The Social Costs of Electricity: Do the Numbers Add Up?

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Abstract

Several recent studies have mounted major efforts to estimate the social cost of electricity generation. This paper provides an overview of this literature and a focused qualitative and quantitative comparison of the most comprehensive and rigorous of these studies. The paper also provides a synthesis that can help reduce the cost of future applications of these methods.

Key Words: electricity, environment, health, social costs, adders, externalities

JEL Classification No(s).: H23, L94, L98

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Research activity to estimate damage functions associated with electricity fuel cycles has skyrocketed in recent years, spurred on by public utility commissions in the U.S. interested in accounting quantitatively for the external costs of new investments in electricity generation and interest in emissions fees or "green" fees in Europe. At least six major studies have been mounted, and as these studies now have been completed, it is an appropriate time to evaluate what we have learned by comparing their approaches and results.

This paper is aimed at two overarching questions: (i) Are the damage estimates credible in the context for which they are offered, and hence reliable data for policy making?[2] (ii) Are the approaches and estimates transferable to other locations and contexts? Looking across the studies, the most important finding is that damages from air pollution to health and other endpoints associated with fossil fuel cycles featuring *new* generation plants are consistently found to be small relative to generation costs, and relative to values in use by some state utility commissions.[3] The relative consistency provides a measure of credibility for the numerical estimates. However, the specific estimates of fuel cycle damages, in terms of mills/kWh (or dollars per ton of emissions) are subject to much "model uncertainty" stemming from atmospheric modeling. For this reason, measures of damage that avoid this uncertainty, such as dollars/person/unit change in pollution concentration, are much preferred for use in any benefit transfer exercise -- the implication being that at least for pathways involving particulate and ozone concentrations, future studies in other geographic areas cannot avoid performing

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[1] Senior Fellow and Fellow at Resources for the Future. The authors would like to thank Kerry Smith, the presenters and attendees at the Social Cost of Electricity session of the Southern Economics Association, 1995, and an anonymous reviewer.

[2] We define "damages" as the monetary value of all possible externalities and do not undertake in this paper an analysis of the extent to which such damages are Pareto-relevant externalities, i.e., damages that have not been taken into account (through regulation, property rights, or other means) in decisions about processes and other activities leading to the pollution. See Lee et al. (1995) for a detailed discussion.

[3] U.S. EIA (1995), Table 18.

location-specific air quality modeling. We also conclude that effects of fuel cycle choice on employment, government revenues, and global warming can easily swamp conventional environmental damage differentials across fuel cycles, making the comparison across cycles problematic until issues regarding these endpoints are resolved.

In the next section we provide a framework for evaluating the studies. Next we briefly describe the studies, present their aggregate results, and compare estimation approaches for the coal fuel cycle. We attempt a detailed reconciliation of the dominant pathways in the coal fuel cycle -- the air-health pathways -- in the three studies that are most comparable and for which we can extract the necessary information from their documentation. These are Lee et al. (1995), Hagler Bailly (1995), and European Commission (1995). Then, we illustrate the effect on the Lee et al. estimates of adding less credible global warming "damage" and nonenvironmental damage estimates to arrive at a comprehensive, though speculative, assessment of social damages from a generation plant at one location. We close with a summary of findings addressing the two overarching questions posed above.

I. FRAMEWORK FOR EVALUATING THE STUDIES

In this section we lay out a mathematical representation of the factors involved in estimating damages from environmental damage "pathways" (which link emissions or concentrations to values for specific damage "endpoints"). A similar framework could be developed for addressing nonenvironmental damages, such as road damages and occupational damages, for instance. For the moment, we ignore the issue of whether a monetary "damage" is a Pareto-relevant externality, leaving this topic for our second paper in this issue ("Second-Best Estimates of Social Costs of Electricity").

Let D_j be the annual damage from a source of a given type (say a steam boiler fired by coal) in location j. To simplify the expression, we assume there is only one emissions type (e.g., nitrogen dioxide), that ambient pollutants (e.g., ozone, nitrogen dioxide, and particulate concentrations) do not have cumulative effects on the environment, damages are additive across endpoints (e) and locations (k) and there is only one time period (say, annual). Then, the damages can be expressed as:

$$D_j = \sum_{z} \sum_{e} \neq \sum_{k \neq j} D_{z,e,k} \tag{1}$$

with damage at j internalized by the source.

The choice of the spatial boundaries (embedded in k) is a critical factor in the damage calculation, as it ultimately determines the size of the population and other "targets" potentially affected. If available pollution dispersion models are thereby stretched beyond their limits by a distant boundary, the credibility of the estimates may be affected.

Equation (1) can be broken down into several components. The first is the change in concentrations of ambient pollutant z at receptor k:

$$\Delta C_{zk} = E(Q, A) * C\left(M, C_{zk}^{b}, S\right) \tag{2}$$

Emissions per unit time (E) depend on the abatement measures installed (A) and their removal efficiencies as well as output in that time period (Q), which can be estimated from knowing the capacity of the unit (e.g., 400 megawatts (MW)) and its capacity utilization factor. For the electricity sector, the characteristics of the fuel being burned would also be important. Plant lifetime also matters in the annualization of any damages that are delayed or cumulative.

For the dominant air pollution pathways, the translation of emissions into concentrations is performed with air quality models. These models vary enormously in their treatment of space, time, meteorology, and air chemistry. Some of the air models are designed only to track the movement of primary pollutants, be they particles or gases. The more complex models feature chemical reactions on these primary pollutants, which are affected by meteorological conditions M and background levels of pollutants C^b as they move in the atmosphere, which create secondary pollutants (such as ozone, sulfates, and nitrates). Concentrations of pollutants in the air are also affected by "stack parameters" (S), such as stack height, stack diameter, and the velocity and temperature of the stack gases and particles. Taller stacks with hotter gases and higher exit velocities will result in greater dispersion.

An oft-used simplifying assumption is that changes in ambient concentrations of pollutants are proportional to changes in emissions. This assumption is considered reasonable for "conservative" types of emissions, such as heavy metals and particulates, as well as emitted gases, such as nitrogen dioxide (NO2) and sulfur dioxide (SO2). It is not considered reasonable for transformations of NO2 and volatile organic compounds (VOCs) into ozone and for transformations of NO2 and SO2 into their ambient particulate species (e.g., ammonia nitrates and sulfates). With this proportionality assumption, the transformation of emissions into concentrations can be portrayed through multiplication of emissions by a source-receptor matrix C_{zk}, specified by concentration type (the primary pollutant and secondary pollutants created in the atmosphere), receptor location and time period. The product is concentrations of ambient pollutants at various times and locations attributable to the source of the emissions. If damage functions are linear in concentrations, the total damage will be constant irrespective of spatial distribution of pollution within the domain. In fact, Rowe (1995a) shows that stack height has a relatively minor effect on damages, even though some degree of nonlinearities are built into their damage model, EXMOD.

If concentrations are not proportional to emissions, however, then some sort of air quality model would need to be used to predict concentration changes as a function of emissions from the new facility the variables noted above. As a consequence, E would enter into the concentration function and (2) would be modified to:

$$\Delta C_{zk} = C\left(M, C_{zk}^{b}, S, E(Q, A)\right) \tag{2a}$$

The change in concentrations enters concentration-response functions R specified for concentrations z and endpoints e. Most such functions are for unit responses, e.g., the response per 100,000 people, the probability of a person being affected, the yield response per acre, etc. The marginal response to the pollutants may depend on initial or background conditions of the endpoint being affected (such as health status) R^b, background concentration of the pollutants, and even the change in concentration.

$$R_{zek} = R\left(R_{ek}^{b}, C_{zk}^{b}, \Delta C_{zk}\right) \tag{3}$$

Generally, such "unit" responses can simply be multiplied by the target population for the endpoint (T_{ek}), e.g., people, for an estimate of the physical impacts. That is:

$$I_{zek} = R_{zek} * T_{ek}$$

These effects are valued through a valuation function which may depend on initial conditions of the endpoint (such as health status, baseline visibility), the degree of impacts, characteristics of the population whose values are being expressed (*P*), and characteristics of the target (people, recreation areas, visual range) being affected.

$$D_{zek} = V\left(R^{b}_{ek}, I_{zek}, P, T\right) \tag{4}$$

Note that while the impact variable enters for impacts at the receptor k, the valuation function is defined over the general population. Hence, the population characteristics variable enters without subscripts to indicate that people outside of the receptor k may hold values for improvements at receptor k. Recreational values could be included here, but this specification also is meant to convey nonuse values, i.e., values people who never will visit a place have for avoiding damages there. The characteristics of the target being affected are also entered without subscripts to connote that characteristics outside the area in question (k) may affect values, through substitution or complementary relationships (again, recreation is the canonical example).

By substituting back through equations (1)-(4) (using (2) rather than (2a)), a general representation of the damage associated with a new source at location j is:

$$D_j = \sum_z \sum_e \sum_{k \neq j} V\left(R\left(R^b_{ek}, C^b_{zk}, E(Q,A) * C\left(M, C^b_{zk}, S\right)\right) * T_{ek}, R^b_{ek}, P, T\right) \tag{5}$$

Thus, D depends on overall modeling choices (such as the number of endpoint at issue, the number of receptor areas), output and abatement technologies, stack parameters, meteorology (or other natural processes involved in dispersion and transformation of emissions in the environment), initial pollution concentration levels and initial condition of the target, the size of the target "population," impacts elsewhere, and individual attributes elsewhere. In addition, the particular ambient models, concentration-response functions, and valuation functions would affect how the above variables are interrelated in the calculation of D.

Equation 5 has been written for maximum generality. In fact, the studies examined below use a much more simplified paradigm, such as:

$$\begin{aligned} D_j &= \sum_z \sum_e \sum_{k \neq j} V\left(I_{zek}, R^b_{ek}, P, T\right) \\ &= \sum_z \sum_e \sum_{k \neq j} V_e * I_{zek} \\ &= \sum_z \sum_e \sum_{k \neq j} V_e * r_{ze} * \Delta C_{zk} * T_{ek} \end{aligned} \tag{6}$$

In this paradigm, the concentration-response and valuation functions are linearized, and nonuse values and substitute sites are ignored ($P = T_k$). The term v_e represents a unit value for endpoint e and r_{ze} represents a unit response coefficient associated with a unit change in concentration z.

