FUEL CYCLE EXTERNALITIES: ANALYTICAL METHODS AND ISSUES

OAK RIDGE NATIONAL LABORATORY
and
RESOURCES FOR THE FUTURE

Report No. 2 on the
EXTERNAL COSTS AND BENEFITS OF FUEL CYCLES:
A Study By The
U.S. Department of Energy
And The
Commission of the European Communities

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PREFACE

The activities that produce electric power typically range from extracting and transporting a fuel, to its conversion into electric power, and finally to the disposition of residual by-products. This chain of activities is called a *fuel cycle*. A fuel cycle has emissions and other effects that result in unintended consequences. When these consequences affect third parties (i.e., those other than the producers and consumers of the fuel-cycle activity) in a way that is not reflected in the price of electricity, they are termed "hidden" social costs or *externalities*. They are the economic value of environmental, health and any other impacts, that the price of electricity does not reflect.

How do you estimate the externalities of fuel cycles? Our previous report describes a methodological framework for doing so -- called the *damage function approach*.[1] This approach consists of five steps:

1. characterize the most important fuel cycle activities and their discharges, where importance is based on the expected magnitude of their externalities,

2. estimate the changes in pollutant concentrations or other effects of those activities, by modeling the dispersion and transformation of each pollutant,

3. calculate the impacts on ecosystems, human health, and any other resources of value (such as man-made structures),

4. translate the estimates of impacts into economic terms to estimate damages and benefits, and

5. assess the extent to which these damages and benefits are externalities, not reflected in the price of electricity.

Each step requires a different set of equations, models and analysis. Analysts generally believe this to be the best approach for estimating externalities, but it has hardly been used! The reason is that it requires considerable analysis and calculation, and to this point in time, the necessary equations and models have not been assembled. Equally important, the process of identifying and estimating externalities leads to a number of complex issues that also have not been fully addressed. This document contains two types of papers that seek to fill part of

[1] Oak Ridge National Laboratory and Resources for the Future, 1992. *U.S.-EC Fuel Cycle Study: Background Document to the Approach and Issues*, Report No. ORNL/M-2500. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

this void. Some of the papers describe analytical methods that can be applied to one of the five steps of the damage function approach. The other papers discuss some of the complex issues that arise in trying to estimate externalities.

This report, the second in a series of eight reports, is part of a joint study by the U.S. Department of Energy (DOE) and the Commission of the European Communities (EC)[2] on the externalities of fuel cycles.[3] Most of the papers in this report were originally written as working papers during the initial phases of this study. The papers provide descriptions of the (non-radiological) atmospheric dispersion modeling that the study uses; reviews much of the relevant literature on ecological and health effects, and on the economic valuation of those impacts; contains several papers on some of the more complex and contentious issues in estimating externalities; and describes a method for depicting the quality of scientific information that a study uses. The analytical methods and issues that this report discusses generally pertain to more than one of the fuel cycles, though not necessarily to all of them. The report is divided into six parts, each one focusing on a different subject area.

Part I illustrates the use of the atmospheric dispersion and transformation modeling that this study recommends for airborne pollutants in the coal, biomass, oil, and natural gas fuel cycles. Paper 1, taken from a working paper by Cynthia M. McIlvaine gives a concise description of modeling the dispersion of primary pollutants that are emitted from the stack of a power plant. The modeling approach is standard, using the U.S. Environmental Protection Agency's (EPA's) Industrial Source Complex Long Term (ISCLT) model.

Paper 2, based on an unpublished paper by Meng-Dawn Cheng, describes the method that our study uses to extrapolate estimates of pollutant concentrations beyond 80 km. The method statistically extrapolates ISCLT results, with an adjustment for long-range wind patterns. While rather crude from an atmospheric science standpoint, this extrapolation is preferred to ad hoc alternatives or to limiting estimates to an 80 km range. The method that Cheng develops provides a reasonable approximation of changes in concentrations beyond 80 km, given that a more complex and data-intensive regional model is not used.

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[2] We use the abbreviation EC to refer to the Commission of the European Communities. Some time after the agreement between the U.S. and the EC to undertake this study, the European Community changed its name to the European Union.

[3] The first report in this series is referred to above and is on the study's methodological framework (ORNL/RFF 1992). The other reports are about coal, biomass, hydro, oil, natural gas, and nuclear fuel cycles. The EC is issuing a similar, but separate, series of reports.

Paper 3 is by McIlvaine, based on her dissertation. The paper details a new model developed specifically for this study, called the Mapping Area-Wide Predictions of Ozone (MAP-O_3) model. This model estimates changes in ozone concentrations that result from emissions of NO_x and other chemical species. MAP-O_3 provides estimates of ozone concentration as a function of direction (relative to the power plant or any other source) and distance. McIlvaine's model has the advantage of having spatial detail, which EPA's Ozone Isopleth Plotting Mechanism (OZIPM-4) model does not.

Part II of this volume contains Paper 4, which reviews the scientific literature on ecological impacts associated with power plant discharges. This paper is taken from an unpublished report by Larry Barnthouse, Glenn Cada, Roger Kroodsma, Dave Shriner, Virginia Tolbert, and Robb Turner. The paper describes how to quantify some of the environmental impacts. The paper also discusses many of the impacts that are potentially important, but that cannot be quantified, given the state of the science. One of the conclusions from their study is the need for considerably more research to identify exposure-response relationships that are applicable in general contexts and that describe regional landscape, rather than species-specific, impacts.

Part III contains papers summarizing the relevant health effects literature. Paper 5 is largely from a working paper by Bart Ostro. It summarizes the literature on the association between premature mortality and exposure to (non-radiological) pollutants. The paper emphasizes that, while specific results vary, the preponderance of the evidence is that exposure to particulate matter is associated with higher rates of mortality. The paper also discusses the key issues that arise in interpreting the scientific evidence, including the issue of thresholds.

Paper 6 is also based on notes by Ostro. The paper summarizes morbidity effects from exposure to pollutants. The paper is divided into a number of self-contained sections, one each on the effects of: particulate matter, sulfur dioxide, lead, and nitrogen oxide. In several of the fuel cycles, some of the externalities attributed to these effects are relatively large. The paper provides a concise summary of much of the state of the science in this important area of research.

Paper 7 is largely based on a working paper by Clay Easterly. It is on the health effects of radon on coal and uranium miners. Although not directly applicable to the two surface mines considered in our study of coal fuel cycle externalities, the methods that the paper describes are relevant for risks to miners in underground mines.

Paper 8 is also based on a working paper by Easterly. The paper summarizes the literature and the method of analysis that our study uses to calculate the impact of lead exposure on children's IQ levels.

The final paper in Part III, Paper 9, is again taken from a working paper by Easterly. It is about the effects of ozone on human health. The paper provides a concise review of the relevant literature, summarizes the key exposure-response functions, and illustrates their application to the coal fuel cycle. The methods also apply to the biomass, oil, and natural gas fuel cycles. This paper is important in that, under certain conditions, the effects of ozone on health contribute much of the externalities associated with those fuel cycles.

Part IV contains papers on methods of economic valuation. Paper 10 is based on a working paper by Alan Krupnick with research assistance from Diane DeWitt, Rebecca Holmes, and Carter Hood. It summarizes the literature on the value of a statistical life. There are many public policy situations where the decisions are economic in nature (e.g. emissions standards for various pollutants), and where the impact that is mitigated by that decision is the likelihood of premature mortality. Such situations demand some sort of valuation of life, either explicit or implicit. Of course, the basis for estimating this value is not any particular person's life, but rather small changes in the risk of premature death that individuals reveal in decisions that they make every day.

Paper 11, based on a working paper by Krupnick (with research assistance by DeWitt, Holmes and Hood), is about the benefits of reduced morbidity. The paper summarizes some theoretical models, methods for estimating these benefits, a survey of the literature, and a discussion of values for individual types of symptoms. Morbidity impacts account for a large portion of the externalities of a number of the fuel cycles, and this paper provides a concise summary of much of the literature that the U.S.-EC Fuel Cycle Externalities Study draws on.

Paper 12 is largely taken from a working paper by Krupnick (with research assistance by DeWitt and Holmes). The paper is about the benefits of visibility improvements. It contains a short review of the science of visibility measurement and the relationship between visibility and pollution. This discussion is followed by a review of the literature on visibility valuation. In some regions, particularly those with national parks or other scenic vistas, decreases in visibility are a significant externality.

Of all valuation areas in environmental economics, studies addressing the non-marketed services of recreation are by far the most prevalent. Paper 13, again taken from a working paper by Krupnick (with research assistance by DeWitt and Holmes), discusses the theory of valuing changes in recreation quality and provides a concise tabular summary of much of the literature.

Paper 14 is by Dallas Burtraw and Jonathan Shefftz. It focuses on the important issue of whether the occupational health and safety damages, that miners incur, are adequately internalized into the cost of the mineral that is mined. Some portion of the damages may be an externality. The paper provides a thorough discussion of the key aspects of this question, that make it a highly complex issue.

The last paper in Part IV, Paper 15, is by Dallas Burtraw, Ken Harrison and JoAnne Pawlowski. This paper describes a method for estimating the damages to road surfaces that result from heavy-truck traffic. These damages are just as large as many of the environmental and health effects. Interest in road damage extends beyond the energy sector, or course, and is of significant concern in regulatory issues related to truck size and weight.

Part V contains four papers on various issues related to the estimation of externalities and their use in public policy. Paper 16, which is taken from a working paper by Doug Bohi and Mike Toman, points out that there is considerable uncertainty about energy security externalities. The paper notes that although such externalities may exist in the oil fuel cycle, considering them as externalities of other fuel cycles double counts their value. The paper also strongly suggests that energy security effects may be over-stated by many analysts -- certainly a contentious position that remains unresolved. For a viewpoint different from that of Bohi and Toman, readers may wish to refer to, for example, the report by David Greene and Paul Leiby cited in Bohi and Toman's discussion.

Paper 17, based largely on a working paper by Dallas Burtraw, describes the measurement of employment benefits. This area is also another contentious issue, the key argument being whether such benefits are properly regarded as externalities. The paper provides a careful analysis that avoids the common pitfalls that many analysts fall victim to in attributing employment benefits to projects (of any type), and describes how employment benefits can be estimated. Many economists, including those commissioned by the U.S. Secretary of Energy's Advisory Board to review this analysis, are highly critical of any suggestion that there are employment externalities. Burtraw's analysis would appear to blunt much of that criticism, but the reader should decide for herself.

Paper 18 is largely based on an earlier paper by Alan Krupnick, Anil Markandya and Eric Nickell. Their paper argues that an approach different from the conventional one is relevant when estimating the external consequences of low probability-high consequence events. Specifically, they note that the conventional approach ignores risk aversion, the *ex ante* perspective in individual decision making, and lay people's assessments of risks. Taken together, these factors lead to significantly different estimates of externalities associated with the risks of nuclear accidents, catastrophic oil spills and dam breaks, and even global warming. The authors' arguments are indeed controversial and interested readers may wish also to read a critique of this paper by Jim Kahn (available as an Oak Ridge National Laboratory Central Files Memo).

Paper 19, based largely on a working paper by Dallas Burtraw, Karen Palmer and Alan Krupnick, is about another controversial issue -- specifically, what should State Public Utility Commissions (PUCs) do with estimates of externalities. Given that a State PUC wishes to use adders, the paper describes a computational method for estimating that adder. This method takes into account not only the estimated externalities, but also the piecemeal nature of regulations on other energy sectors, differences between marginal cost and electricity prices, and the opportunity for customers to switch fuels. The paper also notes that internalizing externalities can be done within the framework of existing policies and regulations.

The final part is Part VI, and it contains a paper by Anthony Schaffhauser Jr. The paper describes a system for summarizing analysts' assessments of the quality of the information that an analysis uses to estimate externalities. This system allows analysts to provide information, not only on their best estimates, but also on a range of estimates, on uncertainty, on the quality of the data, and on other factors that better reflect the full dimension of making estimates under uncertainty. The system has broad applicability beyond fuel cycle externalities, as well.

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Oak Ridge, Tennessee
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Since some of the material in this document was originally part of a large appendix in preliminary drafts of the coal report in our study, many of the people who assisted in that report provided similar support to this one. While the following list of people is rather redundant for those who have read the coal report, these people more than deserve a second mention here.

The overall study of fuel cycle externalities was largely the vision of Vito Stagliano, then at the U.S. Department of Energy (DOE), whose office funded most of this study; Pierre Valette of the Commission of the European Communities (EC), whose office funded the European study; Bob Shelton of Oak Ridge National Laboratory (ORNL), who provided overall programmatic oversight for the study; Robin Cantor, then at ORNL and now at the National Science Foundation (I took over from Robin as the Project Manager when she left for NSF); and Doug Bohi and Ray Kopp (as well as Alan Krupnick and Dallas Burtraw) of Resources for the Future (RFF). Sue Tierney, Assistant Secretary at DOE, took interest in and supported the final passage and publication of the study, with her office funding the final set of revisions.

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Last, but certainly not least, many on our team worked very long hours over many years, and their loved ones no doubt suffered a number of externalities due to our absence from home, family and friends. I wish again, as I have in the other reports in this series, to express my sincere appreciation to my family for their patience and understanding, and I am sure that my colleagues do likewise.

