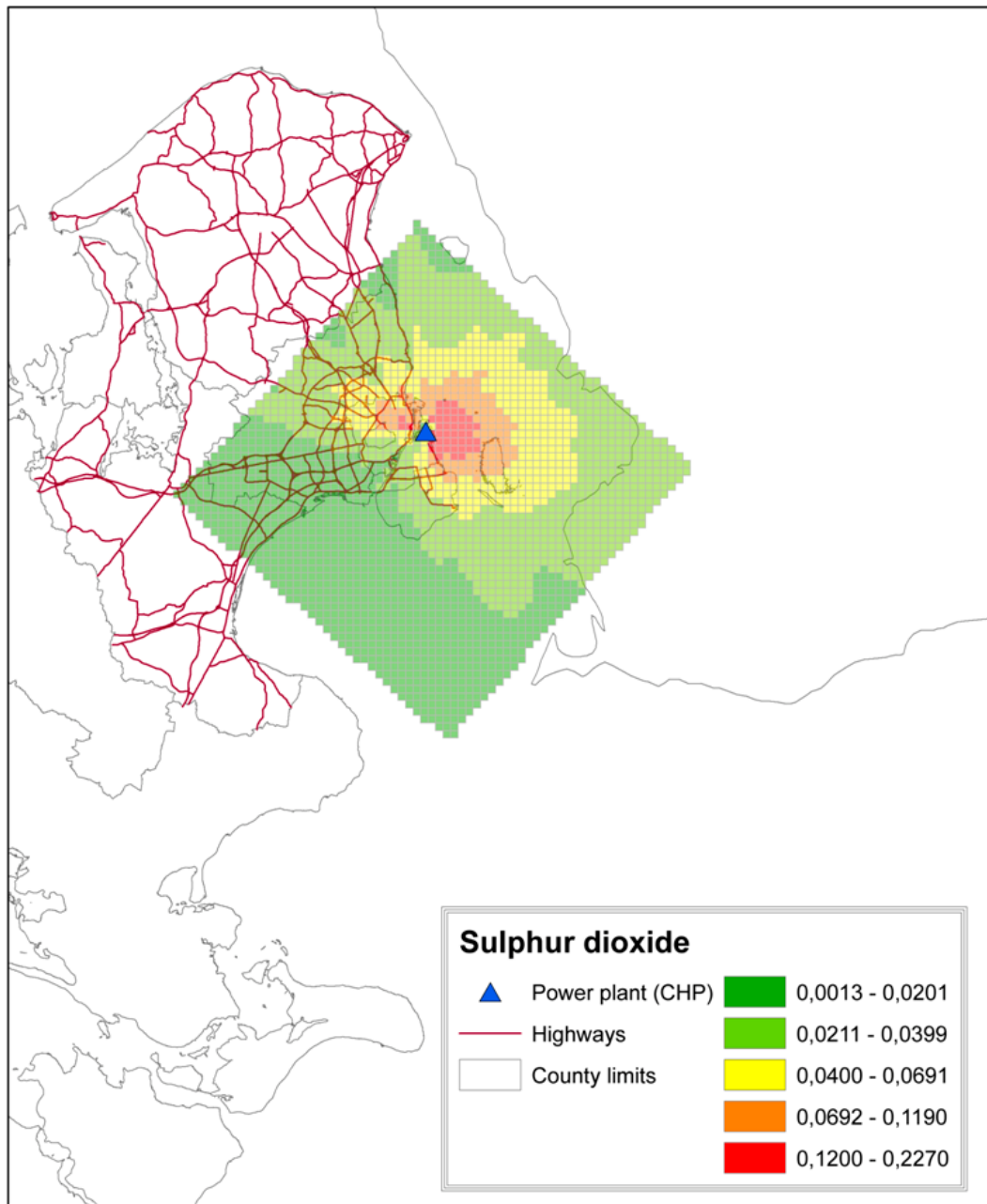


Figure 3: Changes in annual delta-concentrations from SO_2 -emissions of Amagerværket according to the local-scale model of EVA.