This framework spans the approaches used in the studies we analyze below. The modeling of individual pathways can be characterized by a level of detail along the spectrum from equations 5 and 6. The weakest link in the sequence of functions in a pathway sets the standard for the level of detail in the analysis. Consequently there may be little benefit from a high degree of resolution at one stage that is only lost in the final analysis because of less resolution at another stage.

Alternative formulations to the one we describe involve dividing D by Q to express damages in mills/kWh, or dividing D by E for a damage per ton emissions measure. We argue below against the transferability of damages expressed in either formulation. Nonetheless, transferring benefit estimates with this type of approach to a regional or national scale characterizes one group of previous studies, which we call "top-down" efforts because they estimate damages on the basis of average values absent site specific resolution. These early efforts include Hohmeyer (1988), which associated environmental toxicity and resulting damages with sources of pollution throughout the economy. Other efforts include Hall (1990) and Viscusi, et al. (1992).

The focus of this paper is a second group of studies we call "bottom-up" because they provide resolution along a number of dimensions specific to the physical site where emissions and other changes affecting the environment originate, and the mapping to other sites where damage occurs. In the next section we briefly review the key studies in this tradition and then compare in detail the three studies that have been the most comprehensive efforts.

II. DESCRIPTIONS OF KEY STUDIES

The first multi-disciplinary, bottom-up attempt at collating and analyzing the vast scientific and economic literature that was applicable to all of the emissions from electric utilities, irrespective of media was the Pace University-led study, *The Environmental Costs Of Electricity* (1990). This study and its less comprehensive predecessors, including ECO Northwest (1987), had some important limitations. While acknowledging the important role played by location in determining the magnitude of damage, these earlier studies could not piece together literature that would permit a consistent set of location-specific damages to be estimated. In addition, they ignored the fact that some residual damages have been internalized, relied at times on studies that estimated damages on a per ton basis and did not consider the entire fuel cycle. The Pearce et al. (1992) study improved upon Pace by taking a full fuel cycle approach but it was not site-specific.[4]

The subsequent generation of studies were designed by multidisciplinary teams employing sophisticated models for estimating air quality, epidemiological, and economic effects of emissions not only from electricity generation itself but also (for some of the studies) from all other stages of the electricity fuel cycle, including extraction, refining, transportation of fuel, construction of the generation plant, operation, and decommissioning. These studies are integrated assessments, rather than efforts to mount original research, although each study makes some original contributions. The choice of framework in these studies is driven by the tradeoff between degree of precision on the one hand and the cost and complexity of analysis on the other. The increased precision can be along spatial and temporal dimensions, as well as

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[4] Other studies of this type include Rae, et al. (1991) and Bonneville Power Administration (1991). A number of other studies have used estimates of the cost of pollution control as a proxy for the value of environmental damages, rather than estimating damages directly. These studies include South Coast Air Quality Management District (1991), New York State Energy Office (1989), and Bernow and Marron (1990). This paper is focused only on studies that employ a damage function approach; thus, we do not discuss these "cost of control" studies.

on recognizing the nonlinearities that may be present in air pollution, health and environmental and valuation processes.

The studies are based on the damage function approach and are fully committed to estimating site-specific marginal damages by population grid or "small" jurisdiction. Most have averaged over temporal measures, e.g., using average annual concentrations rather than a distribution of hourly concentrations, because, say the authors, emissions, concentration-response functions, and valuation functions do not provide evidence of a high degree of variability over alternative time periods. Most health functions are linearized reflecting their lack of strong non-linearities (the issue of thresholds aside). And, for air quality, models are generally used to embody the nonlinearities (some severe) that plague estimation of atmospheric concentrations resulting from the chemical transformation of emissions in the atmosphere. The studies take a "marginal" approach that estimates impacts and damages associated with the incremental addition of a single power plant (or multiple new plants). Some deal directly with the issue of whether damages would be internalized. Three of these that deserve special mention include:

- **Regional Economic Research (RER) (1991)**
  This study for the California Energy Commission focused on the damages associated with airborne pollutants. Technologies were characterized based on existing plants serving California. Air quality modeling was done for separate air quality basins. Thus, damages were calculated on a project- and site-specific basis. The ozone modeling involved a relatively simple sampling of grid cells in each basin. Analysis was limited to generation activities and the full fuel cycle was not analyzed.

- **National Economic Research Associates (NERA) (1993)**
  This study was conducted for the Nevada Power Company and estimated the damages from electric utility resource selection in Nevada. NERA focused on the

air pollution pathways from new sources in Las Vegas Valley and other areas in Southern Nevada for a 1990 baseline and projected to years 2000 and 2010.

- **Triangle Economic Research (TER) (1995)**
  Conducted for Northern States Power in Minnesota, this study developed damage per ton estimates for bringing on line to the Northern Power System a combination of a new coal plant and several natural gas combined cycle plants in 2006. By focusing only on generation the study ignored the methodological and empirical issues associated with upstream activities. As a further consequence, its damage estimates would be too low. The TER study focused exclusively on the air emissions pathways, arguably the most important set; in particular, health, visibility, materials, and crops. It modeled damages at the smallest spatial and temporal level of the three studies -- using data at the zip code level and estimating damages hourly for the year. Monte Carlo simulation techniques were used to express uncertainty.

The rapid advancement in this literature has culminated in the following three important studies that are the primary focus of our attention. These studies are distinguished by the magnitude of effort and the comprehensiveness of analysis. They were full fuel cycle analyses, and were subject to extensive peer review and subsequent modification.

- **Oak Ridge National Laboratories / Resources for the Future (Lee et al.) (1995).** This study was conducted for the US Department of Energy and involved a research team from Resources for the Future and Oak Ridge National Laboratories and was designed in consultation and generally in parallel with a team put together by the European Commission. In spite of a multiplicity of goals, this study was designed primarily to investigate and develop methods for estimating full fuel cycle external costs appropriate to the contemporary PUC social costing context, i.e. new (circa 1990) generation investment. All stages and all pollutants were considered initially, although many "pathways" were eliminated in a complex screening procedure. Estimates of damages were developed for two "reference" environments (Oak Ridge, TN, near Knoxville; and northern New Mexico except where other sites were appropriate for hydro and biomass) for six generation technologies (coal, oil, gas, hydro, nuclear, biomass). The sites and plant designs were not meant to be generic or representative. Considerable effort was devoted to estimating nonenvironmental externalities and to discussions of the extent to which various types of damages are externalities. Monte Carlo simulation techniques were used to express uncertainties.

- **European Commission (EC) (1995).** This study was planned in collaboration with the Lee et al. study and conducted by Directorate-General XII of the European Commission. The study was designed to develop methods for estimating full fuel cycle costs in the European context. As with Lee et al, the entire fuel cycle was considered. For most fuel cycles two reference environments were studied (West Burton in the UK and Lauffen, in Baden-Wurttemberg, in Germany, except where other sites were appropriate) and nine fuel cycles were studied (coal, lignite, biomass, nuclear, oil, natural gas, photovoltaic, hydro and wind). A limited effort was devoted to estimating nonenvironmental externalities. Uncertainty was described qualitatively and expert judgment was employed, but no formal simulation techniques were applied.

- **Hagler Bailly with the Tellus Institute (HB) (1995).** This study was conducted under a joint industry and governmental effort led by the Empire State Electric Energy Research Corporation and the New York State Energy Research and Development Authority. The emphasis of this study was on building a computer model capable of estimating damages to New York and surrounding states from new and repowered generation plants located anywhere in New York (EXMOD). The scope of the HB project was similar to that of

Lee et al., with the exception that less emphasis was placed on nonenvironmental externalities and the step from damage to externalities. Uncertainty was addressed through a simpler analogue of a Monte Carlo simulation analysis. An internal, quality control/peer review system was used.

III. COMPARISON OF AGGREGATE RESULTS

In this section we compare the aggregate results for three of the damage costs studies: Lee et al, HB, and EC. Because the studies estimated damages arising from several sites, we chose sites to be roughly comparable in location (metropolitan fringe, as opposed to rural or next to a large city).

We exclude TER from most of the discussion because results are reported in terms that make comparisons difficult. With multiple plants of different types in the TER scenario, one cannot easily relate damages to specific types of generation plants, a goal of the other studies. However, we refer to TER when that study makes original contributions to methods development. We ignore the other two studies (NERA and RER) because in our judgment consideration of more studies will not add significantly to our overall evaluation.

Table 1 provides the "raw" results from the three comparable studies. Note that these estimates are for "damages" (i.e., the monetary value of impacts) not Pareto-relevant externalities and that they are provided in terms of the mean damages in mills/kWh. Another section below will address uncertainties. These estimates are accompanied by some important qualifications that explain why a broad-brush comparison of the values is misleading. The most important qualification for the magnitude of the damages is that global warming damages are not included in any of the studies. Another qualification concerns the pathways included in the study, in particular, whether the study featured estimates of (i) occupational health damages, (ii) road damages and (iii) sulfur dioxide (SO2) damages. The last issue arises because electric utilities in the U.S. are subject to a tradable permit program for SO2, with the consequence that increases in SO2 emissions by a new plant would have to be offset by

Table 1 is available from the authors at Resources for the Future.

reductions elsewhere.[5] Both the Lee et al. and the HB studies directly addressed this issue, while the EC study did not because such a program is not in place in Western Europe.

The table also provides estimates of the private costs of electricity generation in the appropriate service territories as a means of benchmarking the damage estimates. Since private costs vary by study site, the reported cost serves as a rough benchmark for comparison with the social cost estimates.

At least with respect to the estimates for the U.S., the damages for each of the fuel cycles are "small," only a few percent of private costs. They are also small relative to estimates from earlier studies, such as Pace (1990), which estimated damages for coal of 60 mills/kWh. And they are small with respect to "adders" in use in many states in resource planning, which range from 10 to 12 mills per kilowatt-hour for coal. The damages estimated by the EC study are much larger than those from the two U.S. studies but, as will be seen in the reconciliation below, when several straightforward adjustments are made, the EC estimates are reasonably in line with those of Lee et al. and HB.