With so many people to thank, I have undoubtedly missed a few, and I extend my apologies to those whom I have inadvertently omitted. We are thankful for *everyone's* assistance and support, mentioned or not.

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PAPER 1[1]

ATMOSPHERIC DISPERSION MODELING OF PRIMARY POLLUTANTS FROM ELECTRIC POWER PLANTS: APPLICATION TO A COAL-FIRED POWER PLANT

1. INTRODUCTION

The normal operation of a power plant generally releases pollutants to the atmosphere. The objective of this paper is to describe a modeling method to estimate the changes in air pollutant concentrations that result from these emissions. This modeling approach is applicable to coal, biomass, oil, and natural gas technologies. As an example, this paper uses a hypothetical 500 megawatt (MW) coal-fired power plant, located at a Southeast Reference site in the U.S. and at a Southwest Reference Site. The numerical results in this paper are used in the ORNL/RFF (1994) report on coal fuel cycle externalities.

The pollutants resulting from the operation of the power plant may be classified as primary (emitted directly from the plant) or secondary (formed in the atmosphere from primary pollutants). The primary pollutants of interest in this paper are nitrogen oxides (NO_x), sulfur dioxide (SO_2), particulate matter and metals.

2. METHODOLOGY

The ground-level pollutant concentrations that could be expected to occur as the result of the operation of a 500 MW coal-fired power plant are predicted using atmospheric dispersion modeling. An atmospheric dispersion model is a set of mathematical equations used to characterize the dilution of pollutants by the wind. Some models also account for the chemical transformation of pollutants over time.

[1]Based largely on a working paper by C. M. McIlvaine.

Using stack information (i.e., stack diameter, exit gas velocity, and exit gas temperature), the model predicts the release height of pollutants to the atmosphere. Wind direction, wind speed and other meteorological measurements made in the vicinity of the stack are used to predict the dimensions (i.e., vertical and horizontal spread) of the plume and its travel path downwind. The model calculates pollutant concentrations at receptor locations which are defined by a system of grid points.

Procedures employed in the computer modeling were based on recommendations of the U.S. Environmental Protection Agency (EPA) as contained in "Guideline on Air Quality Models (Revised)" (U.S. EPA, September 1990). The EPA SCREEN model (Brode 1988) was used to predict the maximum short-term (1-hour and 24-hour average) pollutant concentrations expected to occur in the vicinity of the power plant. The EPA Industrial Source Complex Long-Term (ISCLT) model (EPA 1986) was used to predict the annual average pollutant concentrations expected to occur within an 80 kilometer (km)(50 mile) radius of the power plant.[2]

3. DESCRIPTION OF THE COMPUTER MODELS USED

3.1 ISCLT

The ISCLT model is a computer program developed by the U.S. EPA for assessing the air quality impact of emissions from a variety of sources associated with an industrial source complex. The ISCLT model is a sector-averaged model that uses statistical wind summaries to calculate annual and seasonal average ground-level concentrations. The model uses either a Cartesian or polar coordinate receptor grid. The model uses the steady-state Gaussian plume equation for a continuous source to calculate ground-level concentrations. The generalized Briggs plume-rise equations are employed by the model. The model uses two equations, one developed by Huber and Snyder, and one developed by Scire and Schulman to evaluate the effects of aerodynamic wakes and eddies formed by buildings and other structures on plume dispersion. A wind profile exponent law is used to adjust the observed mean wind speed from the measurement height to the stack height for the plume rise and concentration calculations. The model accounts for variations in terrain height over the

[2]The ISCLT model can be used to predict concentrations out to 100 km, however use of the model beyond 50 km is not recommended due to long plume transport times. Since a meso-scale model is not used as part of this study, a judgement was made to use the ISCLT results in the range 50 km-80 km as well.

receptor grid. The Pasquill-Gifford dispersion curves are used to calculate the vertical plume spread.

3.2 SCREEN

The SCREEN model uses the Gaussian plume equation to predict pollutant concentrations from a continuous source. It is assumed that the pollutants do not undergo any chemical reaction, and that no removal processes, such as wet or dry deposition, act on the plume during its transport from the source. SCREEN can perform single source, short-term calculations on point and area sources. The model gives results as predicted 1-hour average concentrations when flat or simple terrain is specified and 24-hour average concentrations for the complex terrain option (i.e. terrain higher than stack top).

The SCREEN model examines a full range of meteorological conditions to find maximum concentrations. Using full meteorology with the automated distance array, the SCREEN model provides the maximum concentration for each distance, and the highest concentration of the model run with its associated distance. The model can incorporate the effects of building downwash on plume dispersion, including the prediction of concentrations in the cavity region of a building wake. The model has the capability of predicting concentrations due to inversion breakup fumigation and shoreline fumigation.

4. DATA USED IN THE COMPUTER MODELING

4.1 SITE LOCATIONS

The Southeast Reference site is located in, what was to have been, the location of the Clinch River Breeder Reactor in Roane County, Tennessee. This location is on the north side of the Clinch River and is approximately 40 km west of Knoxville and 15 km south of Oak Ridge. The Southwest Reference site is that of the proposed, but never built, coal-fired New Mexico Generating Station (NMGS) in San Juan County, New Mexico - 55 km south of Farmington.

4.2 SOURCE CHARACTERISTICS

For the operation stage of energy production for a coal-fired power plant, there is one source of air emissions: the boiler stack. The source information needed to perform the air dispersion modeling includes the pollutant emission rate, stack height, exit gas temperature, exit gas velocity and stack tip (internal) diameter. The emissions used in the modeling are discussed in the next section.

It is assumed that the boiler is equipped with a wet lime/limestone scrubber and an electrostatic precipitator. The hypothetical coal-fired power plant was modeled with a stack height of 150 meters (m). The exhaust stack was modeled with an exit gas temperature of 325 Kelvin (52 degrees C). The exit gas flowrate was 740 actual cubic meters per second. This flowrate was input to the model as an exit gas velocity of 15 meters per second (50 feet per second) and an inside stack diameter of 7.9 meters.

4.3 EMISSIONS

Pollutant emissions used in this modeling analysis are given in Table 1. The pollutants of interest are NO_x, SO_2, total suspended particulate (TSP), PM_{10} (particulate matter with an aerodynamic diameter less than 10 micrometers), and metals, including: arsenic, cadmium, manganese, lead and selenium. NO_x and hydrocarbon emissions are employed in the ozone analysis in Part 2 of this document. Details describing the emissions estimates are given in Appendix A of ORNL/RFF (1994).

4.4 METEOROLOGY

Annual Average Concentrations

The meteorological data used in the air dispersion modeling of annual concentrations include STAR frequency summaries, ambient air temperature measurements and mixing height data.

Seasonal or annual STAR summaries are the principal meteorological inputs to the ISCLT model. A STAR summary is a tabulation of the joint frequency of occurrence of wind speed and wind direction categories, classified according to the Pasquill stability categories. The six wind speed categories used in the STAR summaries are defined as 0 to 3 knots, 4 to 6 knots, 7 to 10 knots, 11 to 16 knots, 17 to 21 knots and greater than 21 knots. The wind direction categories are the sixteen standard 22.5 degree sectors. The Pasquill stability categories are A through F.

STAR summaries for the Southeast Reference site were prepared from data collected near the site during the period 1985 to 1990 (1988 excluded). The meteorological tower is located approximately 4.5 km north of the proposed plant site. The anemometer height at the tower is 60 m.

Table 1. Controlled emission rates from the operation of the hypothetical 500 MW coal-fired power plant

	Emission rate for operation phase		
Pollutant	Tons/GWh	Tons/year	Grams/sec
Eastern Coal			
NO_x	2.90	9520	273.9
SO_2	1.74	5712	164.3
Hydrocarbon	0.06	210	6.0
TSP	0.15	485	14.0
PM_{10}	0.10	323	9.3
Western Coal			
NO_x	2.20	7236	208.2
SO_2	0.81	2660	76.5
Hydrocarbon	0.09	293	8.4
TSP	0.10	310	8.9
PM_{10}	0.06	207	6.0
Both Coals			
Arsenic	2×10^{-4}	0.66	1.9×10^{-2}
Cadmium	3×10^{-6}	0.01	2.9×10^{-4}
Manganese	1.3×10^{-4}	0.44	1.3×10^{-2}
Lead	9×10^{-5}	0.31	8.9×10^{-3}
Selenium	5×10^{-5}	0.17	4.9×10^{-3}

STAR summaries for the Southwest Reference site were prepared from data collected at Farmington, New Mexico, during the period 1954 to 1959. The anemometer height at the Farmington site was 10 meters.

Seasonal ambient temperatures from the meteorological station near the Southeast Reference site were used for the Southeast Reference site. Seasonal ambient temperatures for the Southwest Reference site were obtained from the National Weather Service Climatological Summary for Gallup, New Mexico. These temperatures were assigned to each stability category for use in the ISCLT model as recommended by the U.S. EPA (September 1990).

Mixing height data for both sites was obtained from Holtzworth's "Mixing Heights, Wind Speeds, and Potential for Urban Air Pollution Throughout the Contiguous United States", 1972. These mixing heights were assigned to the six stability classes according to EPA (September 1990). The ISCLT model assumes that there is no restriction on vertical mixing during hours with E and F stabilities (default = 10000 m).

Short-term Average Concentrations

Short-term concentrations were calculated based on various "worst-case" meteorological conditions that could theoretically occur in the area. These worst-case conditions include:

- unstable atmospheric conditions/limited mixing;
- near-neutral atmospheric conditions/high winds; and
- stable atmospheric conditions.

These conditions were simulated in the SCREEN model using a combination of wind speed and Pasquill stability categories.

4.5 DISPERSION COEFFICIENTS

Rural dispersion coefficients were used for the ISCLT and SCREEN models. The selection of rural dispersion coefficients was based on a land use typing procedure, as recommended by the U.S. EPA (September 1990), to determine whether the characteristics of an area are primarily rural or urban.

4.6 MODELING GRID

A polar coordinate receptor grid was used in the ISCLT model to calculate annual average concentrations. The polar grid is defined by 10 concentric rings every 1 kilometer out to 10 kilometers and 14 concentric rings every 5 kilometers from 15 to 80 kilometers with 36 radials in 22.5 degree increments.

For predicting short-term average concentrations, a modeling grid was constructed along a single radial. The SCREEN model was used to determine short-term concentrations at 1 kilometer increments out to 10 kilometers and every 5 kilometers from 15 to 80 kilometers from the stack.

The effects of changes in terrain elevations within the vicinity of the plant sites are not accounted for in the air dispersion modeling. This dsimplification generally results in an underestimate of pollutant concentration close to the power plant, and an over-estimate at greater distances. This pattern is especially true if terrain is high in the vicinity of the plant.

5. RESULTS

The ISCLT model was run to predict annual average concentrations expected to occur in the vicinity of the power plant at 384 receptor locations (16 directions times 24 downwind distances). The highest concentration at each downwind distance is presented here for the sake of brevity. Results for each receptor location were used to derive a population-weighted concentration, for the health effects analysis, as described in ORNL/RFF (1994).

The ISCLT model was run to predict annual average concentrations expected to occur in the vicinity of the power plant ...

The SCREEN model was run to predict the highest 1-hr average concentrations expected to occur at 24 downwind distances from the power plant. One-hour concentrations predicted with the SCREEN model were multiplied by a persistence factor of 0.4 (Brode 1988) to obtain the highest 24-hour average concentration. Both models were run with an emission rate of 1 g/s. The results from these model runs represent the annual, 1-hr and 24-hr average concentrations expected to occur from a unit emission rate. Finally, these concentrations were multiplied by the emission rates, in grams per second, of each of the pollutants of interest. The SCREEN model predicts the highest concentration at each receptor along a single radial.

5.1 UNIT CONCENTRATIONS

The highest annual average unit concentration for 24 downwind distances, at the Southeast and Southwest Reference sites are presented in Table 2. These highest concentrations are informative for comparisons against National Ambient Air Quality Standards. In the calculation of fuel cycle impacts, however, the analysis does *not* use just the highest concentrations. Rather, the analysis uses estimates of concentrations that are calculated for a spatial grid of receptor locations. In the fuel cycle report (ORNL/RFF 1994), these concentrations are used with estimates of the population to derive population-weighted estimates of the increases in concentrations due to the power plant. As a point of reference, the highest of these concentrations for the Southeast site is 0.007 micrograms per cubic meter ($\mu g/m^3$), occurring 1 kilometer from the plant. The highest of these concentrations for the Southwest site is 0.005 $\mu g/m^3$, occurring 4 kilometers from the plant.

... concentrations are used with estimates of the population to derive population-weighted estimates of the increases in concentrations ...

The highest 24-hour and highest 1-hour average unit concentrations for 24 downwind distances are presented in the second and third columns of Table 2. At both sites, the highest 24-hour average concentration is 0.38 $\mu g/m^3$ and the highest 1-hour average concentration is 0.94 $\mu g/m^3$, both occurring 1 kilometer from the plant.

Differences in annual average unit concentrations (ISCLT) between the two sites are due to different meteorological conditions at each site.