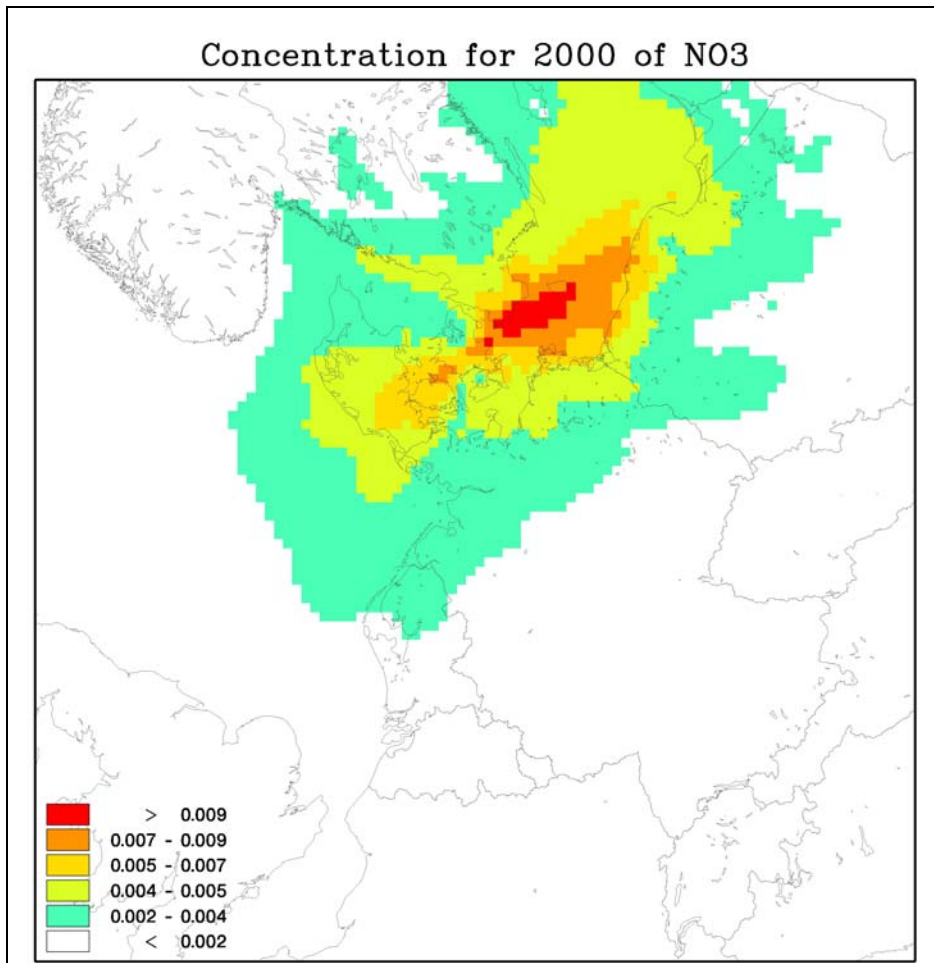


Figure 4: Changes in annual delta-concentrations from NO_x-emissions of Fynsværket according to the regional-scale model of EVA.