Why are the damage estimates so small? Partly, the reason is that the plants are new and are therefore subjected to New Source Performance Standards and what is now a strict siting process; therefore their emissions are quite low relative to some of the existing, dirty coal and oil-fired plants that are commonly associated with pollution in the electric power industry.[6] Partly, the low estimates are a result of omissions of environmental endpoints, although, with the exception of global warming, long-term ecological damages, and the possible effect of air pollution on the incidence of chronic respiratory disease, we feel that the studies are reasonably comprehensive. In particular, the effects of pollution on health, crops, and for HB, visibility and materials, are as well represented in these estimates as is possible, given the literature.

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[5] See for instance Hobbs (1992).

[6] For example, emission rates for some pollutants from a coal-fired facility can vary by up to an order of magnitude or more according to the vintage of technology (Rowe, Smolinsky and Lang, 1996).

The rank ordering of damages across fuel cycles generally meets prior expectations, at least for the fossil fuels, with the coal cycle causing greater damages than the oil cycle and the oil cycle causing greater damages than the natural gas cycle. Given the differences in private costs for generating electricity with these technologies, the rank order in terms of private costs for *construction of new facilities* would be unlikely to change if these facilities were ranked in terms of social costs (i.e., if the damages were assumed to be Pareto-relevant externalities and added to private costs).[7]

The low estimates for the nuclear cycle may appear surprising. These damages are entirely based on expert engineering estimates of accident probabilities and consequences (as opposed to estimates of risks by the general public or environmental groups, or those of Dubin and Rothwell (1990). The estimates support industry statements that the risks of nuclear accidents are very low, though as pointed out by Krupnick, Markandya, and Nickell (1993) the measure of economic costs, stemming from public perceptions, is problematic and could be much higher. What is more surprising is that the U.S. and EC teams operating independently offer essentially the same answer. The high estimates for biomass may also seem surprising. These damages reflect relatively high air pollution emissions and account for net erosion damages and other effects of tree plantations. If significant estimates for global warming damages were included, biomass technologies would fare relatively better when compared to fossil technologies (see below).

In spite of the generalizations made above about the results, the estimates differ significantly for different fuel cycles. Such differences can arise quite expectedly for a variety of reasons, including technology specifications, and initial conditions (air quality, climate, population, etc.), and methodological choices.

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[7] Freeman and Rowe (1995) explicitly compare these rankings in the New York State context.

IV. COMPARISON OF APPROACHES

Table A-1 in the appendix contains pertinent information for some of the variables identified pertaining to the coal fuel cycle as addressed by the three studies.[8]

Endpoints

The differences in endpoints receiving significant attention between the U.S. and European teams is instructive. Most importantly, the U.S. teams emphasized visibility, particularly urban visibility, and de-emphasized forestry and materials damages -- opposite to the European team, which mounted a special effort to build a materials damage inventory and ignored visibility. The European team also estimated damages from noise, an issue absent from the U.S. efforts. All teams emphasize the air-health pathways, however. The Lee et al. study is distinguished from the other studies by leading the broadening of endpoints into so-called nonenvironmental areas, such as employment and fiscal effects (see below). The EC and the Lee et al. study put significant emphasis on occupational and fuel transport damage estimates, such as from deaths resulting from collisions with coal transport by rail.[9] The HB study was particularly distinguished in the non-air pollution pathways it was able to analyze.

Spatial Boundaries

The spatial boundaries (modeling domain) of the analyses are quite different across studies and even among pollutants. Primarily these differences were driven by sponsor interests and by available data and modeling capabilities. In some cases, limitations in available models appropriate to the scale of these efforts have forced the teams to take ad hoc approaches in setting boundaries, actually introducing discontinuities in damage functions where none may exist in reality. HB, for instance, used models that required ozone scavenging by nitrogen oxides (NOx) and assumed zero secondary particulate formation within 50 km of the source while assuming no ozone scavenging beyond this point.

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[8] We add a column for the TER study as further contrast, but in the interest of space do not devote text to it.

[9] Damages from occupational accidents may be partly or fully internalized into wages.

Plant Characteristics

Differences in plant utilization across studies are normalized by presenting estimates in mills per kWh terms (rather than by another popular measure of damages, mills per ton of emissions). Some of the emissions coefficients are significantly different across studies, reflecting coal quality, abatement equipment assumed to be in place, and design features of the plants. These differences can easily be accounted and corrected for in reconciling damage estimates.

Air Quality Modeling

Each of the teams approached the air modeling problem similarly, dividing the problem into short-range and long-range transport of conservative and chemically reactive pollutants. There is little disagreement about the short-range conservative pollutant modeling problem. Each of the studies used a version of the EPA-approved Industrial Source Complex Long Term (ISCLT) models (U.S. EPA, 1992) With EPA approval limited to 50 km, however, some of the studies switched to long range transport models for conservative pollutant concentrations beyond this limit. Lee et al., however, simply used this model beyond, indeed, far beyond, the accepted boundary.

For chemically reactive pollutants (ozone and secondary particulates (PM10)), the strategies were more varied, not surprising since the modeling challenges are hardest in this area. Considering ozone (both a local and long-range pollutant), Lee et al. used a series of models to estimate ozone concentrations resulting from power plant NOx emissions in a NOx-limited region. HB did the same, assuming the study area was NOx-limited. Both teams relied to some extent on the Empirical Kinetic Modeling Approach (EKMA), specifically the OZIPM-4 model (SAI, 1987), which is a trajectory model generally used to simulate an hourly peak concentration over a single trajectory, rather than a multi-day episode throughout a city or region. However, Lee et al. developed a new model (MAPO3) around the generic OZIPP model that provided greater temporal and spatial resolution. HB, in contrast, used a simple graphical output of the OZIPP model to derive an equation relating the increment in the average annual ozone concentration to the NOx emissions increment, as a function of the total

NOx concentration. The studies dealt with NOx scavenging (the ozone depletion that results near the power plant stack when NOx emissions enter an existing ozone plume) very differently, as well (see below).

To address ozone formation and transport in completely defensible ways would have required much more sophisticated modeling, such as the use of the Urban Airshed Model,[10] that would be impractical for these already complex studies. It is unclear how much, if any, bias is introduced by using these simpler modeling strategies, although uncertainties are surely magnified.

Secondary PM10 formation was modeled by both HB and the EC. The former assumed that there would be no short-range sulfate or nitrate formation and constructed an original model (SLIM3) to estimate long-range conversions. The EC team used the Windrose Trajectory model (derived from the Harwell Trajectory Model) to estimate secondary particle formation over the modeling domain.

Concentration-Response (C-R) and Valuation Functions

The most important concentration-response (C-R) functions are linearized by all the studies, generally in a semi-log form, where C-R coefficients give the percentage change in an endpoint for a unit change in pollution. In such cases, initial conditions for the endpoint (e.g., the mortality rate) are necessary to obtain the absolute change in the endpoint measure.

With the exception of Lee et al. for some clinical C-R functions for symptoms, none of the studies specify health C-R functions that permit the marginal response to vary continuously with the size of the change in or level of concentrations.

This is not to say that there are no nonlinear C-R functions included. Lee et al. and HB (the latter at the user's option with the computer program EXMOD) have initial pollution concentrations that serve as a threshold, below which value there is no presumed response and above which the slope coefficient of the C-R function applies. Lee et al. builds in a 0.08 ppm

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[10] Even the use of three dimensional grid models, such as UAM, are limited when it comes to estimating long-range transport of pollutants. See National Research Council (1992) for a description of various types of ozone models.

threshold for daily ambient ozone peaks (the current National Ambient Air Quality Standard is 0.12 ppm) and performs sensitivity analysis with and without a 0.30 ug/m3 response threshold on annual PM10 concentrations.[11] A threshold means that below a given area's baseline ambient concentration, an increase in concentration has no detrimental effect on health (except to the extent that the new concentration exceeds the threshold). Such thresholds are very important because areas (such as a county) that have baseline concentrations below the threshold are zeroed out in the damage calculations for small changes in concentrations resulting from changes in emissions.[12]

Table A-1 also compares the approaches taken by the studies to estimate damages for other pathways, including nonconventional health pathways (such as occupational health, accidental deaths to the public from transport of fuel and health effects from mercury exposure through the air and eating contaminated fish); visibility damages (both recreational and residential), materials damages (to conventional structures and materials, not historic structures), crops, recreational fishing (primarily from acidic deposition), a host of additional environmental pathways with small effects (including noise, loss of open space and existence value losses from groundwater pollution), and nonenvironmental pathways (including employment effects and road damages).

We offer a few observations and draw attention to some "secondary" findings:

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[11] In addition, Lee et al. use a function for estimating adult chronic bronchitis cases, which implies that for a non-zero damage, a region has to evidence at least 10% of its days with PM10 exceeding 55 ug/m3 (100 ug/m3 TSP).

[12] It is worth noting the approach taken by TER for valuing acute health effects, which fit a "quality of well being" score and other variables to the willingness-to-pay (WTP) estimates for a variety of health endpoints. The resulting estimates for one-day symptoms are higher than those used in the other studies. Also, TER uses a C-R function relating NOx concentrations to eye irritation. This function ends up contributing 36% to the health damages from air pollution. The epidemiologists on the other teams gave very little attention to the effect of NOx on health, as there are few C-R functions that show NOx has any effect on health at ambient concentrations common to the areas in the modeling domain. Finally, TER uses a function relating PM10 exposure to the probability of having chronic respiratory diseases, such as emphysema and asthma. The authors of the underlying study strongly cautioned against the use of their very tentative results. Nevertheless, this linkage is a most active area of current, still quite inconclusive, research in the health science community.

- The damage from accidental deaths to the public as a result of the transport of coal by rail is an unexpectedly major pathway (Lee et al.).
- Occupational damages are relatively large; but we note that such effects may be internalized.
- Health damages from mercury exposures are exceedingly difficult to estimate but HB has made a start.
- There is consensus for a nonlinear valuation function for visibility with diminishing marginal values with baseline distance.
- Welfare-theoretic estimates of materials damages are based on old, problematic studies and "ginned up" inventories. The engineering cost estimate from the EC team is based on a major original effort to establish inventories. The damages are relatively large even with an incomplete analysis.
- Crop damages are easy to estimate from estimates of yield changes. Yield C-R functions are widely available for major U.S. crops and therefore, are important for analyzing damages in the grainbelt
- The only reasonable acidification-recreation damage estimates are for the Adirondacks (e.g., Englin, 1991). There is some consensus on the use of MAGIC[13] (Sullivan and Cosby, 1995) for modeling effects of deposition on water body chemistry. HB estimated fishing damages from entrainment and thermal plume.
- None of the studies successfully address damages to forests, whole ecosystems, and nonuse values.