5.2 POLLUTANT CONCENTRATIONS

The maximum pollutant concentrations of total suspended particulate (TSP), PM_{10}, NO_x, SO_2 and metals, predicted to occur at 24 downwind distances from the power plant at the Southeast site are presented in Table 3. The corresponding results for the Southwest site are presented in Table 4. These concentrations were determined by multiplying the unit concentrations in Table 2 by the controlled emission rate (grams per second) in Table 1 for each of the pollutants of interest. As emphasized in the previous section, the highest concentrations serve as a point of reference. The *concentrations computed at different locations* are the values used in the calculation of impacts in the fuel cycle analyses (ORNL/RFF 1994).

Table 2. Highest unit concentrations at downwind distances from the coal-fired power plant stack at the Southeast Reference site (micrograms/cubic meter)

Downwind Distance From Stack (km)	Highest Unit Concentration		
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT
1	0.375	0.938	0.007
2	0.299	0.748	0.004
3	0.227	0.566	0.005
4	0.182	0.456	0.005
5	0.161	0.403	0.005
6	0.160	0.400	0.005
7	0.148	0.371	0.004
8	0.135	0.337	0.004
9	0.123	0.307	0.004
10	0.113	0.282	0.004
15	0.104	0.260	0.003
20	0.086	0.216	0.003
25	0.072	0.179	0.003
30	0.061	0.153	0.002
35	0.054	0.134	0.002
40	0.048	0.119	0.002
45	0.043	0.108	0.002
50	0.039	0.098	0.002
55	0.026	0.065	0.002
60	0.024	0.060	0.001
65	0.022	0.056	0.001
70	0.022	0.055	0.001
75	0.021	0.053	0.001
80	0.021	0.052	0.001

Table 2. (cont'd) Highest unit concentrations at downwind distances from the coal-fired power plant stack at the Southwest Reference site (micrograms/cubic meter)

Downwind Distance From Stack (km)	Highest Unit Concentration		
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT
1	0.375	0.938	0.003
2	0.299	0.748	0.004
3	0.227	0.566	0.005
4	0.182	0.456	0.005
5	0.161	0.403	0.005
6	0.160	0.400	0.004
7	0.148	0.371	0.004
8	0.135	0.337	0.003
9	0.123	0.307	0.003
10	0.113	0.282	0.003
15	0.104	0.260	0.002
20	0.086	0.216	0.002
25	0.072	0.179	0.001
30	0.061	0.153	0.001
35	0.054	0.134	0.001
40	0.048	0.119	0.001
45	0.043	0.108	0.001
50	0.039	0.098	0.001
55	0.026	0.065	0.001
60	0.024	0.060	0.001
65	0.022	0.056	0.001
70	0.022	0.055	0.001
75	0.021	0.053	0.001
80	0.021	0.052	0.001

The highest annual average incremental concentration of PM_{10} at the Southeast and Southwest sites is 0.061 $\mu g/m^3$ and 0.032 $\mu g/m^3$ respectively. The highest annual average incremental concentration of SO_2 for the Southeast and Southwest sites is 1.08 $\mu g/m^3$ and 0.41 $\mu g/m^3$ respectively. The highest annual average incremental concentration of NO_x is 1.8 $\mu g/m^3$ and 1.1 $\mu g/m^3$ for the Southeast and Southwest sites respectively.

Annual average concentrations of arsenic, cadmium, manganese, lead and selenium are also given in Tables 3 and 4. The highest annual average arsenic concentration at both sites is 0.0001 $\mu g/m^3$. The highest annual average cadmium concentration at both sites is less than 0.0001 $\mu g/m^3$. The highest annual average manganese concentration at both sites is 0.0001 $\mu g/m^3$. The highest annual average lead concentration at the Southeast site is 0.0001 $\mu g/m^3$. Annual average lead concentrations at the Southwest site are less than 0.0001 $\mu g/m^3$. Annual average selenium concentrations at both sites are less than 0.0001 $\mu g/m^3$.

5.3 DEPOSITION

Potential deposition impacts due to dry deposition of TSP, PM_{10}, SO_2 emissions were determined. The deposition rates were determined from the product of the predicted concentration (mass per unit volume) and the particle deposition velocity. A dry particle deposition rate of 0.02 meters per second was used (California Air Resources Board 1987). The highest annual average dry deposition rate for SO_2 was calculated to be 0.022 micrograms per square meter per second ($\mu g/m^3$-s) for the Southeast site and 0.008 $\mu g/m^3$-s for the Southwest site. The corresponding values for TSP are 0.002 $\mu g/m^3$-s and 0.001 $\mu g/m^3$-s. The highest annual average dry deposition rate for PM_{10} is 0.001 $\mu g/m^3$-s for both sites. Since dry deposition rates were so small, no estimate was made of wet deposition, which is expected to be significantly lower than dry deposition.

5.4 COMPARISON TO NAAQS

Under current Federal law, National Ambient Air Quality Standards (NAAQS) have been established for sulfur dioxide, nitrogen dioxide, lead, carbon monoxide, ozone and inhalable particles (PM_{10}). Tables 5 and 6 present a comparison of the total concentration (the sum of the incremental concentration due to the power plant plus the background concentration) and the NAAQS for PM_{10}, NO_2 and SO_2, at both sites. As shown in Tables 5 and 6, the total ambient concentration of these pollutants is below the NAAQS. (For

Table 3. Highest pollutant concentration (micrograms/cubic meter) at downwind distances from the coal-fired power plant stack at the Southeast Site

Downwind Distance From Stack (km)	Highest TSP Concentration			Annual TSP Deposition (microgm/m^2-s)	Highest PM_{10} Concentration			Annual PM_{10} Deposition (Microgm/m^2-s)
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT		24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT	
1	5.25	13.1	0.092	0.002	3.49	8.73	0.061	0.001
2	4.19	10.5	0.062	0.001	2.78	6.96	0.041	0.001
3	3.17	7.93	0.065	0.001	2.11	5.27	0.044	0.001
4	2.55	6.38	0.069	0.001	1.69	4.24	0.046	0.001
5	2.26	5.64	0.068	0.001	1.50	3.75	0.045	0.001
6	2.24	5.60	0.065	0.001	1.49	3.72	0.043	0.001
7	2.08	5.19	0.062	0.001	1.38	3.45	0.041	0.001
8	1.89	4.72	0.059	0.001	1.25	3.14	0.039	0.001
9	1.72	4.30	0.056	0.001	1.14	2.86	0.037	0.001
10	1.58	3.94	0.053	0.001	1.05	2.62	0.035	0.001
15	1.46	3.64	0.044	0.001	0.968	2.42	0.030	0.001
20	1.21	3.02	0.039	0.001	0.803	2.01	0.026	0.001
25	1.004	2.51	0.035	0.001	0.667	1.67	0.023	0.000
30	0.858	2.14	0.032	0.001	0.570	1.42	0.021	0.000
35	0.750	1.88	0.029	0.001	0.498	1.25	0.019	0.000
40	0.669	1.67	0.027	0.001	0.444	1.11	0.018	0.000
45	0.604	1.51	0.025	0.000	0.401	1.003	0.016	0.000
50	0.551	1.38	0.023	0.000	0.366	0.916	0.015	0.000
55	0.363	0.908	0.021	0.000	0.241	0.603	0.014	0.000
60	0.337	0.842	0.020	0.000	0.224	0.560	0.013	0.000
65	0.314	0.786	0.019	0.000	0.209	0.522	0.012	0.000
70	0.308	0.765	0.018	0.000	0.203	0.508	0.012	0.000
75	0.298	0.745	0.017	0.000	0.198	0.495	0.011	0.000
80	0.290	0.725	0.016	0.000	0.193	0.481	0.010	0.000

Annual Dry Deposition = Annual Concentration * 0.02 meters/second

Table 3. (cont'd)

Downwind Distance From Stack (km)	Highest TSP Concentration			Annual SO_2 Deposition (microgm/m^2-s)	Highest NO_x Concentration		
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT		24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT
1	61.7	154	1.08	0.022	103	257	1.80
2	49.2	123	0.731	0.0150	82.0	205	1.22
3	37.2	93.1	0.769	0.0150	62.1	155	1.28
4	29.9	74.9	0.804	0.016	49.9	125	1.34
5	26.5	66.2	0.797	0.016	44.2	110	1.33
6	26.3	65.7	0.765	0.015	43.8	109	1.28
7	24.4	60.9	0.727	0.015	40.6	102	1.21
8	22.2	55.4	0.689	0.014	37.0	92.4	1.15
9	20.2	50.5	0.656	0.013	33.6	84.1	1.09
10	18.5	46.3	0.627	0.013	30.8	77.1	1.04
15	17.1	42.7	0.521	0.010	28.5	71.2	0.869
20	14.2	35.5	0.458	0.009	23.7	59.1	0.764
25	11.8	29.5	0.411	0.008	19.6	49.1	0.685
30	10.1	25.2	0.374	0.007	16.8	42.0	0.623
35	8.81	22.0	0.341	0.007	14.7	36.7	0.568
40	7.85	19.6	0.313	0.008	13.1	32.7	0.523
45	7.09	17.7	0.289	0.006	11.8	29.6	0.482
50	6.47	16.2	0.268	0.005	10.8	27.0	0.447
55	4.26	10.7	0.249	0.005	7.11	17.8	0.416
60	3.95	9.89	0.233	0.005	6.59	16.5	0.389
65	3.69	9.23	0.219	0.004	6.15	15.4	0.364
70	3.59	8.98	0.205	0.004	5.99	15.0	0.342
75	3.50	8.74	0.194	0.004	5.83	14.6	0.323
80	3.40	8.50	0.183	0.004	5.67	14.2	0.305

Annual Dry Deposition = Annual Concentration * 0.02 meters/second

Table 3. (cont'd)

Downwind Distance From Stack (km)	Highest Arsenic Concentration			Highest Cadmium Concentration		
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT
1	0.0071	0.0178	0.0001	0.0001	0.0003	0.0000
2	0.0057	0.0142	0.0001	0.0001	0.0002	0.0000
3	0.0043	0.0138	00.0001	0.0001	0.0002	0.0000
4	0.0035	0.0087	0.0001	0.0001	0.0001	0.0000
5	0.0031	0.0077	0.0001	0.0000	0.0001	0.0000
6	0.0030	0.0076	0.0001	0.0000	0.0001	0.0000
7	0.0028	0.0070	0.0001	0.0000	0.0001	0.0000
8	0.0026	0.0064	0.0001	0.0000	0.0001	0.0000
9	0.0023	0.0058	0.0001	0.0000	0.0001	0.0000
10	0.0021	0.0053	0.0001	0.0000	0.0001	0.0000
15	0.0020	0.0049	0.0001	0.0000	0.0001	0.0000
20	0.0016	0.0041	0.0001	0.0000	0.0001	0.0000
25	0.0014	0.0034	0.0000	0.0000	0.0001	0.0000
30	0.0012	0.0029	0.0000	0.0000	0.0000	0.0000
35	0.0010	0.0025	0.0000	0.0000	0.0000	0.0000
40	0.0009	0.0023	0.0000	0.0000	0.0000	0.0000
45	0.0008	0.0021	0.0000	0.0000	0.0000	0.0000
50	0.0007	10.0019	0.0000	0.0000	0.0000	0.0000
55	0.0005	0.0012	0.0000	0.0000	0.0000	0.0000
60	0.0005	0.0011	0.0000	0.0000	0.0000	0.0000
65	0.0004	0.0011	0.0000	0.0000	0.0000	0.0000
70	0.0004	0.0010	0.0000	0.0000	0.0000	0.0000
75	0.0004	0.0010	0.0000	0.0000	0.0000	0.0000
80	0.0004	0.0010	0.0000	0.0000	0.0000	0.0000

Table 3. (cont'd)

Downwind Distance From Stack (km)	Highest Manganese Concentration			Highest Lead Concentration			Highest Selenium Concentration		
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT
1	0.0049	0.0122	0.0001	0.0033	0.0083	0.0001	0.0018	0.0046	0.0000
2	0.0039	0.0097	0.0001	0.0027	0.0067	0.0000	0.0015	0.0037	0.0000
3	0.0029	0.0074	0.0001	0.0020	0.0050	0.0000	0.0011	0.0028	0.0000
4	0.0024	0.0059	0.0001	0.0016	0.0041	0.0000	0.0009	0.0022	0.0000
5	0.0021	0.0052	0.0001	0.0014	0.0036	0.0000	0.0008	0.0020	0.0000
6	0.0021	0.0052	0.0001	0.0014	0.0036	0.0000	0.0008	0.0020	0.0000
7	0.0019	0.0048	0.0001	0.0013	0.0033	0.0000	0.0007	0.0018	0.0000
8	0.0018	0.0044	0.0001	0.0012	0.0030	0.0000	0.0007	0.0017	0.0000
9	0.0016	0.0040	0.0001	0.0011	0.0027	0.0000	0.0006	0.0015	0.0000
10	0.0015	0.0037	0.0001	0.0010	0.0025	0.0000	0.0006	0.0014	0.0000
15	0.0014	0.0034	0.0000	0.0009	0.0023	0.0000	0.0005	0.0013	0.0000
20	0.0011	0.0028	0.0000	0.0008	0.0019	0.0000	0.0004	0.0011	0.0000
25	0.0009	0.0023	0.0000	0.0006	0.0016	0.0000	0.0004	0.0009	0.0000
30	0.0008	0.0020	0.0000	0.0005	0.0014	0.0000	0.0003	0.0008	0.0000
35	0.0007	0.0017	0.0000	0.0005	0.0012	0.0000	0.0003	0.0007	0.0000
40	0.0006	0.0016	0.0000	0.0004	0.0011	0.0000	0.0002	0.0006	0.0000
45	0.0006	0.0014	0.0000	0.0004	0.0010	0.0000	0.0002	0.0005	0.0000
50	0.0005	10.0013	0.0000	0.0004	0.0009	0.0000	0.0002	0.0005	0.0000
55	0.0003	0.0008	0.0000	0.0002	0.0006	0.0000	0.0001	0.0003	0.0000
60	0.0003	0.0008	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000
65	0.0003	0.0007	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000
70	0.0003	0.0007	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000
75	0.0003	0.0007	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000
80	0.0003	0.0007	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000