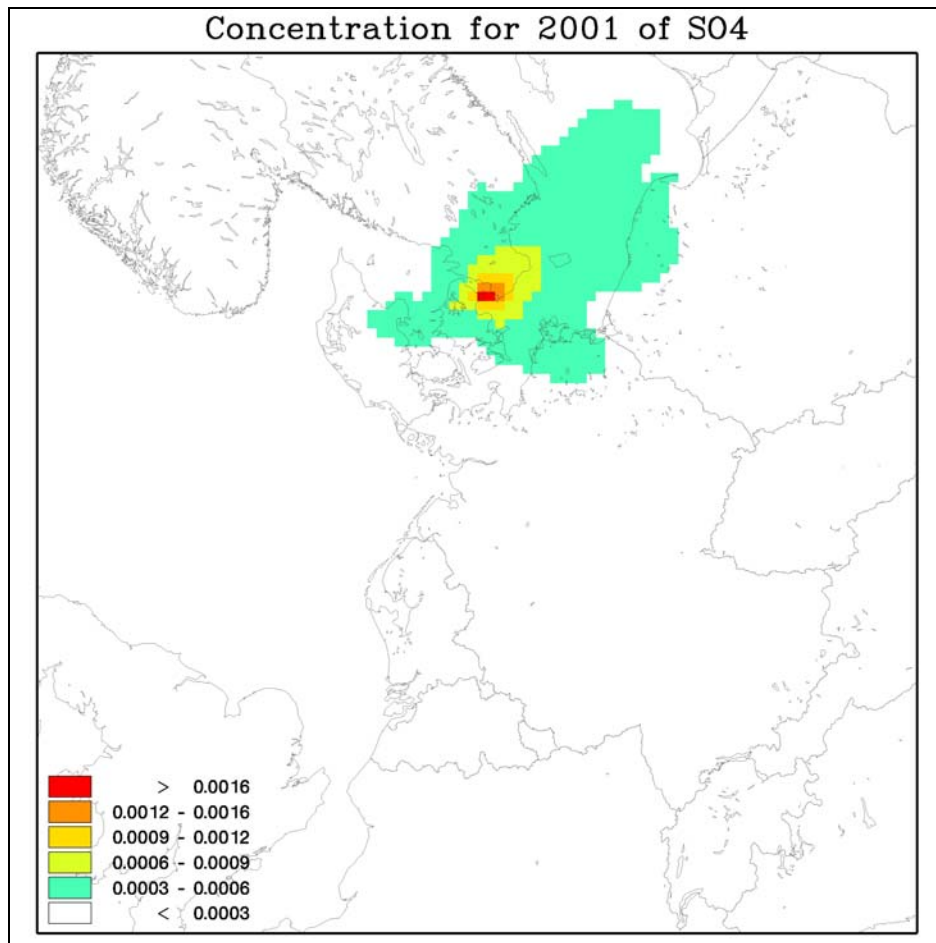


Figure 5: Changes in annual delta-concentrations of SO_4^- of Amagerværket according to the regional-scale model of EVA.

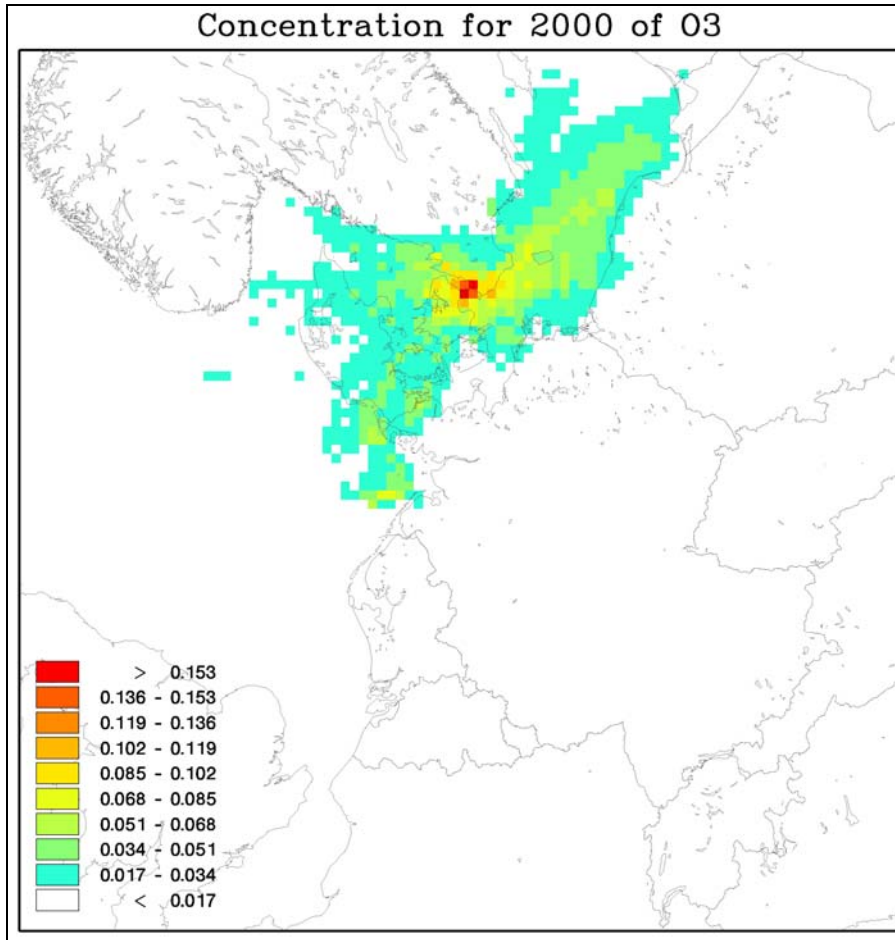


Figure 6: Changes in annual delta-concentrations of O₃ of Amagerværket according to the regional-scale model of EVA.