Greenhouse Gases

Though the studies devote considerable attention to global climate change from greenhouse gas (GHG) emissions, all conclude that damage estimates in the literature are too

[13] Model of Acidification of Groundwater in Catchments.

uncertain to be included with other estimates. Some critics may say that leaving out GHG damage estimates and arguing, as we will below, that the other damages from *new sources* of electricity are relatively small, is like trumpeting the successful takeoff of an experimental plane without mentioning that it crash landed. If GHG damages ultimately prove to be large, any analysis that doesn't include them will be highly misleading.

The EC provides a nice summary (reproduced, in part, below as Table 2) of the estimates available in the literature, arrayed by the discount rate being assumed. Each of the studies concludes that it is not currently possible to use an impact pathway approach to provide reliable estimates of damage, and they place very low confidence on the range of estimates from the literature. However, in the context of greenhouse gases, the use of a unit value per ton of emission or per kilowatt-hour is much more robust than for other pollutant and environmental pathways because these gases are uniformly mixing in the atmosphere and because emission rates vary little with regard to fuel usage. Several studies provided a useful rule of thumb: each $1 of damage estimated or assumed to result per ton of carbon dioxide (CO2) roughly translates into 1 mill/kWh. In view of the "small" damages estimated for other pathways, even a relatively "small" estimate for global warming damages can dominate damages from other pathways.

Table 2: Range of Damage Estimates for Global Warming from Coal

	Damages (mills/kWh) under various discount rates			
Source	0%	1%	3%	10%
Cline (1992)	18.6		2.8	0.8
Fankhauser (1993)	13.0		1.9	0.5
Tol (1994)		22.9	14.6	3.3

note: Converted at $1.25 = 1 ECU.

Note: The EC also included estimates from Hohmeyer, O., and Gärtner, M. (1992) The Costs of Climate Change. Fraunhofer Institut fur Systemtechnik und Innovationsforschung. These are 65 times larger than Tol's at a 3% discount rate.

Energy Security

The term "energy security" is used to describe a variety of issues, ranging from the economic cost of oil supply disruptions to the cost of military expenditures to secure international trade. None of the studies calculated damages for energy security. Lee et al. addressed the issue directly and sponsored a detailed literature review and analysis on the subject (Bohi and Toman, 1992), which concluded that energy security damages were likely to be very small, and the uncertainty around such estimates could not be characterized in a way that would allow quantitative probabilities of damages or monetary estimates. The study pointed out that this conclusion remains controversial, and cites Green and Leiby (1993) for an alternative view that maintains energy security damages are significant. All the other studies we reviewed either do not address the issue, or reach a conclusion similar to Lee et al.

Population

The damage estimates expressed in any terms other than a per person basis will be highly (generally proportionally) sensitive to the population assumed to be affected by the new plant. There are huge differences in the total population included in the studies, ranging from 93 million in HB to 477 million (basically the population of Western Europe). There are also differences in some of the estimates of population subgroups (fraction of children, fraction with asthma) and in baseline health conditions (the mortality rate).

Discounting and Base Year

Lee et al. use a 5% rate, EC uses a 3% discount rate,[14] while HB uses a 5% discount rate as a default. The particular rate doesn't have much of an effect on damages for the coal cycle, however, (although it can make a big difference for the nuclear cycle), because few of the pathways involve effects that are cumulative or delayed, i.e., where discounting would be required. Finally, the base years for the studies are somewhat different. To the extent that the valuation functions or unit values are drawn from the same basic literature, the latter studies will feature higher damages, due to inflation.

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[14] Lee et al. estimates nuclear damages with both a 3% and 5% rate to be comparable to the EC.

Uncertainty

Finally, the studies address uncertainties in different ways. Lee et al. uses Monte Carlo simulation techniques to capture and appropriately propagate uncertainties in key parameters through the various linkages of the analysis. HB, due to tighter computer resource constraints, took a short-cut approach, using a technique that propagates a three-point probability distribution (a beta distribution) for each uncertain parameter rather than a continuous distribution. The EC did not perform an uncertainty analysis.

V. RECONCILIATION OF AIR-HEALTH PATHWAYS

The largest fraction of *quantifiable* damages (in mills per kWh) associated with the fossil fuel cycles in all of the studies is attributable to the air pollution-health pathways: 55% for Lee et al., 82% for the EC, and 93% for HB. For this reason, we choose these pathways for the coal fuel cycle for a somewhat detailed comparison and reconciliation of approaches and results. The purpose is to learn what portion of the variation in damage estimates is explained by explicit differences in assumptions or site characteristics, and what portion can not easily be explained. If a large portion of the variation can be explained, this would suggest that the methods employed yield replicable and robust results, evidence in favor of their credibility.

Table 3 provides the summary results for these pathways and studies and table 4 provides the details of the reconciliation. In a "raw" comparison, the three studies appear to come to very different conclusions about the size of these damages. For roughly comparable locations and technologies, Lee et al. estimates mean damages of 0.72 mills/kWh, while the EC study estimates damages of 15.63 mills/kWh, and HB estimates mean damages of 2.70 mills/kWh. Underlying these damage estimates are pathways linking direct and secondary PM10 (through NOx conversion) to mortality and morbidity, NOx as an ozone precursor to mortality and morbidity, NOx as a gas to morbidity, lead to mortality and morbidity and SO2 as a gas and as secondary PM10 to mortality and morbidity. Some of these pathways are combined in the tables for ease of exposition.

Table 3 is available from the authors at Resources for the Future.

Table 4. Detailed Reconciliation of Air-Health Pathways

Factor causing adjustment (Lee et al; EC; HB)	***Lee et al***	***EC***	***HB***
Total Health Damages (mills/kWh)	0.72	15.63	2.7
Impute NOx-PM10-Damages	0.85		
Drop SO2 and NOx-Ozone-Mortality pathways		-6.3	-0.75
Adjust PM10-Mortality Coefficient (0.64%; 1%; 1%)		-2.7	-0.42
Adjust Mortality Rate (960/100,00; 990/100,000; 800/100,000)		-0.15	0.14
Adjust NOx emissions coefficients (2.6 g/kWh; 0.8;2.1)		9.5	0.38
Adjust Direct particulate coefficients (0.14 g/kWh; 0.18; 0.14)		-0.55	-0.02
TSP-PM10 conversion (TSP*55%=PM10; 90%; 55%)		-0.64	0
Ozone population adjustment (7.8 million; 9 million;NA)		-0.02	0
Eliminate Ozone scavenging		no adjustment possible	0.13
Morbidity Value Adjustments (PM,Ozone)		0	-0.66
Particulate and lead population adjustments (193 million; 477 million; 93 million)		-8.87	2.04
Total Adjustments	0.85	-9.73	0.84
Total Reconciled Air-Health Damages	1.57	5.90	3.54

We make a preliminary adjustment to address the omission of the NOx-secondary PM10-health pathways by Lee et al. These pathways are clearly important to estimating the damages from power plant emissions. Because of their importance, for the reconciliation, we crudely impute damage estimates to the Lee et al. study from HB, by assuming that the ratio of mortality damages from direct particulates (which both studies estimate) and secondary particulates for the HB study applies to the Lee et al. study. This results in an increase in damages of 0.85 mills/kWh in the latter study.

The remaining reconciliation proceeds by adjusting the EC and HB studies to Lee et al. The first adjustment is to drop the SO2 pathways. As described previously, differences in the treatment of SO2 relate primarily to differences in *environmental policy* towards this pollutant in the U.S. and Europe. The tradable SO2 allowance program in the U.S., required by the Clean Air Act Amendments of 1990, means that increases in SO2 from a new power plant are permitted only if the new plant has allowance permits for such emissions. Given that the total number of permits available is fixed, these additional emissions will be offset by reductions in emissions elsewhere. As a first approximation, it may be presumed that net damages from SO2 are zero.[15] In Europe, in contrast, with no such offset system, SO2 emissions are presumed to increase with an additional coal plant. Hence it is appropriate for the EC to count the SO2 pathways.

The second adjustment is to set the mean ozone-mortality effect to zero. There is some limited epidemiological evidence that daily ozone concentrations are related to the risk of death. This evidence comes from two studies by Kinney and Ozkaynak (1992), one for New York, the other for Los Angeles. These studies used daily time series of death rates and

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[15] In practice, both U.S. teams developed similar approaches to estimate the net effects (which for the case shown under HB turn out to be a net damage). The HB study assumes the location of the seller of a permit will not be known, hence they assume the region around the location of the seller has an average population density of 38 persons per square kilometer and meteorological conditions are identical to that of the buyer's location. Therefore, the net effect of the trade is to increase damage if population density in the buyer's location exceeds 38 and to decrease damage if density is less. Lee et al. take a more targeted approach, estimating population density for states with likely sellers of allowances to compare to population density at their study sites. They find the density of areas likely to be supplying allowances is 112 persons per square kilometer, or almost three times that used by HB. Lee et al. forecast that net benefits may accrue from the trading program, but for the purposes of their summary and conclusions, they maintained that the net effect of the program would be neutral.

pollution levels, following protocols similar to those followed by Schwartz and Dockery in their widely accepted studies establishing a link between particulate concentrations and mortality (1992a, 1992b). However, unlike the body of particulate-mortality studies that find a relationship there, the cross-sectional studies have not identified an ozone-mortality link and the Schwartz and Dockery studies found no such link, either (although ozone levels were far lower in the cities they examined). The draft *Ozone Staff Paper* (U.S. EPA, 1995) concludes that the effect of ozone on mortality is not well enough understood to use the literature as a basis for setting the ozone National Ambient Air Quality Standard (NAAQS). Lee et al. concluded that it is premature to accord this link a central role in their damage estimates and follow NERA (1993) in assigning only a small probability that these effects exceed zero. Specifically, for the Monte Carlo analysis, Lee et al. assigned 90% of the mass at zero, with 10% normally distributed around the Los Angeles study point estimate.