Table 4. Highest pollutant concentration (micrograms/cubic meter) at downwind distances from the coal-fired power plant stack at the Southwest site

Downwind Distance From Stack (km)	Highest TSP Concentration			Annual TSP Deposition	Highest PM_{10} Concentration			Annual PM_{10} Deposition (microgm/m^2-s)
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT		24 hr Avg SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT	
1	3.34	8.35	0.029	0.001	2.25	5.63	0.020	0.000
2	2.66	6.66	0.032	0.001	1.80	4.49	0.021	0.000
3	2.02	5.04	0.046	0.001	1.36	3.40	0.031	0.001
4	1.62	4.05	0.048	0.001	1.09	2.73	0.032	0.001
5	1.43	3.59	0.044	0.001	0.97	2.42	0.030	0.001
6	1.42	3.56	0.040	0.001	0.96	2.40	0.027	0.001
7	1.32	3.30	0.035	0.001	0.89	2.22	0.024	0.000
8	1.20	3.00	0.031	0.001	0.81	2.02	0.021	0.000
9	1.09	2.73	0.028	0.001	0.74	1.84	0.019	0.000
10	1.00	2.51	0.025	0.000	0.68	1.69	0.017	0.000
15	0.93	2.31	0.016	0.000	0.624	1.56	0.011	0.000
20	0.77	1.92	0.014	0.000	0.518	1.30	0.009	0.000
25	0.638	1.60	0.013	0.000	0.430	1.08	0.009	0.000
30	0.545	1.36	0.012	0.000	0.368	0.92	0.008	0.000
35	0.477	1.19	0.011	0.000	0.322	0.80	0.007	0.000
40	0.425	1.08	0.010	0.000	0.287	0.72	0.007	0.000
45	0.384	0.96	0.009	0.000	0.259	0.647	0.006	0.000
50	0.351	0.88	0.009	0.000	0.236	0.591	0.008	0.000
55	0.231	0.577	0.008	0.000	0.156	0.389	0.005	0.000
60	0.214	0.536	0.007	0.000	0.144	0.361	0.005	0.000
65	0.200	0.500	0.007	0.000	0.135	0.337	0.005	0.000
70	0.195	0.486	0.007	0.000	0.131	0.328	0.004	0.000
75	0.189	0.474	0.006	0.000	0.128	0.319	0.004	0.000
80	0.184	0.461	0.008	0.000	0.124	0.311	0.004	0.000

Annual Dry Deposition = Annual Concentration * 0.02 meters/second

Table 4. (cont'd)

Downwind Distance From Stack (km)	Highest SO_2 Concentration			Annual SO_2 Deposition (microgm/m^2-s)	Highest NO_x Concentration		
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT		24 hr Avg SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT
1	28.7	71.8	0.250	0.005	78.1	195	0.681
2	22.9	57.2	0.274	0.005	62.3	156	0.745
3	17.3	43.3	0.395	0.008	47.2	118	1.07
4	13.9	34.9	0.413	0.008	37.9	94.9	1.12
5	12.3	30.8	0.382	0.008	33.6	83.9	1.04
6	12.2	30.6	0.340	0.007	33.3	83.2	0.926
7	11.3	28.4	0.300	0.006	30.9	77.2	0.817
8	10.3	25.8	0.266	0.005	28.1	70.2	0.723
9	9.40	23.5	0.237	0.005	25.6	63.9	0.645
10	8.61	21.5	0.213	0.004	23.4	58.6	0.580
15	7.96	19.9	0.141	0.003	21.7	54.2	0.384
20	6.61	16.5	0.120	0.002	18.0	45.0	0.327
25	5.49	13.7	0.110	0.002	14.9	37.3	0.299
30	4.69	11.7	0.101	0.002	12.8	31.9	0.275
35	4.10	10.3	0.093	0.002	11.2	27.9	0.252
40	3.65	9.13	0.086	0.002	9.944	24.9	0.233
45	3.30	8.25	0.079	0.002	8.986	22.5	0.215
50	3.01	7.53	0.073	0.001	8.201	20.5	0.199
55	1.99	4.96	0.068	0.001	5.40	13.5	0.186
60	1.84	4.60	0.064	0.001	5.01	12.5	0.175
65	1.72	4.30	0.060	0.001	4.68	11.7	0.164
70	1.67	4.18	0.057	0.001	4.55	11.4	0.155
75	1.63	4.07	0.054	0.001	4.43	11.1	0.146
80	1.58	3.96	0.051	0.001	4.31	10.8	0.139

Annual Dry Deposition = Annual Concentration * 0.02 meters/second

Table 4. (cont'd)

Downwind Distance From Stack (km)	Highest Arsenic Concentration			Highest Cadmium Concentration		
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT	24 hr Avg SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT
1	0.0071	0.0178	0.0001	0.0001	0.0003	0.0000
2	0.0057	0.0142	0.0001	0.0001	0.0002	0.0000
3	0.0043	0.0108	0.0001	0.0001	0.0002	0.0000
4	0.0035	0.0087	0.0001	0.0001	0.0001	0.0000
5	0.0031	0.0077	0.0001	0.0000	0.0001	0.0000
6	0.0030	0.0078	0.0001	0.0000	0.0001	0.0000
7	0.0028	0.0070	0.0001	0.0000	0.0001	0.0000
8	0.0026	0.0064	0.0001	0.0000	0.0001	0.0000
9	0.0023	0.0058	0.0001	0.0000	0.0001	0.0000
10	0.0021	0.0053	0.0001	0.0000	0.0001	0.0000
15	0.0020	0.0049	0.0000	0.0000	0.0001	0.0000
20	0.0016	0.0041	0.0000	0.0000	0.0001	0.0000
25	0.0014	0.0034	0.0000	0.0000	0.0001	0.0000
30	0.0012	0.0029	0.0000	0.0000	0.0000	0.0000
35	0.0010	0.0025	0.0000	0.0000	0.0000	0.0000
40	0.0009	0.0023	0.0000	0.0000	0.0000	0.0000
45	0.0008	0.0021	0.0000	0.0000	0.0000	0.0000
50	0.0007	0.0019	0.0000	0.0000	0.0000	0.0000
55	0.0005	0.0012	0.0000	0.0000	0.0000	0.0000
60	0.0005	0.0011	0.0000	0.0000	0.0000	0.0000
65	0.0004	0.0011	0.0000	0.0000	0.0000	0.0000
70	0.0004	0.0010	0.0000	0.0000	0.0000	0.0000
75	0.0004	0.0010	0.0000	0.0000	0.0000	0.0000
80	0.0004	0.0010	0.0000	0.0000	0.0000	0.0000

Table 4. (cont'd)

Downwind Distance From Stack (km)	Highest Manganese Concentration			Highest Lead Concentration			Highest Selenium Concentration		
	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT	24 hr Avg. SCREEN	1 hr Avg. SCREEN	Annual Avg. ISCLT
1	0.0049	0.0122	0.0000	0.0033	0.0083	0.0000	0.0018	0.0046	0.0000
2	0.0039	0.0097	0.0000	0.0027	0.0067	0.0000	0.0015	0.0037	0.0000
3	0.0029	0.0074	0.0001	0.0020	0.0050	0.0000	0.0011	0.0028	0.0000
4	0.0024	0.0059	0.0001	0.0016	0.0041	0.0000	0.0009	0.0022	0.0000
5	0.0021	0.0052	0.0001	0.0014	0.0036	0.0000	0.0008	0.0020	0.0000
6	0.0021	0.0052	0.0001	0.0014	0.0036	0.0000	0.0008	0.0020	0.0000
7	0.0019	0.0048	0.0001	0.0013	0.0033	0.0000	0.0007	0.0018	0.0000
8	0.0018	0.0044	0.0000	0.0012	0.0030	0.0000	0.0007	0.0017	0.0000
9	0.0016	0.0040	0.0000	0.0011	0.0027	0.0000	0.0006	0.0015	0.0000
10	0.0015	0.0037	0.0000	0.0010	0.0025	0.0000	0.0006	0.0014	0.0000
15	0.0014	0.0034	0.0000	0.0009	0.0023	0.0000	0.0005	0.0013	0.0000
20	0.0011	0.0028	0.0000	0.0008	0.0019	0.0000	0.0004	0.0011	0.0000
25	0.0009	0.0023	0.0000	0.0006	0.0016	0.0000	0.0004	0.0009	0.0000
30	0.0008	0.0020	0.0000	0.0005	0.0014	0.0000	0.0003	0.0008	0.0000
35	0.0007	0.0017	0.0000	0.0005	0.0012	0.0000	0.0003	0.0007	0.0000
40	0.0006	0.0016	0.0000	0.0004	0.0011	0.0000	0.0002	0.0006	0.0000
45	0.0006	0.0014	0.0000	0.0004	0.0010	0.0000	0.0002	0.0005	0.0000
50	0.0005	10.0013	0.0000	0.0004	0.0009	0.0000	0.0002	0.0005	0.0000
55	0.0003	0.0008	0.0000	0.0002	0.0006	0.0000	0.0001	0.0003	0.0000
60	0.0003	0.0008	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000
65	0.0003	0.0007	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000
70	0.0003	0.0007	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000
75	0.0003	0.0007	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000
80	0.0003	0.0007	0.0000	0.0002	0.0005	0.0000	0.0001	0.0003	0.0000

regulatory purposes the highest, second highest receptor concentration is added to the background concentration and compared to the NAAQS).

The quarterly average NAAQS for lead is 1.5 $\mu g/m^3$. The highest quarterly average incremental concentration of lead, due to the power plant, is 0.0001 $\mu g/m^3$ at both sites. The background concentration of lead was not available at either site.

Table 5. Summary of modeling results and monitoring data for a coal-fired boiler located at the Southeast Reference site (micrograms per cubic meter)

	Particulate		PM_{10}		NO_x	SO_2	
Highest Incremental Impact of the Facility	5.3	0.092	3.5	0.061	1.8	62	1.08
Background Concentration*	108	47	71	37	23	76	25
Total Concentration	113	47	74	37	25	138	26
Primary NAAQS**	None	None	150	50	100	365	80

*From 1990 EPA AIRS database McMinn Co., TN monitoring station (Site I.D. 47-107-0101); 2nd highest 24 hour average and annual mean.

**For regulatory purposes the highest second highest receptor concentration is added to the baseline concentration and compared to the National Ambient Air Quality Standard (NAAQS).

Table 6. Summary of modeling results and monitoring data for a coal-fired boiler located at the Southwest Reference site (micrograms per cubic meter)

	Particulate		PM_{10}		NO_x	SO_2	
	24 hour	Annual	24 hour	Annual	Annual	24 hour	Annual
Highest Incremental Impact of the Facility	3.3	0.048	2.3	0.032	1.12	29	0.41
Background Concentration*	66	42?	64	24	15	93	14
Total Concentration	69	42	66	24	16	122	14
Primary NAAQS**	None	None	150	50	100	365	80

*From 1991 EPA AIRS database San Juan Co., NM monitoring stations; 2nd highest 24 hour average and annual mean concentrations.

**For regulatory purposes the highest second highest receptor concentration is added to the baseline concentration and compared to the National Ambient Air Quality Standard (NAAQS).

?Indicates that the mean does not satisfy AIRS summary criteria.

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PAPER 2[1]

ESTIMATING POLLUTANT CONCENTRATIONS BEYOND 80 KM FROM A COAL POWER PLANT STACK

1. INTRODUCTION

Concentrations of criteria pollutants, i.e., SO_2, NO_2, TSP, and PM_{10} were estimated at various downwind distances from the stack (refer to Paper No. 1). The distances ranged from 1 to 10 km (at 1 km intervals), and 11 to 80 km (at 5 km intervals). This modeling was limited to 50 km, and then to 80 km, as an upper bound of reasonable results using the Environmental Protection Agency's approved Industrial Source Complex-Long Term (EPA ISCLT) model. Ambient concentrations at distances greater than 80 km from the stack were desired at a number of selected points, i.e., 99.2, 112, 160, 240, 320, 560, 800, and 1600 km. These concentrations are used in the subsequent health impact analysis as part of the U.S.-EC fuel cycle study.