6. Comparing damage estimates from EVA and Ecosense.

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 both 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 directly on the meteorological data. For both models in Ecosense 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 meteorological conditions 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. Below the results are applied to provide for an assessment of the external costs with the actual emissions for the years 2003-2005 (note that external costs related to greenhouse gases are not included).

Tables 2-4 show the higher resolution of the EVA-model in that results are specified for primary and secondary components of sulphur. In addition EVA includes the net effect of ozone that is not captured by Ecosense. NO_3 and SO_4^- are secondary particulates which tend to be transported over longer distances, while $\text{PM}_{2.5}$ and SO_2 are emissions that produce damage predominantly in the local scale area (within the 50kmx50km area).

Ozone health effects are included in EVA for days where the 8-hour maximum average exceeds 35 ppb/m³. 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 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 widely 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 (Loft et al., 2006). Once NO_2 is properly accounted for, the net contribution of ozone formation and depletion to the damage estimates would undergo adjustment. Ozone has not been accounted for in Ecosense.

| FV | EVA (m_2000) | EVA (m_2001) | EVA (m_2002) | EVA Average | Ecosense |
|----------------------------------|-----------------|-----------------|-----------------|----------------|----------|
| PM _{2,5} -primary | 21,31 | 30,22 | 25,62 | 25,72 | 15,70 |
| SO ₄ /SO ₂ | 5,09 | 4,23 | 4,91 | | |
| SO ₂ /SO ₂ | 0,86 | 1,23 | 0,91 | | |
| SO ₂ -total | 5,95 | 5,46 | 5,82 | 5,74 | 10,79 |
| NO ₃ /NO _x | 15,81 | 19,57 | 18,65 | | |
| O ₃ /NO _x | -0,04 | 0,00 | -0,01 | | |
| NO _x -total | 15,76 | 19,57 | 18,65 | 17,99 | 17,20 |
| 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 | EVA (m_2000) | EVA (m_2001) | EVA (m_2002) | EVA Average | Ecosense |
|----------------------------------|-----------------|-----------------|-----------------|----------------|----------|
| PM _{2,5} -primary | 35,78 | 46,83 | 41,38 | 41,33 | 34,83 |
| SO ₄ /SO ₂ | 9,51 | 10,57 | 10,29 | | |
| SO ₂ /SO ₂ | 2,00 | 2,45 | 2,07 | | |
| SO ₂ -total | 11,51 | 13,02 | 12,36 | 12,30 | 11,84 |
| NO ₃ /NO _x | 15,03 | 18,13 | 17,63 | | |
| O ₃ /NO _x | -0,38 | -0,36 | -0,33 | | |
| NO _x -total | 14,65 | 17,77 | 17,30 | 16,58 | 12,93 |
| 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 | EVA (m_2000) | EVA (m_2001) | EVA (m_2002) | EVA Average | Ecosense |
|----------------------------------|-----------------|-----------------|-----------------|----------------|----------|
| PM _{2,5} -primary | 46,94 | 49,84 | 40,55 | 45,78 | 17,77 |
| SO ₄ /SO ₂ | 9,21 | 10,09 | 9,85 | | |
| SO ₂ /SO ₂ | 3,08 | 2,60 | 1,88 | | |
| SO ₂ -total | 12,29 | 12,69 | 11,73 | 12,24 | 5,42 |
| NO ₃ /NO _x | 14,81 | 18,44 | 17,33 | | |
| O ₃ /NO _x | -0,43 | -0,40 | -0,34 | | |
| NO _x -total | 14,38 | 18,04 | 16,99 | 16,47 | 5,75 |
| 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.