The EC study used this point estimate for illustration, but withheld including this endpoint in its final analysis. HB used the New York City results as a point estimate, which show a somewhat larger effect than the Los Angeles results

The third adjustment -- in the PM10-mortality pathways -- highlights issues of the interpretation of evidence and benefits transfer. Each of the studies reviews the epidemiological literature and comes to basically the same conclusion: there are significant increased mortality risks from exposure to PM10, although there is a fair range of uncertainty about the magnitude and the size and composition of particles causing the effects.

To process this information for a benefits transfer, HB and the EC took the average of the results of the ten or so daily time series studies in the literature (1.0 % increase in the total mortality rate for every 10 ug/m3 increase in PM10), although, here again, the EC was reluctant to include these controversial effects in its summary tables..[16] In contrast, Lee et al.

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[16] HB and TER actually disaggregated the mortality function into two functions, for those aged 65 or more and for those less than 65. TER advanced the debate by performing a meta-analysis on these studies that the marginal effect on the mortality rate was slightly nonlinear (decreasing) with baseline PM10 concentrations. Nevertheless, they used the linear model in their damage calculations and did not seek to use the meta-analysis results to particularize them for their northern midwest modeling domain

chose the results of a study performed in an area included in the study's modeling domain -- Steubenville, Ohio -- which showed a smaller than average response of mortality rates to changes in PM10 (0.64% change in mortality rates per 10ug/m3).

An additional adjustment to damages from this important endpoint concerns the baseline mortality rate, which is slightly higher for the EC than for Lee et al. (a rate for the modeling domain), while the HB rate (a national estimate) is lower. Substituting the Lee et al. rate raises EC damages for this endpoint and lowers HB damages, widening differences between the studies.

The fourth adjustment concerns emissions coefficients (expressed in g/kWh). In each of the models, larger coefficients raise damages proportionally (in mills/kWh). Direct particulate emissions for the EC coal plant are 30% larger than those from the U.S. plants. However, nitrogen dioxide emissions for the EC and HB plants are only 31% and 73%, respectively, of Lee et al. plant emissions.[17]

A fifth adjustment to reduce EC estimates associated with direct particulates addresses assumptions made about the particle size distribution of PM10 versus all particles, called total suspended particulates (TSP). The U.S. and EC studies adopt different conventions about the PM10 component of TSP (see Table A-1). Other things equal, this convention will result in an overestimate of direct particulate mortality and morbidity damages relative to the U.S. studies by about 60%.

Another set of adjustments concerns the unit values used to compute morbidity benefits for PM10 and ozone pathways. The EC used Lee et al. estimates for the dominant endpoints, but higher estimates for some of the less important endpoints. HB used higher estimates for most endpoints, including the dominant ones. The differences between Lee et al. and HB are mainly a result of judgments about central tendencies in the valuation literature and use of different base years.

The last set of adjustments concerns the population in the modeling domain. Without modeling nonlinearities, changes in population affect damages proportionally. However,

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[17] A NOx emissions adjustment was not made to the NOx-Ozone-Morbidity pathway because of the highly non-linear nature of the ozone modeling.

because of the thresholds introduced in air modeling and the different air models used to model short-range and long-range changes in concentrations by the teams (and in C-R functions by Lee et al.), it is not possible to make an accurate adjustment for the effect of using different population levels. That is, the distribution of the population throughout the modeling domain matters in some of the calculations. Nevertheless, as a crude adjustment, we assume linearity and scale the damages to reflect total population differences. The total population in the EC analysis is 2.5 times that in Lee et al., while the population in HB is 48% of Lee et al.

For damages from ambient ozone, using the entire population in the reconciliation is more questionable. The modeling domain for ozone was much smaller than for PM10 in these studies. In fact, only 7.8 million people live in areas within the Lee et al. ozone modeling domain, while 9 million live in the EC domain.[18] et al. Thus, a simple adjustment would be to decrease the EC estimate to account for the smaller Lee et al. target population.

The appropriate adjustment for HB is unclear, in part, because the computational algorithm followed by HB is complex. Most of the C-R functions used by HB are identical or very similar to the Lee et al. functions and, if anything, target sub-populations are defined more inclusively (i.e., are larger as a fraction of total population) by HB.[19] Further, the report suggests that ozone changes were estimated at all receptors registering changes in NOx, i.e., perhaps the entire modeling domain. Therefore, one would expect very large ozone damages relative to Lee et al. Yet, the HB estimates of ozone morbidity damages are half of those for Lee et al.

One possible explanation for the relatively small damages found by HB is that unlike Lee et al., HB assumes that NOx increases within 50 km of the source (home to 638,000 people) will lead to a "mole-for-mole" reduction in ozone, creating a benefit for these pathways that is subtracted from damages to the population living beyond the 50 mile limit.

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[18] Both Lee et al. and the EC find that NOx scavenging is confined to about a 10 km radius, so this issue should not confound the adjustment process.

[19] For instance, Lee et al. applied the Krupnick, Harrington, and Ostro (1990) C-R function for any symptom-day to the adult population only, based on a finding of no effects in children using the same data set. Hagler Bailly applied this C-R function to the entire population.

Indeed, Rowe et al. (1995a) note that eliminating scavenging would increase ozone damages by 2-3 times. Thus, for purposes of the reconciliation, we multiply these HB damages by 2.

After these adjustments, there are a number of potentially important differences remaining for which we did not develop reconciliation factors. First, we have not addressed the morbidity C-R functions directly, because choices about functions for these pathways are either quite similar across the studies or would make little difference to the damage estimates. An exception concerns thresholds in ozone C-R functions. Lee et al. used a 0.08 ppm threshold, with the consequence that damages are registered on only 51 days of the year. The EC used a 0.12 ppm threshold, with the consequence that damages would only occur on 75 summer days. Removing these thresholds would raise ozone damages dramatically. Note, however, that the EC damage estimate for ozone-morbidity is *lower* than the Lee et al. estimate, suggesting that the greater number of days admitted into the EC calculation was more than offset by other factors. These factors did not include to any great degree differences in C-R functions.

The much taller stack height for the EC plant relative to the U.S. plants could also be accounting for some differences, depending on air chemistry embodied in the models, assumptions about mixing height and other meteorological variables, population distribution, and C-R functions (if nonlinear). Rowe et al. (1995a) find that halving the stack height results in a 6% increase in damages for a Syracuse, N.Y. site but an 82% increase in damages for a site at JFK International Airport. Based on these results, adjusting for the higher EC stack height should further raise EC damages above Lee et al. and HB, but the EC damages are already significantly above this study's estimates.

Conclusion about Reconciliation

The net effect of the reconciliation is partly encouraging and partly disappointing. The damages from the EC study, with corrections for its very large target population and focus on SO2 emissions, approach those of the other studies. In particular, the EC and Lee et al. estimates of direct particulate-mortality are now nearly identical.[20] Another encouraging

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[20] A remaining area of concern in interpreting this result is that the direct particulates-PM10-morbidity estimates are not similarly close. With the C-R and valuation functions nearly the same and the direct damages

finding is that Lee et al. and HB agree closely on the NOx-ozone-morbidity pathway damages when scavenging is removed from the latter's ozone model.

The disappointing results are, first, that the unreconciled gap between the Lee et al. and HB studies has grown rather than shrunk, primarily because the HB population is smaller and the coal plant is cleaner than Lee et al.'s. Second, a major gap remains between the Lee et al. and the EC estimates because of the Lee et al's much smaller estimate for the NOx-PM10-mortality and morbidity pathways. But, this estimate is strictly the result of the imputation from the HB study to the Lee et al. study. Thus, the major source of difference is the low ratio of NOx-PM10-mortality damages to direct particulates-PM10-mortality damages in the HB study relative to the EC study. Given the reconciliation above, this gap can only be based on differences in air quality modeling. Since all the studies use the ISCLT model for direct particulates (at least within 50 km), short-range direct particulate model choice can be ruled out. However, there are a number of other choices that confound the issue: choices of meteorological parameters, assumptions about baseline ammonia levels and other initial pollution conditions, and the air chemistry embodied in the models themselves.

Third, note that the gap in the NOx-ozone-morbidity pathway damage between Lee et al. and the EC has not closed, a particularly surprising development in view of the use of nearly identical C-R and valuation functions. Again, ozone modeling differences must be implicated. In particular, based on figure 3.6 in the EC Coal Study that shows no tapering off in changes in ozone concentrations at the assumed geographic boundary for ozone effects, there are apparently large changes in ozone concentrations occurring outside the boundary of the EC ozone analysis.

Thus, differences in air quality modeling seem to be the primary unreconciled factor contributing to differences in the studies' damage estimates. Unfortunately, there is insufficient information in the studies to probe these issues further. However, some idea of the importance of these differences can be gleaned from Knecht and Levine (1995), who compared ambient pollution effects for a natural gas power plant sited in Southern California as estimated by the

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for the particulates-mortality pathway the same, implying that the direct particulate to PM10 modeling choices are nearly the same, there would seem to be no logical explanation for this difference.

Air Quality Valuation Model (AQVM) (Thayer et al., 1994) and by EXMOD (HB). The former model uses basin-wide (space independent) transfer coefficients for the concentration of PM10 resulting from a unit of NOx or SO2 emissions, while the latter, as noted above, assumes there is no nitrate or sulfate formation within 50 km and then uses an air quality model (SLIM3) to forecast secondary particulate formation outside this radius. These differences in approaches led to one to two orders of magnitude larger estimates of secondary particulate concentrations from the AQVM.

VI. NONENVIRONMENTAL DAMAGES

The Lee et al. study is distinguished from other studies by its relatively heavy emphasis on nonenvironmental damages. Estimation of some types of these damages, such as road damage and occupational health effects is not particularly controversial. Road damages from coal trucks, for instance, can be estimated from information on the type of road surface being used, the weight and number of axles of the trucks, and the frequency of use. Whether these damages are "Pareto-relevant externalities" depends in a straightforward way on whether the roads are publicly owned or owned by the coal company, and whether road use fees calibrated to the damage of the vehicle are in use. Similarly, occupational health effects are generally considered not to be Pareto-relevant externalities, if one assumes workers understand the risks they take and their wages and compensation packages are determined in competitive labor markets, where premiums would be paid for riskier work. Controversy abounds, however, in considering two other types of nonenvironmental effects: employment and fiscal.