The initial prediction was performed using an emission value of 1 gram per second (g/s). Ambient concentrations for SO_2, NO_2, TSP, and PM_{10} resulting from the coal-fired power plant emissions were then estimated by multiplying the concentrations derived in Paper No. 1 by the corresponding emission rates in g/s. Table 1 shows the emission rate values used. Seasonal and annual average concentrations were computed for distances ranging from 1 to 80 km from the stack for 16 sectors on a polar grid with a 22.5° of angular increment. The spatial and temporal resolutions in the analyses were limited to the 16 directions and 4 seasons, respectively. Since the health impact analysis does not require air quality data on a seasonal basis and the yearly average concentrations appear to be adequate, the annual averages were used. ISCLT computed the annual average as the arithmetic average of the four seasonal averages.

[1]Based largely on a working paper by M. D. Cheng

2. BRIEF DESCRIPTION OF THE ISCLT MODEL

The ISCLT model uses joint frequency distributions of wind speed and wind direction by stability category as the main meteorological input. These are sometimes called "STAR" summaries (for STability ARray). These summaries can include frequency distributions over a monthly, seasonal, or annual basis. Other meteorological inputs include, for instance, average temperatures, average mixing heights, anemometer height, and a number of source and receptor parameters. The ISCLT model uses a Gaussian plume equation as the basis for calculating pollutant concentrations on a long-term basis. Since the wind direction input is the frequency of occurrence over a sector, with no information on the distribution of winds within the sector, the

Table 1. Emission rates (g/s) used in ISCLT modeling

Pollutant	Southeast (SE)	Southwest (SW)
SO_2	164.3	76.5
NO_2	273.9	208.2
TSP	14.0	8.9
PM_{10}	9.3	6.0

Note: The southeast site is at Oak Ridge, Tennessee, and the southwest site is at Farmington, New Mexico.

concentrations are sector-averaged. Seasonal or annual emissions from the source are partitioned among the sectors according to the frequencies of winds blowing toward the sectors. For further details, readers are referred to the user's guide for the ISC Model (EPA-450/4-88-002a, December 1987). There is a newer (1992) version of ISCLT available from EPA; however, the previous version of the model was used as the more recent version was not available when the analysis was initially done. Analysis indicated that results were identical when both models were run with the same input data (Cindy McIlvaine, September 1993, personal communication).

3. CLIMATOLOGY

This section provides the climatological background for the southeast and the southwest sites. A type of graph known as a wind rose is particularly useful in air-dispersion climatology. A wind rose is a diagram in which the frequency of winds from each compass direction is indicated by the length of a bar extending from the center of the diagram towards the direction from which the corresponding winds originate. Each bar is divided into several segments, each segment having a characteristic width and shading, to represent various wind speed categories. Wind speeds are conventionally given in knots; a knot is approximately 0.5 m/s.

Figure 1 shows a wind rose for the southeast (Oak Ridge) site from 1985 to 1990 without 1988. Most of the operations at the facility which provided the wind data were terminated in 1988, and the data from that year were sparse and not quality-assured (R. Sharp, Computer Application Division, ORNL, personal communication, September 23, 1993).

For the Oak Ridge site, prevailing winds from NE (45°), ENE (67.5°), SW (225°), and WSW (247.5°) directions could be found from 1985 to 1990. These represent approximately 10%, 12%, 13%, and 10% of the times that winds arrive at the site, respectively. The remaining 55% of the time the winds came from other directions, among which NW (315°), WNW (292.5°), W (270°), and SSW (202.5°) were most frequent.

Figure 2 shows the wind rose plot for the southwest site (i.e., at Farmington, NM) from 1954 to 1959. A wind rose plot using data from a different period such as 1985-1990 may have slightly different frequency distributions. The major wind direction was from the ENE, indicating pollutants could be transported with a relatively high frequency toward the WSW direction.

4. DESCRIPTION OF EXTRAPOLATING METHODOLOGY

Transport of pollutants to a distance greater than 80 km from a stack is a long-range transport (LRT) problem in atmospheric dispersion. There are processes important to LRT but less significant to short-range dispersion problems that include atmospheric deposition (wet and dry), chemical transformation, and boundary-layer effects induced by terrain. In contrast, the ISCLT model was designed for estimating concentrations at a short distance (less than 50 km) from the stack. Concentrations estimated by a model when these effects are taken into account could be rather different from those estimated by another model when these effects are not taken into account. However, there are no dispersion programs that are recommended by the

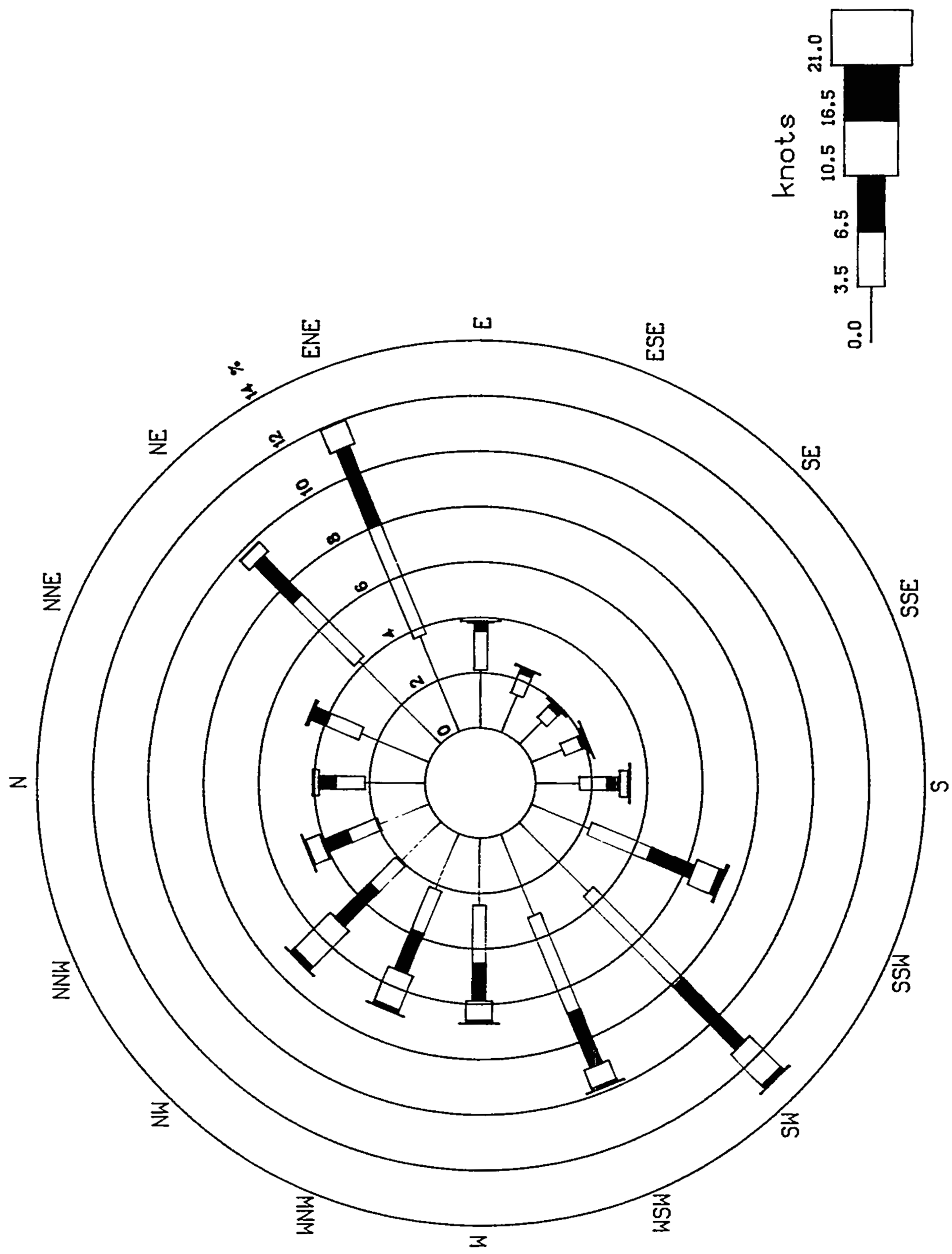

Fig. 1. WIND ROSE for Tower K (@60m), 1985-1990 (w/o 1988)

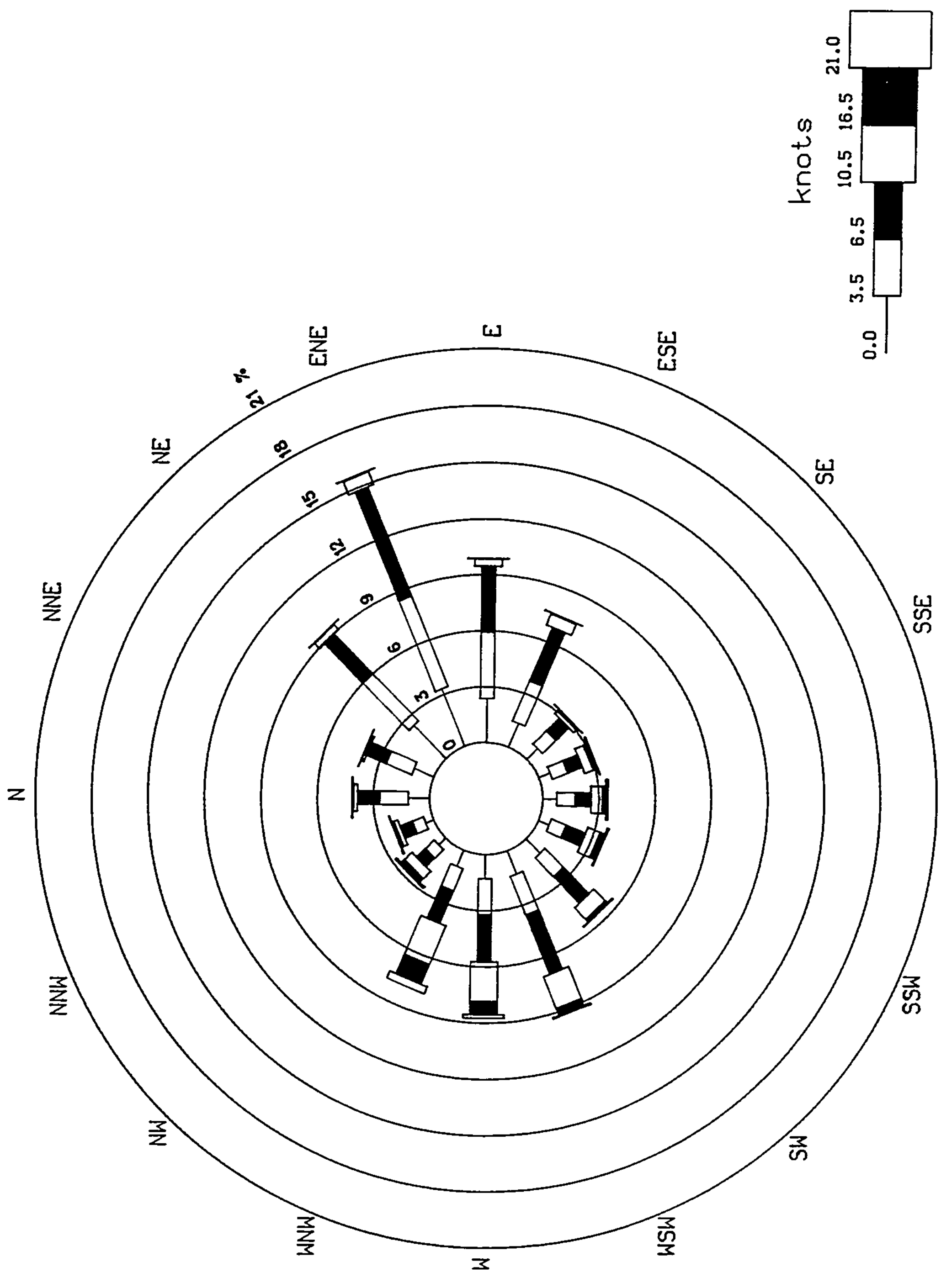

Fig. 2. WIND ROSE for Farmington for 1954 - 1959

Environmental Protection Agency (EPA) to use to estimate pollutant concentrations at distances greater than 50 km from the stack (Federal Register 1993). Existing data bases from field studies at meso-scale and long range transport distances are limited in detail; thus, LRT models (e.g., Regional Oxidant Model (Lamb 1983), Regional Acid Deposition Model (Chang et al. 1987) and MESOPUFF (Scire et al. 1984)], which have been submitted to EPA, have not been subjected to rigorous verification.

In theory, one would like to use one of the existing LRT models for the circulation of ambient pollutant concentrations. However, this more detailed approach was beyond the scope of this study. Instead, to approximate concentrations at longer distances from the stack, we chose to use a statistical calibration approach for estimating concentrations at distances greater than 80 km.[2] A judgment was made to use the ISCLT Model results in the range from 50 - 80 km, as this distance is still a relatively short-range phenomenon.

Calibration models were developed based on the annual average concentrations for the 16 directions. For a given pollutant in each sector, a calibration equation was derived by regressing the prorated concentrations for distances from 15 to 80 km. The concentrations within the 15 km range from the stack were not included in the calibration procedure. This is because within this range concentration curves are complicated to fit with any simple function.