Comparing the three tables it can be noted that damage estimates from the EVA-model are relatively consistent between the three power plants. It is particularly interesting to note that although NO_x emissions are substantially lower in absolute terms at the Amagerværk (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 what we would expect for a regional pollutant as NO_x, when the sources are within short distances of each other,

and transport and atmospheric chemistry are considered at both local and regional scale. There is consistency too for EVA-results for SO₂ and PM_{2.5} for the two units within the urbanised area, while Fynsværket as a unit in a rural area, as expected, records lower SO₂/PM damage costs due to lower population densities in the local area.³

The results from Ecosense, on the other hand, are not internally consistent. The NO₃⁻ damage costs vary from about 5 € for the smallest plant and up to 17 € for the largest emitter, but as NO₃ is a regional pollutant this difference has no intuitive explanation. As to the two pollutants known for mainly local effects, in Ecosense the differences between SO₂ and PM damage costs do not vary with the exposed population densities. Fynsværket, which is located in a rural area, has damage costs for SO₂ and PM roughly similar to or even higher than Vestforbrændingen located in an urban area. Amagerværket which is located in the Copenhagen city centre has the highest costs for PM, while there is hardly any difference for SO₂-damages to the urban plant.

Reasons for these differences were hypothesized in Section 3 and relate to the difference between the linear and simplified approach of Ecosense versus the non-linear meteorology in the Eulerian long-transport modelling of EVA. While the Eulerian model produces consistent damage estimates, Ecosense's linear approach apparently provides for rather non-linear results! It would be desirable to compare regional and local scale damages from the two models in more detail. It can also play a role that population data in Ecosense is at NUTS2-level, while EVA takes advantage of the Danish CPR-registry to disaggregate population data to 1x1 km. gridcells in the local scale. Care has been taken to run Ecosense properly and in accordance with the instructions.

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.⁴ Ecosense does not include heavy metals.

6. External costs per kWh.

In order to explore the external costs relative to the energy output of the three CHP plants, Table 4 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; 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.

³ The damage costs are in both models sensitive to the VOLY value; if the NewExt median-VOLY (cf. Alberini et. al., 2006) was applied the per kilogramme damage cost for NO_x would in EVA be about 7 €/kg. A similar relationship between use of average and median values would be found for the other two pollutants in both urban and rural locations.

⁴ 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.

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) | | |
|-------------------|-------------------------------|------|-----|------------------------------------|------|-----|--|------|-------|
| | FV | AV | VF | FV | AV | VF | FV | AV | VF |
| CHP | FV | AV | VF | FV | AV | VF | FV | AV | VF |
| PM _{2,5} | 115 | 26 | 2 | 3,0 | 1,1 | 0,1 | 0,15 | 0,05 | 0,08 |
| SO ₂ | 917 | 2028 | 36 | 5,3 | 25,0 | 0,4 | 0,27 | 1,24 | 0,38 |
| NO _x | 4717 | 2515 | 529 | 84,9 | 42,0 | 8,7 | 4,28 | 2,08 | 7,51 |
| | | | | 93,1 | 68,1 | 9,3 | 4,70 | 3,37 | 7,98 |
| 2004 | Emissions in 2004 (tonnes) | | | External costs 2004 (million €) | | | External costs 2004 (eurocents/kWh) | | |
| | FV | AV | VF | FV | AV | VF | FV | AV | VF |
| CHP | FV | AV | VF | FV | AV | VF | FV | AV | VF |
| PM _{2,5} | 21 | 17 | 3 | 0,5 | 0,7 | 0,1 | 0,03 | 0,04 | 0,16 |
| SO ₂ | 971 | 750 | 19 | 5,6 | 9,2 | 0,2 | 0,28 | 0,57 | 0,29 |
| NO _x | 5378 | 1347 | 294 | 96,8 | 22,4 | 4,8 | 4,85 | 1,38 | 6,00 |
| | | | | 102,9 | 32,4 | 5,2 | 5,15 | 1,99 | 6,45 |
| 2005 | Emissions in 2005 (tonnes) | | | External costs 2005 (million €) | | | External costs 2005 (eurocents/kWh) | | |
| | FV | AV | VF | FV | AV | VF | FV | AV | VF |
| CHP | FV | AV | VF | FV | AV | VF | FV | AV | VF |
| PM _{2,5} | 20 | 18 | 3 | 0,5 | 0,8 | 0,1 | 0,03 | 0,06 | 0,18 |
| SO ₂ | 669 | 89 | 21 | 3,8 | 1,1 | 0,3 | 0,21 | 0,08 | 0,31 |
| NO _x | 4541 | 448 | 519 | 81,7 | 7,5 | 8,5 | 4,52 | 0,53 | 10,28 |
| | | | | 86,1 | 9,4 | 9,0 | 4,76 | 0,66 | 10,77 |