Employment

In a fully employed economy, by definition, employment damages (and Pareto-relevant externalities) are zero. However, where unemployment is above the "natural" rate, Lee et al. argue that changes in employment (appropriately estimated and valued) can be considered a damage and a Pareto-relevant externality for purposes of a comparison of social costs across fuel cycles, because of the implied difference between the labor wage and the social opportunity cost of labor resources (Hamilton, et al., 1991). Lee et al. attempted to generate comparable estimates of employment effects across all fuel cycles. The EC study applied its methodology only to the hydroelectric and natural gas fuel cycles. The HB study avoided a detailed analysis, relegating employment effects to future research.

Two observations about estimated employment damages are worthwhile. First, the estimates vary greatly across the study sites, which would be expected if damages depend largely on conditions in the local or regional labor market. However, the range of estimates resulting from sensitivity analysis at each site is broad. Both Lee et al. and the EC studies refrained from reporting these estimates within summary tables of estimates from other pathways because of uncertainty about estimation methods. Second, these estimates are consistently large compared to those from other pathways, including health. The estimates indicate that improvement in the calculation of employment benefits would deliver valuable information about social costs and should be a priority for further research.

Fiscal Effects

Standard welfare analysis describes tax payments as transfer payments and as a pecuniary externality, rather than a technical externality relevant to the attainment of Pareto efficiency. However, from any of a variety of perspectives, the *differences* in tax payments embedded in choices among fuel cycles for generating electricity should be viewed as Pareto-relevant externalities in the consideration of social cost.

Taxes create a difference between the prices of goods and services and their social opportunity cost. If taxes affect alternative fuel cycles differently, then relative prices would vary *even if the social opportunity cost were equal.* Hence, differences in tax payments have the potential to create sizable financial advantages and disadvantages among fuel cycles, so that relative prices would not necessarily reflect the relative social opportunity cost of resources used by each fuel cycle.

Further, an important issue in public finance is the marginal excess burden of sources of government revenue. Raising one dollar of tax revenue may impose a gross cost on the economy (not counting welfare benefits from government spending) of $1.20 to $2.00 in lost welfare, depending on the nature of the tax. To the extent that fuel cycles employ different factors of production that are treated differently by the tax system, then one fuel cycle may impose relatively greater costs, even if tax revenues raised by the two fuel cycles are equivalent.

The EC study of the hydroelectric fuel cycle calculated income tax revenue and developer fees paid from construction and operation of the facility. The study chose not to treat these revenues as social benefits because of the expectation that another project with the same cost would have given rise to the same total government revenues. Unfortunately a comparison with tax payments stemming from other fuel cycles was not attempted.

Lee et al. draw attention to the more important question of the difference in tax payments among fuel cycles. The project supported a comprehensive study that employed discounted cash flow models to investigate taxes paid in construction and operation of a facility, fuel mining and transportation, and indirect tax revenue from personal income accruing to labor and capital (Burtraw and Shah, 1994). For the generation stage of the fuel cycle, the study considered reference environments for investor owned utilities in California and Massachusetts.

A sample from Burtraw and Shah of fiscal effects in the Massachusetts reference environment is presented in Table 5. Direct taxes include tax payments in the construction and operation of the facility by an investor owned utility. Total taxes also include direct taxes paid by firms in fuel extraction and transportation that are embedded in the cost of fuel, plus indirect taxes paid as personal income tax on labor and capital income. The third column includes an adjustment for the relative marginal excess burden associated with various forms of taxation, which the authors consider less reliable, due to significant uncertainty about the magnitude of marginal excess burden in the public finance literature. The striking conclusion is that the *difference* in total taxes paid per kilowatt-hour by fuel cycle is of equal or greater magnitude than the difference in environmental damage identified by the three social costing studies reconciled above. Tax policies were found to afford financial advantage to natural gas, relative to coal and biomass technology.[21]

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[21] The study found differences between fuel cycles stemmed from the relatively favorable treatment of fuel costs, relative to capital and labor expenses. Also, costs that enter the rate base produce relatively greater fiscal benefits due to taxes on corporate and personal earnings. In contrast, upstream taxes in fuel supply are relatively insignificant. Another recent "bottom-up" study (Hadley, Hill and Perlack, 1993) employed a

Table 5: Comparison of Tax Revenues from Fuel Cycles in Massachusetts

mills/kWh ($1993)	Direct Taxes	Total Taxes	Total Taxes with Marginal Excess Burden
Natural Gas	3.10	5.07	5.34
Coal	8.30	13.33	14.12
Biomass	7.40	15.21	15.92
*Biomass**	*2.60*	*7.72*	*8.29*

Notes: Source: Burtraw and Shah, 1995.

*Includes Renewable Energy Production Credit. The authors' preferred estimate excludes this credit in part because there are no biomass facilities currently operating or planned that would qualify.

VII. ASSESSING THE LITERATURE

This paper set out to compare some of the recent social cost studies and in so doing to address two questions that should provide guidance for the interpretation of these studies and the future application of the methodology.

1. Are the Results Credible?

One of the overarching questions we asked is the extent to which estimates of these air-health and other types of damages are credible and acceptable data for policy making in the social costing context and the regions studied.

There is no unique definition of credibility and acceptability, but surely qualities such as transparency of approach, internal validity (i.e., appropriately sensitive to changes in input assumptions), replicability, consensus, plausibility, and degree of uncertainty are important factors. The damage function model, by its nature, will provide a reasonable degree of explicitness or transparency for understanding the inputs and steps used to calculate damages. For the coal fuel cycle, the air-health, crop damage, visibility, and some of the estimates of occupational health and public accidents (say from the rail transport of coal) potentially can

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discounted cash flow model to estimate taxes paid by investor owned utilities (IOUs), and to a limited extent by nonutility generators (NUGs). A third study (Viscusi, et al., 1994) used a "top-down" approach by applying average tax rates by industry as a measure of marginal tax rates.

make the grade. For these pathways C-R and valuation techniques and studies are sufficiently developed to provide the opportunity for internal validity and consensus.

We judge that these opportunities have been realized. The comparison of estimation approaches and the reconciliation of the studies (presented above) show that there is a clear consensus on the general approaches for estimating concentration-response and damages for what have turned out to be the most important environmental pathways, air--health. Nevertheless, there appears to be a lack of consensus and a need for careful exploration of choices made in modeling air pollution, with respect to the effect of choices about weather patterns, of baseline conditions, and about the appropriate representation of air chemistry.

The plausibility of the results can be tested in a limited way, by juxtaposing them with baseline impacts, such as the current prevalence of a particular disease, to determine if changes in emissions from a particular scenario result in unreasonably large reductions in impacts as a fraction of the baseline. In principle, this kind of testing could be performed for many of the health pathways considered in these studies. The *Health Interview Survey* (NCHS, 1996), for instance, could be used for testing the plausibility of acute health effects estimates; mortality statistics could be consulted for testing dose-response functions for premature death. Krupnick and Portney (1991) performed such a plausibility test for the particulate-mortality functions applied to attainment of the National Ambient Air Quality Standards for particulates in Los Angeles. These functions implied, implausibly, that meeting particulate standards would cut deaths from respiratory disease by about half.[22] More research of this kind could usefully be performed.

The uncertainty about these pathways is of three basic types: (i) the statistical uncertainty around the mean values for the pathways estimated; (ii) qualitative uncertainty about the models chosen to provide estimates; and (iii) uncertainty about the pathways left out of the estimations.

The statistical uncertainties are easy to characterize. The distributions of the estimates are relatively consistent across studies and reasonably "tight" where the largest uncertainties in

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[22] However, this analysis assumed a zero threshold for the dose-response function. Lee et al. used a threshold of 30 ug/m3 and zero. The HB EXMOD model uses a zero threshold as a default.

damages in the four studies reviewed, as a ratio of the high estimate (ranging from the 80th to the 95th percentile) to the mean estimate for the air pathways are a bit more than 2.

The model uncertainties are obviously much more difficult to characterize. C-R functions for the effects of PM on mortality are driving the concern in national regulatory policy for a possible move to a fine particle standard (PM-2.5). At the same time, there is some suggestion that statistical life is being shortened by only several days by high daily PM10 readings, at least when temperatures are below 85 degrees (Wyzga and Lipfert, 1995). And, more importantly, there is not agreement among health scientists about the particle sizes and types that are most dangerous to health, particularly the relative role of sulfates, nitrates, and road dust.

Valuation functions, while enjoying a high degree of consensus in the social costing studies, are not free from controversy. There is a growing recognition that the compensating wage studies are problematic for valuing mortality risk reductions in an environmental context. The limitations of such studies for valuing environmental risks are four-fold: (1) they reflect risk preferences of perhaps a less risk adverse group than the average in society; (2) they reflect voluntarily borne risks; (3) more life years are lost to an accidental death than those associated with, say cancer, which has a latency period, and the effects may be discounted because they occur far into the future; (4) the source of the risk is an accident rather than, say, a business polluting as part of its normal operations.

To partly address these risks, the HB study above used the small literature relating age of death risk onset to willingness-to-pay to adjust the wage compensation study results to an environmental context. To make more substantive progress will require a more basic understanding of how people perceive mortality risks from pollution.

Another type of modeling uncertainty comes from the linearization of C-R and valuation functions, a common practice in the social costing studies. On the one hand, there are good conceptual reasons for expecting at most small departures from linearity in the aggregate damage function (Dewees, 1995) where the response to pollution by individuals, meteorology, and other driving forces can be assumed to be heterogeneous (over individuals, time, and space, as the case may be). Further, Curtiss and Rabl (1995) test whether *total* damages can be acceptably estimated with their proportional damage model, based on the idea that with linear C-R functions, holding total population in the modeling domain constant, the population distribution relative to the source of emissions is nearly irrelevant to the damage estimate because of conservation of mass. They find that for most atmospheric conditions this insensitivity to source location holds within an order of magnitude. On the other hand, there are numerous studies suggesting that nonlinearities are significant for air modeling, C-R functions, and valuation functions for a variety of pathways.

Last, there are uncertainties from what is left out. A key missing air pollution-health pathway, for instance, is the effects of long-term ozone exposure on chronic lung damage and respiratory disease prevalence.[23]

Turning to other environmental pathways, those likely to be most important, but for which more research is needed before they are judged as credible include: global warming, materials damages, any damages to terrestrial or aquatic ecosystems, other than that from acid deposition in the Adirondacks (where we feel there is a reasonable set of studies for damage estimation),[24] and damages to groundwater.