After 15 km, concentrations in a given sector follow a monotonically decreasing trend. This trend was observed for all sectors at both the southeast and southwest sites, as shown in the two three-dimensional (3D) plots in Figs. 3 and 4. Concentrations in these plots are the ISCLT predictions prior to multiplication by the emission rates. Since the emission rate for a pollutant is constant, each of the two figures can be considered as the representative concentration surface for pollutants at one site. Concentrations for different pollutants are expected to be different because of different emission rates. However, the shape of the concentration surface will be the same.

A negative exponential (with the square root of distance) model appears to fit the concentration data reasonably well (with R^2, the coefficient of determination, > 0.99):

$$ln[C] = a * e^{[b \bullet \sqrt{D}]} \tag{1}$$

[2]A worthwhile topic for future research is to compare results from our approximating extrapolation methodology with those from an LTR model(s).

where C is the pollutant concentration in $\mu g/m^3$, D is the downwind distance from the stack in km, a and b are the model parameters that produced the best-fit model to the data, and e is the exponential operator.

A negative exponential (with the square root of distance) model appears to fit the concentration data reasonably well ...

More than 10 mathematical forms such as C=a+b*D, ln[C]=a+b*D, etc. were tested, but Eq. (1) yielded the best fit in terms of R^2 in the model development. One calibration equation of the above form was developed for each of the four pollutants for each of the 16 sectors (i.e., 64 equations for each site). Table 2 shows the estimated values for a and b for each sector and pollutant, for both sites.

5. KEY-HOLE EFFECTS

There may be conditions by which pollutants travelling toward the westerly directions would reverse, leading to a concentration contour in the shape of a key hole. Concentrations of pollutants from a particular stack would be zero beyond a specified distance to the west, because the winds would cease to continue in that direction. Winds flowing across the continental United States have a general tendency to travel from west to east. For the eastern site, which is confined by the Cumberland Plateau to the west and by the Appalachian mountains to the east, winds coming from the northeast direction would be channeled toward the southwest and vice versa. Wind roses for near the surface and for 500 m above the surface at Knoxville and Chattanooga (which are near the site) show that winds from the NE near Knoxville tend to shift so as to be from the north at Chattanooga. Taking the large-scale wind pattern (prevailing westerlies) and local topography into account, it is likely that pollutants would only travel for a limited distance in the westerly directions. This raises the question of how far that limited distance is likely to be.

It is difficult to address that question appropriately without employing a meso-scale or long-range transport model. The ISCLT model was not designed for modeling pollutant dispersion at long range (i.e., at distances greater than 50 km from the stack) nor was it designed for complex terrain. In order to account for the wind reversal phenomenon using existing ISCLT predictions, we devised

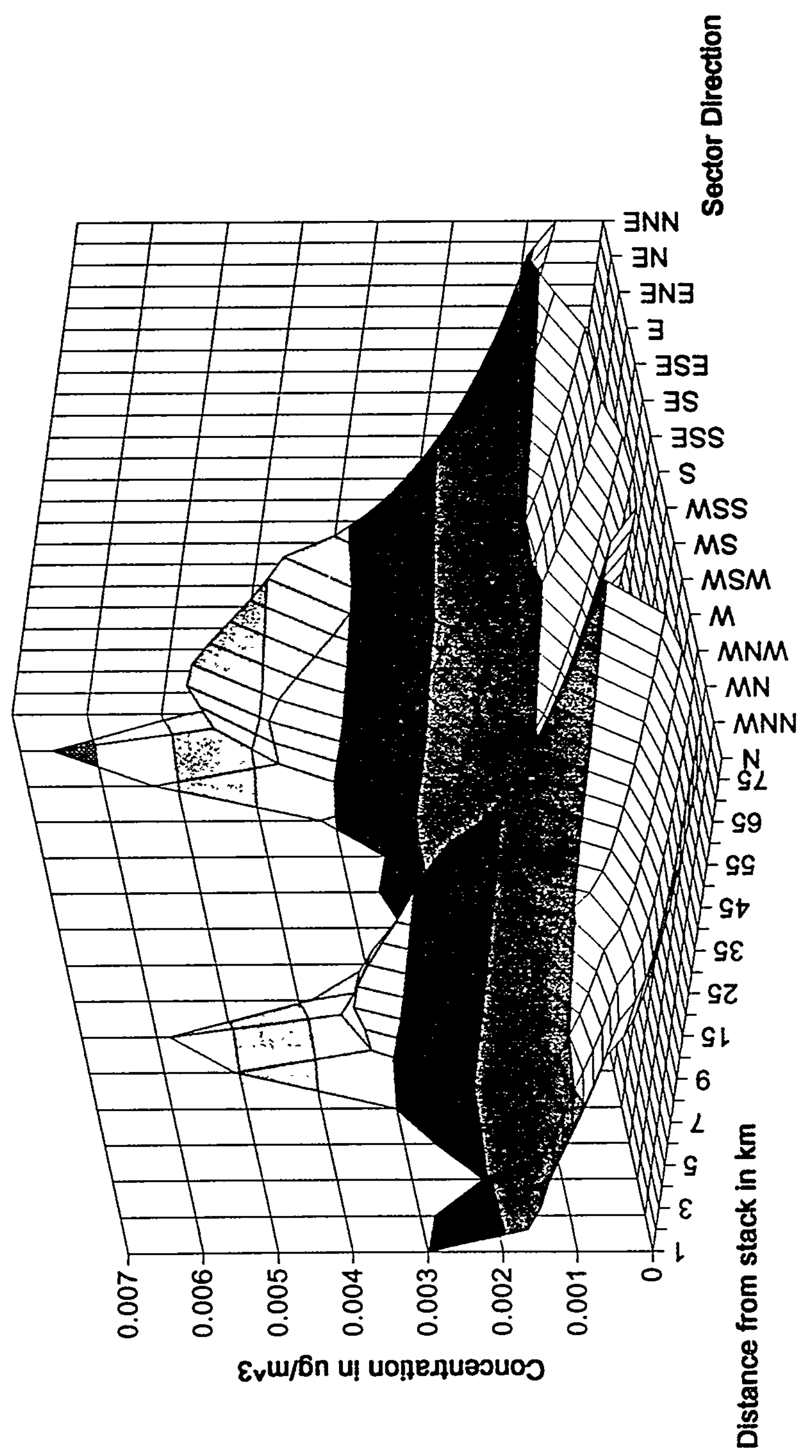

Fig. 3. Concentration of unit emission from stack at the Southeast Site

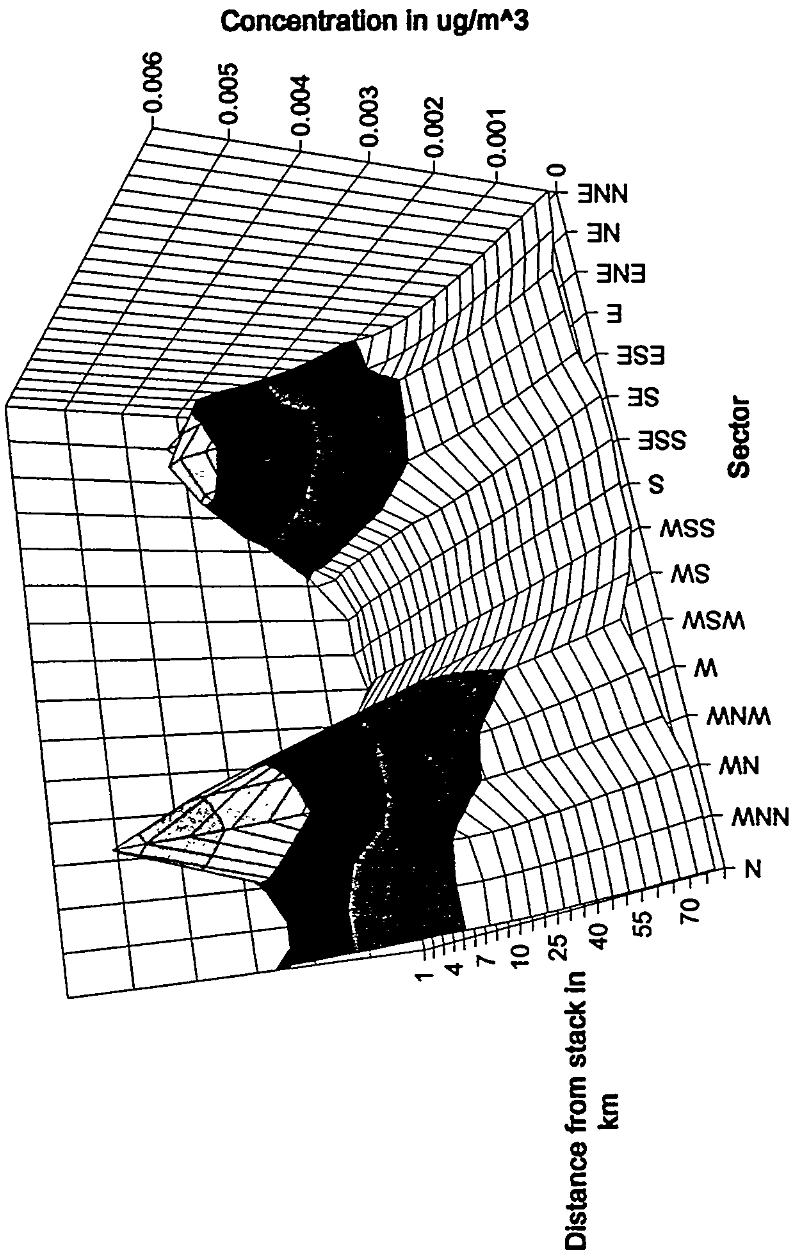

Fig. 4. Concentration of unit emission from stack at the Southwest Site

a short-cut methodology to model the process. The method assumes that a pollutant traveling toward the west will be reflected back to the east at a certain distance (δ) from the stack. Selection of δ was made on a semi-subjective basis as explained below.

Reflected concentrations were modeled as if there were a virtual stack located 320 km upwind of the real stack. This concept is shown in Fig. 5, in which the upwind direction is taken as west. The total concentration at various distances within a given sector (to the east in Fig. 5), after reflection is taken into account, is the sum of the original ISCLT model predictions and the reflected concentrations. Two δ values, one at 80 km and the other at 160 km, were chosen. The 80 km range was selected because this was the distance of initial interest to the project, which was to predict concentration at distances greater than 80 km. Initial ISCLT concentrations were also available up to this distance. Concentrations greater than 80 km have to be extrapolated from the initial ISCLT predictions. However, since Chattanooga is about 160 km from Oak Ridge, this distance was selected as the maximum westward distance that any easterly winds would proceed before reversing. In other words, pollutants would be completely reflected back to the east when they had traveled 160 km west. Reflected concentrations would then be added to the original ISCLT predictions to obtain the total concentrations at various distances from the stack. A similar approach and assumptions was used for the southwest site.

The method assumes that a pollutant traveling toward the west will be reflected back to the east at a certain distance (δ) from the stack.

In short, winds initially traveling in some direction, θ, with an easterly component were assumed to follow the initial (ISCLT) dispersion of pollutants in that direction, beyond δ, without any concentration reversal. For winds having an initial direction, θ, with a westerly component, the centerline direction of the reversed winds was assumed to be diametrically opposite to (i.e., rotated 180 degrees from) the direction of the initial wind. The reflected pollutants, traveling in the $180° + \theta$ direction, were assumed to follow theGaussian dispersion process that was modeled by ISCLT for the virtual stack. This methodology, however, does not account for the possibility that the reflected wind direction could be something other than $180° + \theta$. However, extrapolating ISCLT results beyond 50 km range has stretched the model's capability, and therefore further modifications will not necessarily constitute improvements.