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, while the other two are without such de- NO_x equipment. It is notable that NO_x -emissions are more than a factor 10 lower at Amagerværket. The highest external costs are found at the waste incinerator, Vestforbrændingen, with more than

10 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. This is the model used in Danish taxation legislation to split the energy taxation burden between heat and electricity production. In Denmark 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 accords the greater share of the emissions to electricity. Table 6 provides for the three CHP plants an overview of external costs per kWh and per heat unit (one kWh is 3,6MJ) according to the 200 percent model.

| 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 | 3,44 | 3,76 | 3,44 | 2,02 | 1,13 | 0,42 | 4,41 | 3,49 | 6,34 |
| Heat | 1,26 | 1,39 | 1,32 | 1,35 | 0,86 | 0,24 | 3,58 | 2,96 | 4,45 |
| Sum | 4,70 | 5,15 | 4,76 | 3,37 | 1,99 | 0,66 | 7,98 | 6,45 | 10,77 |

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

External costs related to electricity generation are of interest when considering the relative advantages of renewable energy for reasons other than climate benefits. When splitting the external effects on heat and power, the figures in Table 6 indicate that external costs for electricity range from about 0,5 eurocents/kWh and up to about 6 eurocents/kWh depending on fuels, flue gas cleaning and location of the power plant in relation to populated areas.

- the highest external costs (about 6 eurocents/kWh) are present in the case of a municipal waste incinerator located in a suburban area with an older desulphurisation unit only. Here power and heat generation is a side-effect of the incineration of waste so the energy efficiency does not match that of dedicated CHP units. There is relatively high exposure and no immediate advantage of the prevailing western winds as the emissions are carried over large housing areas. The waste has to be disposed of and so incineration with energy recovery is an option, but despite CO₂-neutrality there are substantial external costs. The flue gas treatment installed in 2006 will alleviate some of these.

- 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 prevailing western winds may go some of the way to explain this result; however, for 2005 there is an anomaly in that the reserve unit was not in operation. The findings indicate that it appears feasible for fossil fuel based plants effectively to internalise the health costs and to remain competitive; the 2004-figure of 1.13 eurocent/kWh is more representative for operation with the phasing in and out of reserve units, however, than the 0.42 eurocents/kWh for continuous operation only.

- for the third unit, Fynsværket, the external costs of about 3.5 eurocents/kWh are fairly stable over the years, as no major changes in flue gas cleaning has been introduced. 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. If Fynsværket installed a de- NO_x unit its external costs could be reduced to less than half. 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 EVA model allows for detailed calculation of the external costs of each individual source, which is preferable if one wants to take account of the characteristics related to the site, fuels and flue-gas treatment.

The figures indicate that the advantage of renewable energies without air pollutants under the present circumstances is in the range from 0.5-6 eurocents/kWh depending on 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 (however, an initial attempt was presented 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 are statistical *predictions* of damages, based on current knowledge regarding the relationships between exposure and health effects and the related costs.

It has been 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 externalities is, however, that the methodology is based on a carefully modelled average of the meteorological conditions; taking into account the

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 adequately in the predictions of the health damages related to air pollution has hence been carried out. The resulting argument is that a move is made away from the weather forecast approach of Ecosense to a better and more robust framework for assessing the external effects relating to the individual emissions.

That EVA is a more robust system is supported by the findings above, as the damage costs for the individual pollutants, when aggregated per kilogramme, are within a common range. For NO_x the average damage costs is similar across urban and rural sources, as would be expected for a regional pollutant, whereas for PM and SO_2 the damage costs appear to be in conformity for rural and urban sources respectively, as would be expected for local scale pollutants.

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