Fortunately, these studies have identified many pathways that probably can be safely ignored in future applications. Lee et al. and the EC found that all secondary effects (the emissions from manufacturing the cement used to pour a foundation for the new generation plant plus all the other indirect emissions) were two to three orders of magnitude smaller than the direct emissions from a coal-fired power plant, on a per kilowatt-hour (kWh) basis. HB, who pushed harder than any other studies to estimate damages from pollution of water and land, as well as land uses found that such damages were quite small relative to the damages from conventional air emissions, although the authors were cautious about generalizing their findings beyond the sites considered. HB also found that the effects of toxic air pollutants on health were insignificant.

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[23] A new study has been reported that shows a very strong link between ozone concentrations and asthma prevalence in males (McDonnell et al. 1996).

[24] The recreation valuation literature is probably more developed than for any other endpoint. However, the link from pollutant loadings in surface waters to catch per unit effort of some other valued impact is not well understood. Some studies have had some success modeling implicit linkages, say by using nitrogen loadings as a covariate to explain recreation behavior (Kaoru, Smith, and Liu, 1995).

Finally, with respect to the extent estimates are credible and yield acceptable data for policy making, one must ask to what degree fuel cycle damages can be compared? The short answer is that a rough ordering of the magnitude of estimated damages across fuel cycles confirms *a priori* rankings when the focus is placed on conventional pathways -- i.e., from most to least damage, coal, biomass, oil, and then gas. We observe this ordering even across studies that employ somewhat different methodologies including air quality modeling, and in light of uncertainties that have been discussed, and even in different geographical settings. However, the short answer may unravel when analysis is extended to include employment, fiscal and global warming effects, for which estimates are less credible.

Table 6 presents a comparison of three fuel cycles -- coal, gas and biomass. The first category reports estimated environmental damages excluding the SO2 pathways based largely on the Lee et al. estimates for "Knoxville."

Table 6. Estimates and Best Guesses of Damages

(mills/kWh)	**Coal**	**Gas**	**Biomass**
Environmental and Occupational Damage (Knoxville)	2.2[a]	0.3[b]	1.7
Employment (Knoxville)	(2.1)[c]	(0.6)	(1.5)
Tax Revenues (Mass., w/o REPC)	(13.3)	(5.1)	(15.2)
$C0_2$[d] (EC avg., 3% disc rate)	6.4	2.9	0
TOTAL	**(6.8)**	**(2.5)**	**(15.0)**

a. Includes imputation for secondary PM10 (table 3).
b. From HB. Imputation for PM10 to Lee et al is too uncertain.
c. Parentheses indicate negative values of damages, or equivalently, positive benefits.
d. $1.25 = 1 ECU. Note the sensitivity of these estimates to choice of discount rate, as indicated in table 2. The EC's preferred discount rate of about 0% would yield CO2 damages of 18.2, 7.8, and 0, for coal, gas, and biomass, respectively, and total damages (benefits) of 5.0, 2.4, and (15.0), respectively.

The next category reports nonenvironmental externalities. Both estimates of employment benefits and fiscal effects were not included in the final results of these studies. Nonetheless, the exploratory work that was performed illustrates the potential relative importance of these issues. Estimates of employment benefits are drawn from those calculated in the Lee et al. study for the Knoxville site. The employment estimates are of the same magnitude approximately as the subtotal for conventional damages, and in the opposite direction. (Parenthesis in the table indicate negative damage values, or benefits in other words.) The next category contributes illustrative estimates of the fiscal effects, measured as differences in tax revenues, resulting from investment and operation of each fuel cycle for an incremental investment in Massachusetts, taking into account fully the particular characteristics of that state's fiscal environment.[25] These estimates swamp either of those derived from environmental damages or potential employment benefits.

The final line in the table incorporates an estimate of damages from CO2, based on the average of the range of numbers presented by the EC study. These estimates are of greater magnitude relative to the environmental damages from conventional pathways.[26] Note that the range of estimates by fuel cycle is large, since biomass is assumed to be neutral with respect to CO2 emissions.

The total represents a compilation of estimates and best guesses of damages in comparing the three fuel cycles. Taking the less credible and more controversial estimates of employment, fiscal and global warming pathways into account, the ordering of social costs for the fossil fuel cycles is reversed, compared to the first line of the table that considered the more credible conventional pathways alone as reflected in the studies we review. The social "benefits" (negative costs) are least for gas, significantly greater for coal, and greater still for biomass.

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[25] Burtraw and Shah (1994). The estimates exclude the Renewable Energy Production Credit and effects stemming from the relative marginal excess burden of taxes (see Table 5).

[26] Note the sensitivity of these estimates with respect to discount rate. As indicated in the table notes, the EC's preferred discount rate of 0% would inflate these numbers substantially.

2. Are the Results Transferable?

Given that estimates pass the credibility threshold, the second overarching question is the degree to which these estimates can be transferred to other settings. Each of the three major studies we cite in our reconciliation examined more than one geographical location. Looking across locations within each study, strong evidence can be found that direct transfer of damage estimates (in $ per ton or per kWh) between geographic locations is inappropriate, both with respect to environmental damages which vary with geography, meteorology and population, and with respect to nonenvironmental externalities which vary with local economic features. The only likely exception is global climate change damages, which do not depend on the geographic source of emissions.

Some damage estimates such as damages to crops (from ozone) and damages from accidents are easy to estimate directly in various locations. Visibility damages, given an estimate of visibility change at the appropriate level of spatial and temporal aggregation, can be estimated using the literature evaluation and synthesis by Chestnut and Rowe (1990). Further research is needed to clarify the judgments about the visibility studies made in the social costing literature. In any event, the valuation function for estimating visibility damages must be transferred, rather than using an average unit value, due to the nonlinearity of the function.

For the most important pathways, air-health, given the lack of consensus about air quality modeling, the sensitivity of damages to such modeling, and the proportionality of damages to population exposed, we feel most comfortable with a measure that links concentrations to per person values, endpoint by endpoint, rather than a "mills per kWh" or a "dollar/ton" measure. We label this measure D^{U}_{ze}. For any given location:

$$D^{U}_{ze} = \left(\frac{D_{ze}}{T_e \Delta C_z}\right) = r_{ze} * v_e \tag{7}$$

an expression measuring damages for an endpoint e per person per unit change in concentration z (ug/m3; or ppm for ozone), where T is target population, *r* is the unit impact per concentration change and *v* is the unit value per impact change.

Such a measure avoids what is likely to be the most significant and tangled source of nonlinearities as well as an area subject to large modeling uncertainties -- the pollution dispersion and transformation models, leaving this for explicit analysis. This measure also has the advantage of normalizing for the population differences that can drive the analysis. The downside, of course, is that the damage function analyst is not relieved of responsibilities for estimating ambient effects if the scenario of interest involves changes in emissions.

In this spirit, we offer in Table A-2 estimates of damage by air-health pathway estimated using relationships in the Lee et al. study (Krupnick, Rowe and Lang (forthcoming) provides a more limited set of such estimates). The totals for morbidity are not additive of the individual pathway estimates because (i) appropriate aggregation over endpoints requires corrections for double-counting (say of RADs and symptom days); (ii) the individual pathway estimates are in terms of the target population while the total morbidity estimates are in terms of the entire population; and (iii) rounding. Also, all the estimates assume no thresholds in responses.

VIII. CONCLUSION

Perhaps the most important quantitative result to take from the social costing studies is that the damages from the air-health pathways are small (relative to generation costs) for fossil fuel cycles at new facilities. This is a significant conclusion because such pollutants have been the primary focus of scientific and regulatory effort for the past twenty-five years, apparently with some impact. Regulations for the ambient concentrations of these pollutants and their precursor emissions are one of the major sources of abatement costs in the U.S. economy, and a major share of such costs are borne by the electric utility industry.

Assuming that marginal damages are constant with very small changes in emissions, other things equal, existing plants, which generally emit higher quantities of pollutants per kWh produced, cause greater damages than new plants. We estimate that an "average" existing coal plant has NOx and SOx emissions about twice those of an "average" new coal plant per kWh, although the ratio of CO2 and particulate emissions is about 1.0. Other analysts (Pearce et al, 1992; Rowe, Smolinsky, and Lang, 1996) find that the ratios for NOx and SOx are larger still.

These quantitative results have their limitations and major uncertainties, of course, part of which is addressed by error propagation routines embedded in the computations models. Non-linearities in air quality modeling and uncertainties about the degree of nonlinearities in other aspects of the analyses have as yet not been carefully addressed, as the drive to simplify the computational requirements of these large models has led to much linearization. In addition, there are surely some important pathways (such as employment and fiscal effects, and global warming) that cannot yet be confidently quantified, and that are potentially quite significant. Still, the degree of consensus around health-related concentration-response and valuation functions, and the relative transferability of such functions, puts the heart of this social costing effort on reasonably solid footing.