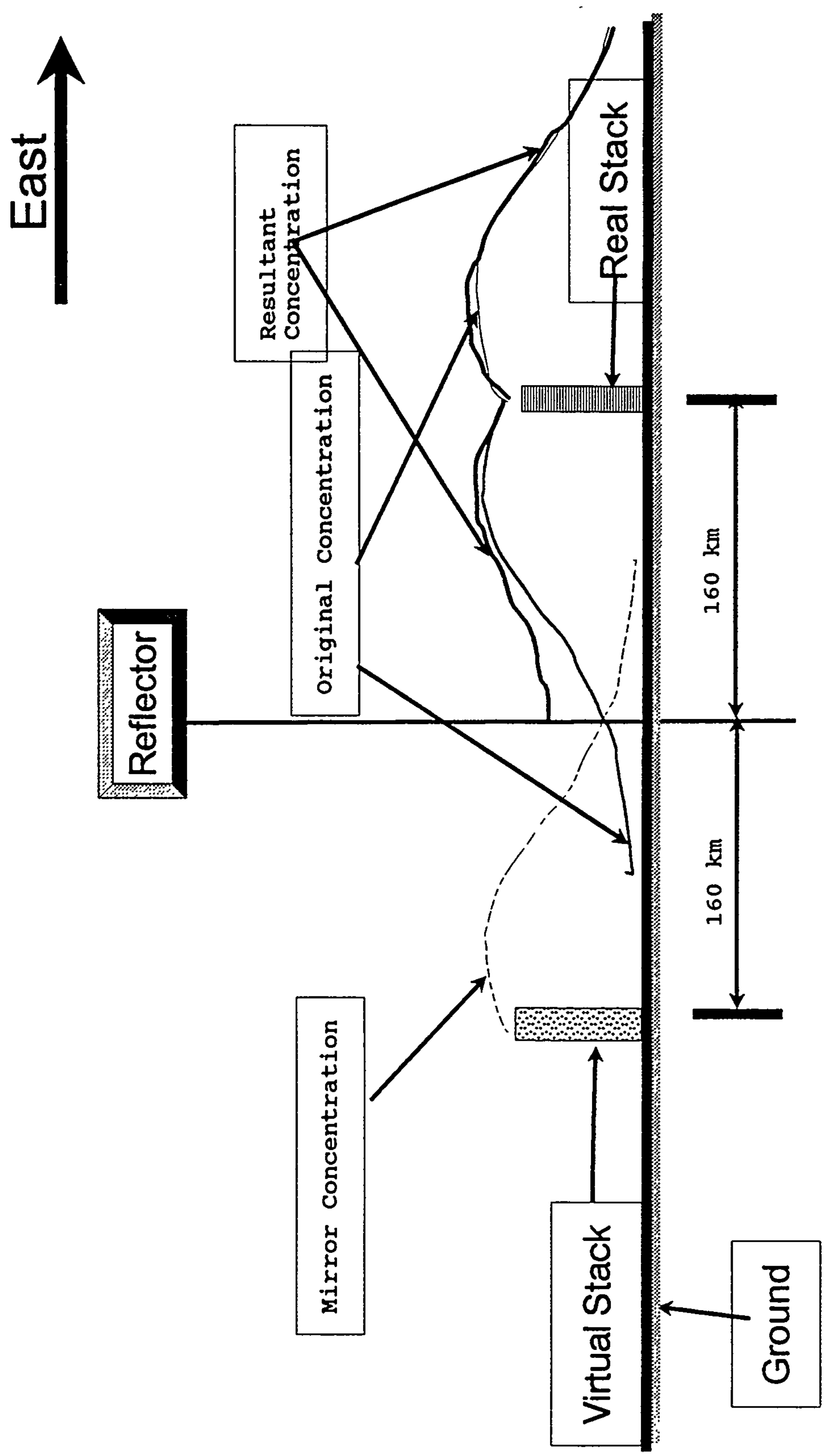

Fig. 5. Illustration of concentration reflection concept to account for the key-hole effect

6. UNEXPECTED PATTERN IN PREDICTED CONCENTRATION NEAR THE STACK FOR SE SITE

For the southeast site, the concentration value decreased abruptly between 1 to 2 km from the stack for all sectors (see Figs. 6 - 21). The dramatic decrease in concentration within such a short distance (1 km) is *unexpected* for annual averages, because the meteorological conditions (e.g., stability category A of the Pasquill classification scheme) that are the most likely cause of such a sharp concentration gradient occur only during a small fraction of the year.

Table 3 summarizes the joint frequencies of occurrence of wind speed and wind direction by stability category for each season for both sites. The symbol in the first column denotes the season and the stability category, e.g., 1-B stands for the season 1 (winter) and stability category B, while 2-C stands for the season 2 (spring) and stability category C, etc. Stability category D, in fact, occurred most frequently in all seasons at both sites. Stability category A, on average, accounted for about 7.5% and 2.5% of the total frequencies at the southeast and the southwest sites, respectively. Whether the frequency of the stability A caused the concentration "drop" between 1 and 2 km at the southeast site is unclear at this point. Such abrupt changes in concentrations within a short distance of the stack were not generally conspicuous in the model results for the southwest site.

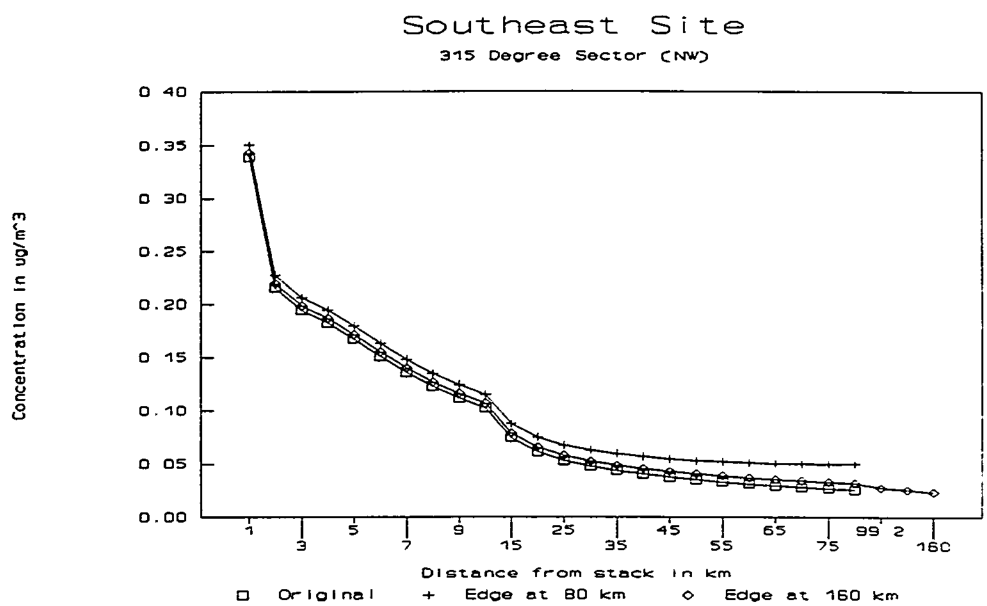

Fig. 6. Southeast Site 315 degree sector (NW)

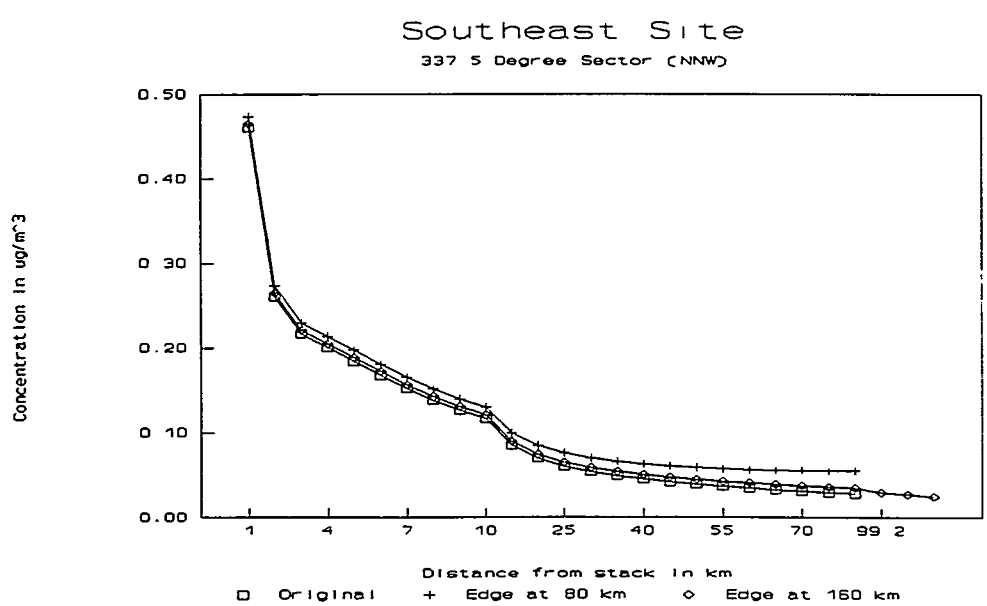

Fig. 7. Southeast Site 337.5 degree sector (NNW)

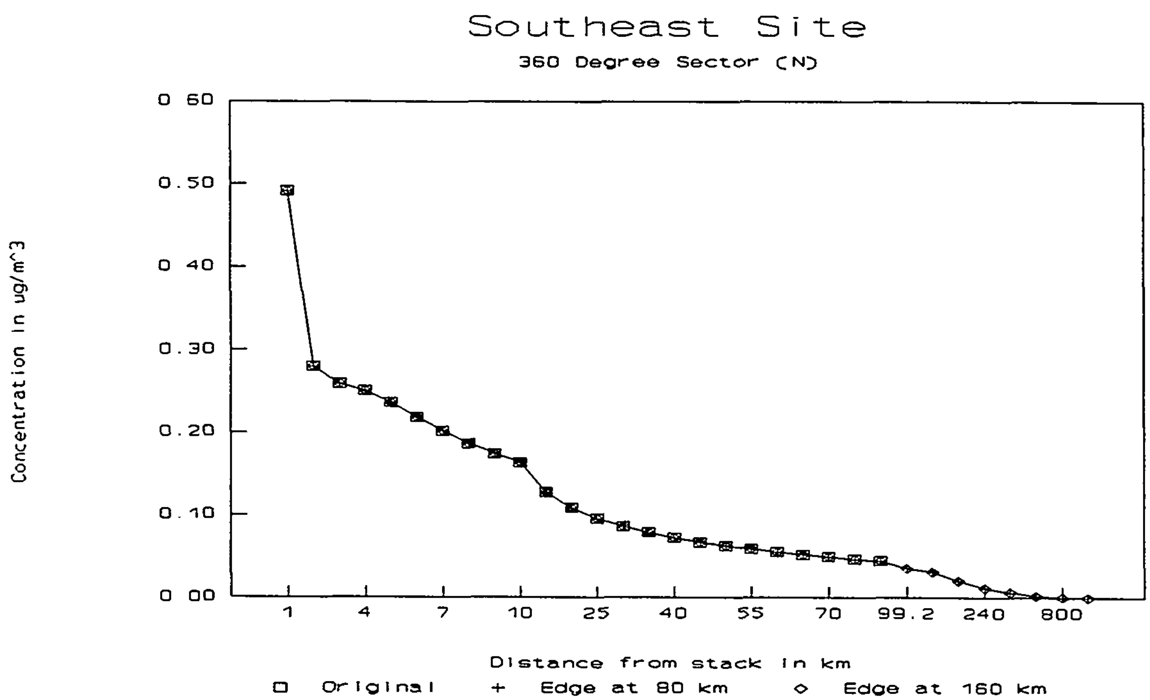

Fig. 8. Southeast Site 360 degree sector (N)

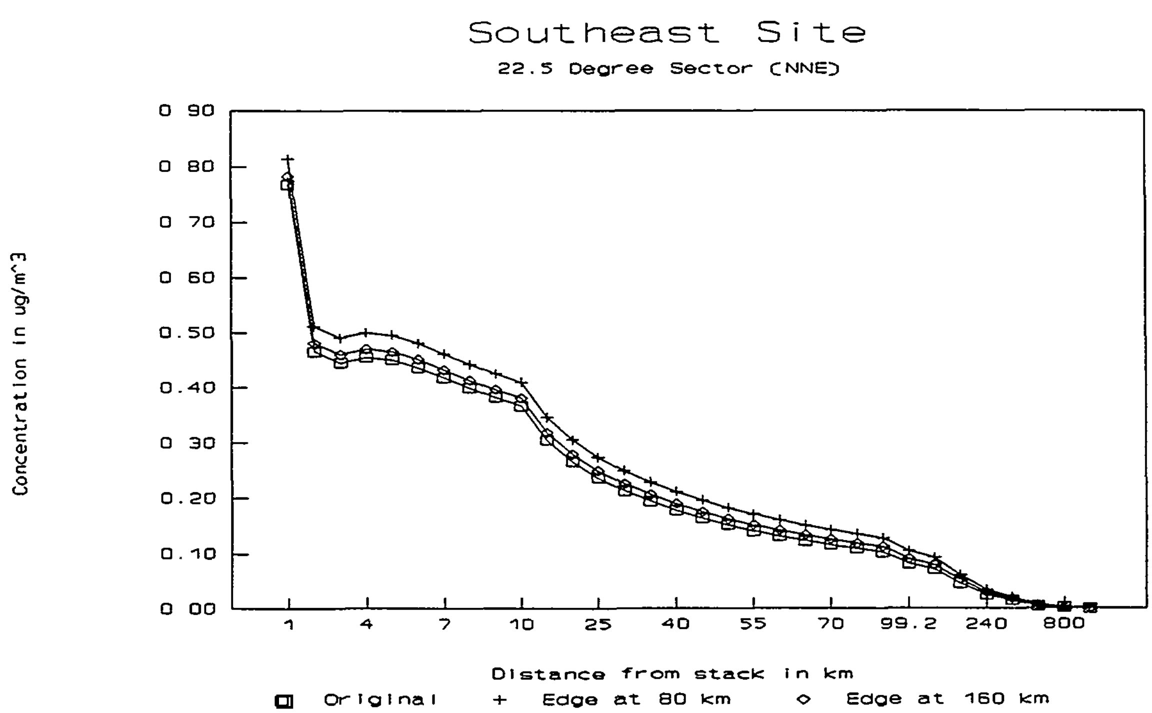

Fig. 9. Southeast Site 22.5 degree sector (NNE)