APPENDIX

Table A-1. Comparison of Approaches

Study	Lee et al	EC	Hagler Bailly	TER
Site	Knoxville, TN	Lauffen,Germany	Capital District, New York	Metropolitan Fringe, Minnesota
Scenarios	New 1990 coal plant	New 1990 coal plant	New 1990 coal plant	New 2006 plants: Mix of coal and GCC
Fuel Cycles Upstream	Comprehensive	Comprehensive	Comprehensive	Coal and natural gas Generation only
Pathways	Comprehensive	Comprehensive	Comprehensive	Air: health, visibility, materials, crops
Local Area; Total area (Domain)	**Non-ozone**: local: 80 km radius; Domain: Eastern U.S.,1600 km radius **Ozone**: As large as 200x150km, to increment <1ppb.	**PM**: Local: 100km2 Domain: Europe 2700x4100km Grid:10,000km2 **ozone**:174x 162km, but almost all in a 150km plume W. of Lauffen.	local: 50km radius Domain: Northeastern U.S. (14 states as far west as Ohio and south as Va., plus southeastern Canada. 1300x1300km	Domain: Minnesota, W. Wisconsin, S. SD; 800x700 km Zip code/618 receptors
Plant Characteristics: Capacity Utilization rate Lifetime	- 500MW, - 75% (3286 GWh/yr). - 40 yr lifetime	- 690MW	- 300MW - 65% - 30 yr.	400MW coal plant, 4 192MW GCCs
Net emissions (g/kWh)	SO2: 1.58 NOx: 2.6 Part: 0.14	SO2: 0.8 NOx: 0.8 Part: 0.18	SO2: 1.74 NOx: 1.9 Part: 0.14	Not provided; but probably very small
Stack parameters Height Diam Temperature Velocity	150m 9.4m	240m 10m 130C	150m 4m 400K 30m/s	For 400MWplant: 137m 5.6 m 30m/s
Air modeling				
Ozone Model name Type Temporal Scavenging Population in domain	OZIPM-4: trajectory model and MAP-O3 to estimate daily peak O3 over space for ozone season. 51 actual weather days in 1990 modeled (where O3>0.08ppm). Scavenging estimated within ≈ 10km plume 7.8 million	KRAMM-DRAIS (Eulerian/ RADM): 1 day in summer modeled*75 days>120ppb. Scavenging estimated within ≈10 km long plume 9 million	Annual ave O3 extracted from EKMA diagram at >50km, based on predicted annual ave. NOx concentrations and increment. Assumption of NOx limited Scavenging within 50km Appears to be entire non-local population	Regression model. (50% of days predicted no change) NOx limited. Scavenging estimated

Conventional (Direct PM10, SO2, NO2, Pb)	ISCLT to 80 km; Statistical model fit for long-range transport for 16 wind roses to 1600 km. Abrupt reduction in average concentrations near stack.	Gaussian plume up to 100km Windrose Trajectory Model beyond. TSP estimated and converted to PM10 using 90%TSP= PM10 (others use 55%)	ISC2LT within 50km for annual ave. SCREEN2 up to 500km for # days >threshold	ISCST2 over domain. Hourly
Secondary PM10	No secondary PM10 modeled.	EMEP source-receptor matrices: SO2, SO4, NO2, NO3, Deposition	SLIM3 at >50km	No secondary
C-R Functions				
Linearization	Generally; except thresholds, semi-log	Generally; except threshold, semi-log	Generally, except semi-log; thresholds an option	Generally, except semi-log
Thresholds	With and w/o 30 ug/m3 PM10; 0.08 ppm daily ozone	0.12 ppm daily ozone	zero is default	none
Pathways Health:				
PM-Mortality Coefficient (ΔMR/10 ug/m3) Other	0.64 (Steubenville)	1.0 (Average of Daily studies) (Illustrative only)	1.0 (Average of daily studies), age adjustment 1.5 > = 65 and 0.6 < 65 yrs old	1.0 (meta analysis; larger std error than original papers) 1.3 >= 65 and 0.04 < 65
PM-acute	Hospital, emergency rooms, symptoms, RADs, chronic episodes	Most of the same functions as Lee et al.	Most of the same functions	Fewer. Add PM-chronic cases
Ozone-Mort Coefficient (%change in MR/ppb)	Coef = 0 at expected value. 90th percentile is 0.015	0.015 (illustrative only)	0.02	None
Ozone-acute	symptoms asthma RADs. Symptoms applied to adults only	Most of the same functions as Lee et al.	Same plus hospital admissions; Symptoms applied to entire population	New studies on acute symptoms and astham prevalence
Direct NOx-health	trivial	trivial	trivial	NOx-eye irritation: 36% of total health damages
Population Local Total Key types/rates	local: 866,000 total: 193 mil. Children <18 25.4% 5% asthma MR:960/100000	3.8M local. 477.4M region 9M in ozone area children<19: 19% 3.5% asthma MR: 990/100000	local: 638,000 Region 500,000 total: 93 mil. 4.3% asthma MR: 800/100,000	6.4 million total

Health Valuation				
Value Statistical life	$3.5 M	$3.25 M (2.6 MECU; $1.25=1ECU) .	$3.0 M >65 $4.0 M <65 $3.3 overall	$3.6 M
Morbidity	Literature	Same	Generally larger unit values that Lee et al and EC	Quality of well-being scale approach yields still larger unit values
Other Pathways				
Visibility	Case Study	No, unimportant	WTP=b*ln(Vnew/V initial)	LnWTP=b*(ΔV/V initial)
Materials	No; old, inconclusive studies	Inventory built;engineering costs only	Yes	Yes
Crops	Yes	Yes	Yes	Yes: O3 (15% of total damage)
Recreational Fishing	Yes for water bodies affected	Yes for water bodies affected; liming costs	More comprehensive estimates than other studies	No
Global Warming	Discussion and illustrative estimates only	Discussion and illustrative estimates only	Discussion and illustrative estimates only	Discussion only
Other Pathways	employment, roads fiscal(a), accidents in transport, occupational	employment; fiscal, noise, occupational	Roads	No
Discount rate	5%	3%	5% default?	?
Base Year	1989	1989	1992	1993
Uncertainty	Monte Carlo	Qualitative	Beta Distribution	Monte Carlo
Unique analyses	distinction between damages and externalities	Noise, forests (based on minimum standard); Materials inventory constructed.	Development of user-friendly computer model, EXMOD; mercury-health	Acute health valuation from meta-analysis tied to health index

Table A-2. Unit Damages for Air-Health Pathways.

PM10-Human Health Damage Estimates - Unit Values ($1989)				
Pollutant/Endpoint	**Annual $ per Person per ug/m^3**			
	5%	**Central**	**95%**	**Target Group**
Childhood Chronic Coughing	0.00	0.00	0.01	Children (<17) 25.4%
Adult Chronic Bronchitis	0.35	1.74	3.21	Adults (>25) 63.7%
Respiratory Hospital Admissions	0.02	0.65	1.27	All
Emergency Room Visits	0.00	0.04	0.08	All
Child Chronic Bronchitis	0.01	0.05	0.10	Children (<17) 25.4%
Restricted Activity Days	0.42	2.07	3.75	Non-asthmatics (>17) 70.97%
Asthma Attack Days	0.07	0.49	1.06	Asthmatics 5%
Respiratory Symptom Days	0.56	1.22	2.18	Adults (>17) 74.4%
Total Morbidity Damages[a]	**3.80**	**6.00**	**8.35**	**All**
Total Mortality Damages	**10.70**	**26.63**	**52.34**	**All**
Total Annual Health Damages[b]	**16.50**	**32.63**	**58.79**	**All**

a. Values in this row are not the sum of rows above because adjustments have been made to avoid double counting of endpoints and the population to which unit damages apply varies by endpoint (note fifth column).

b. The sum of mortality and morbidity damages may not sum to total damages because the latter represents percentiles of the sample of sums rather than sums of the sample percentiles. Rounding errors also affect totals.

Note: Zero threshold assumed.

Lead Human Health Damage Estimates - Unit Values ($1989)				
Pollutant/Endpoint	**Annual $ per Person per .01 ug/m^3**			
	Low	**Central**	**High**	**Target Group**
IQ Loss	0.81	1.21	1.62	Birth rate 1.3%
Hypertension	0.11	0.11	0.11	Adult Males (25-64) 25%
Coronary Heart Disease 45-54	0.03	0.03	0.03	Males (45-54) 5%
Coronary Heart Disease 55-64	0.02	0.02	0.02	Males (55-64) 4%
Total Morbidity Damages[a]	**0.96**	**1.36**	**1.77**	**All**
Neonatal Mortality	0.05	0.11	0.20	Birth rate 1.3%
Adult Male Mortality in 12 yrs 45-54	1.41	3.11	5.76	Males (45-54) 5%
Adult Male Mortality in 12 yrs 55-64	1.22	2.70	5.00	Males (55-64) 4%
Total Mortality Damages[a]	**2.68**	**5.91**	**10.95**	**All**
Total Annual Health Damages[b]	**3.97**	**7.28**	**12.30**	**All**

a. Values in this row are not the sum of rows above because adjustments have been made to avoid double counting of endpoints and the population to which unit damages apply varies by endpoint (note fifth column).

b. The sum of mortality and morbidity damages may not sum to total damages because the latter represents percentiles of the sample of sums rather than sums of the sample percentiles. Rounding errors also affect totals.

SO_2 Human Health Damage Estimates - Unit Values ($1989)				
Pollutant/Endpoint	**Annual $ per Person per ug/m^3**			
	Low	**Central**	**High**	**Target Group**
Childhood Chronic Coughing	<0.01	0.03	0.07	Children (<17) 25.4%
Adult Chest Discomfort	0.01	0.05	0.12	Adults (>17) 74.4%
Emergency Room Visits	<0.01	0.01	0.02	All
Mortality Damages[a]	1.00	7.93	17.42	All

a. Not used in Lee et al, 1995

NB: Total damages are not calculated for SO_2 endpoints.

Ozone Human Health Damage Estimates - Unit Values ($1989)				
Pollutant/Endpoint	**Annual $ per Person per .01 ppm**			
	Low	**Central**	**High**	**Target Group**
Clinical Studies[a]				
Cough Incidents	<1	<1	<1	All
Chest Discomfort	5	12	23	All
Lower Respiratory Symptoms	6	15	30	All
Upper Respiratory Symptoms	1	4	7	All
Shortness of Breath	2	10	28	All
Nose or Throat Irritation	1	6	14	All
Respiratory Symptom Days	7	14	23	All
Epidemiological Studies				
Respiratory Symptom Days	1	3	7	Adults (>17) 74.4%
Eye Irritation Days	<1	<1	<1	All
Asthma Attacks	<1	1	2	Asthmatics 5%
Minor Respiratory Related Restricted Activity Days	<1	4	9	Adults (>17) 74.4%
Total Morbidity Damages[b]	**3**	**7**	**12**	**All**
Total Annual Mortality Damages	**0**	**0**	**70**	**All**
Total Health Damages[c]	**3**	**7**	**82**	**All**

a. Not used for total morbidity damages or grand total.

b. Values in this row are not the sum of rows above because adjustments have been made to avoid double counting of endpoints and the population to which unit damages apply varies by endpoint (note fifth column).

c. The sum of mortality and morbidity damages may not sum to total damages because the latter represents percentiles of the sample of sums rather than sums of the sample percentiles. Rounding errors also affect totals.

Note: Zero threshold assumed.

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