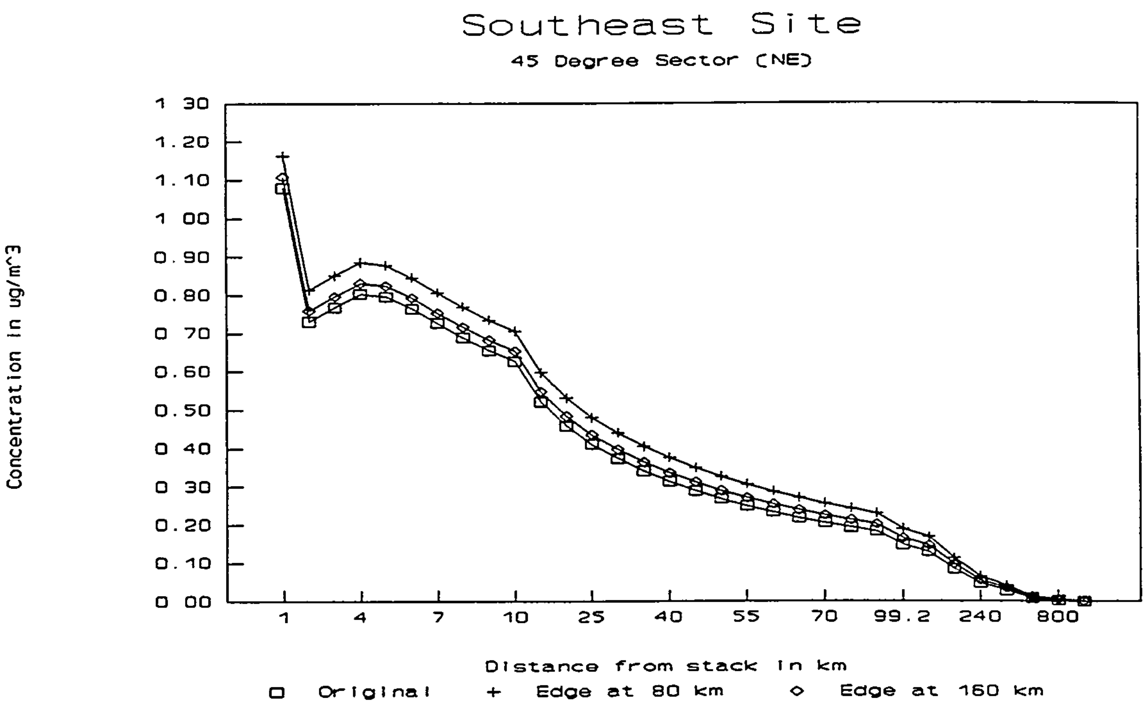

Fig. 10. Southeast Site 45 degree sector (NE)

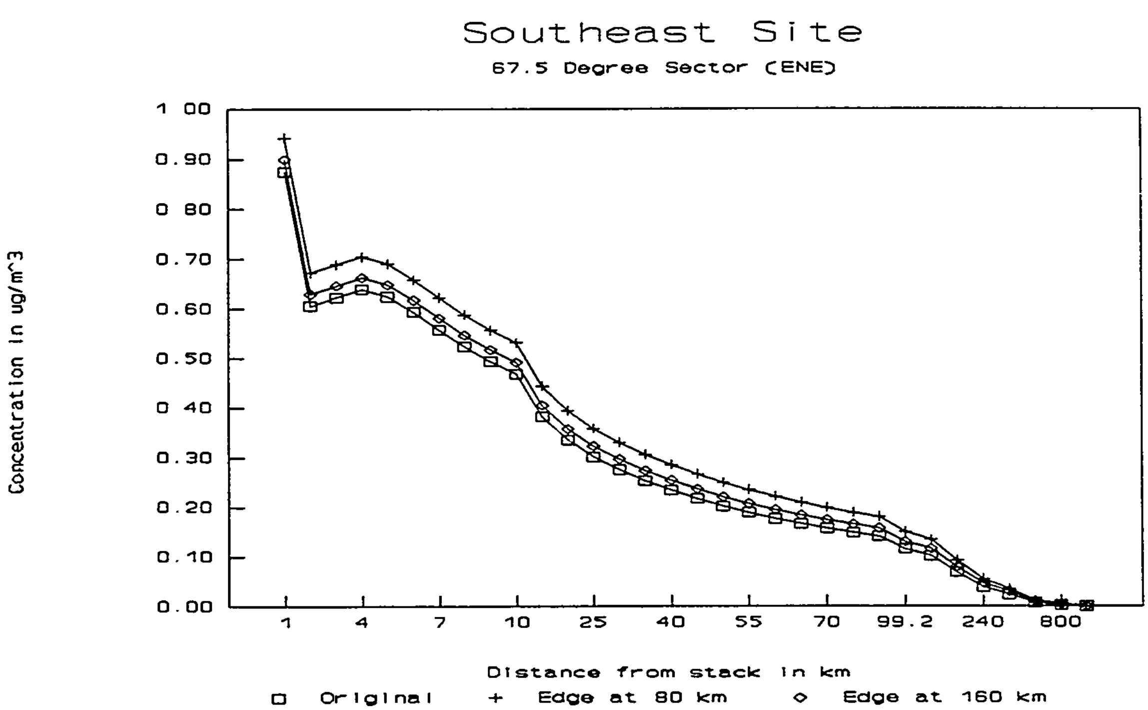

Fig. 11. Southeast Site 67.5 degree sector (ENE)

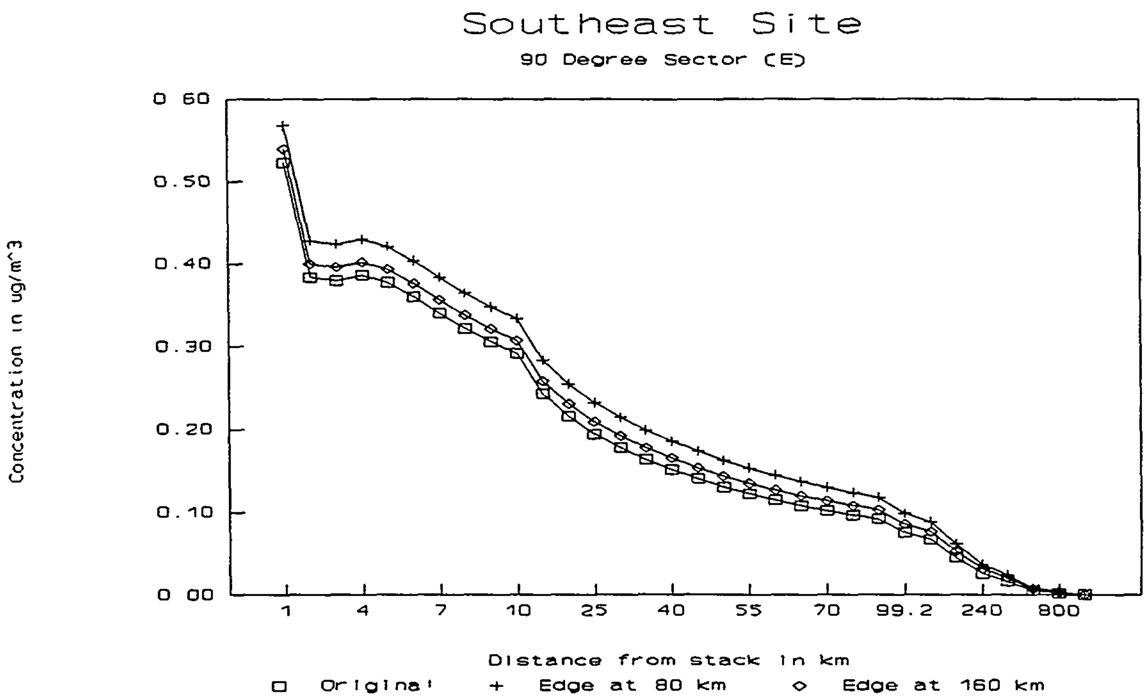

Fig. 12. Southeast Site 90 degree sector (E)

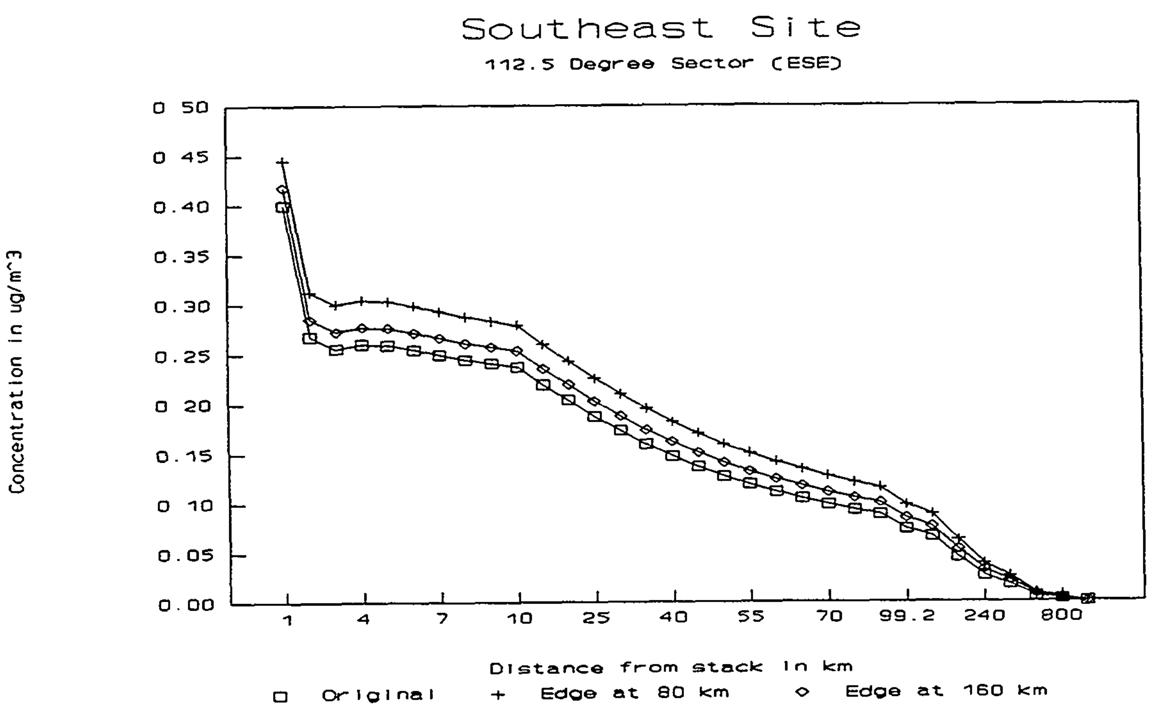

Fig. 13. Southeast Site 112.5 degree sector (ESE)

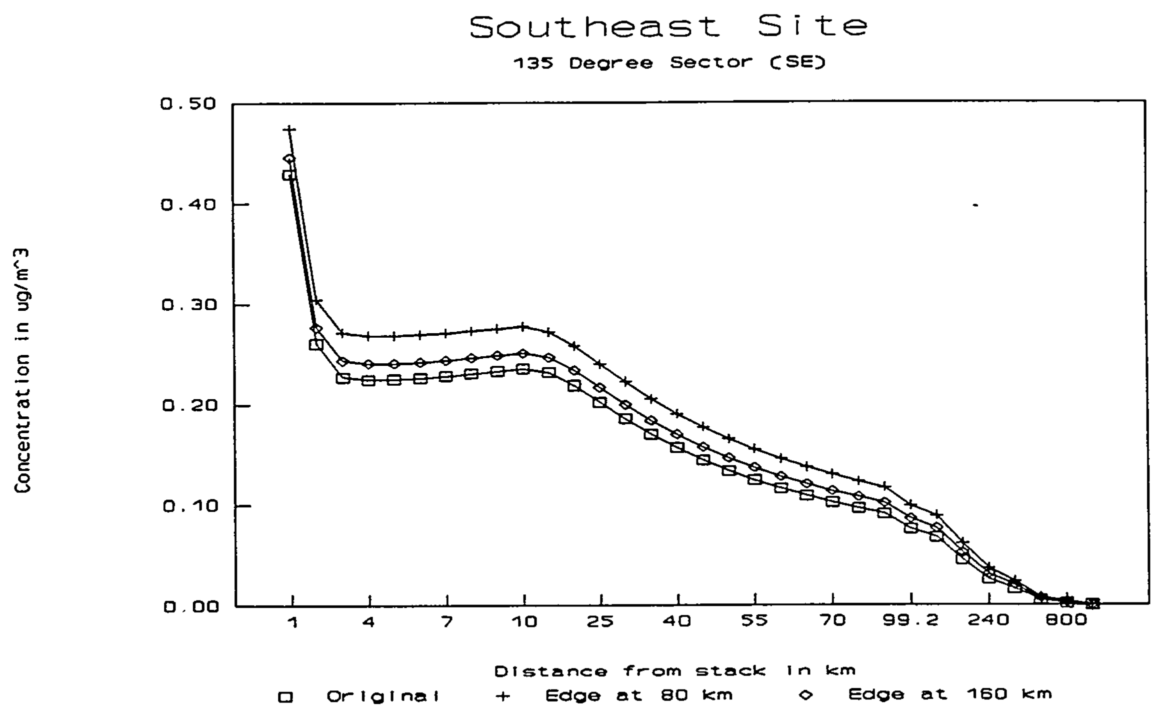

Fig. 14. Southeast Site 135 degree sector (SE)

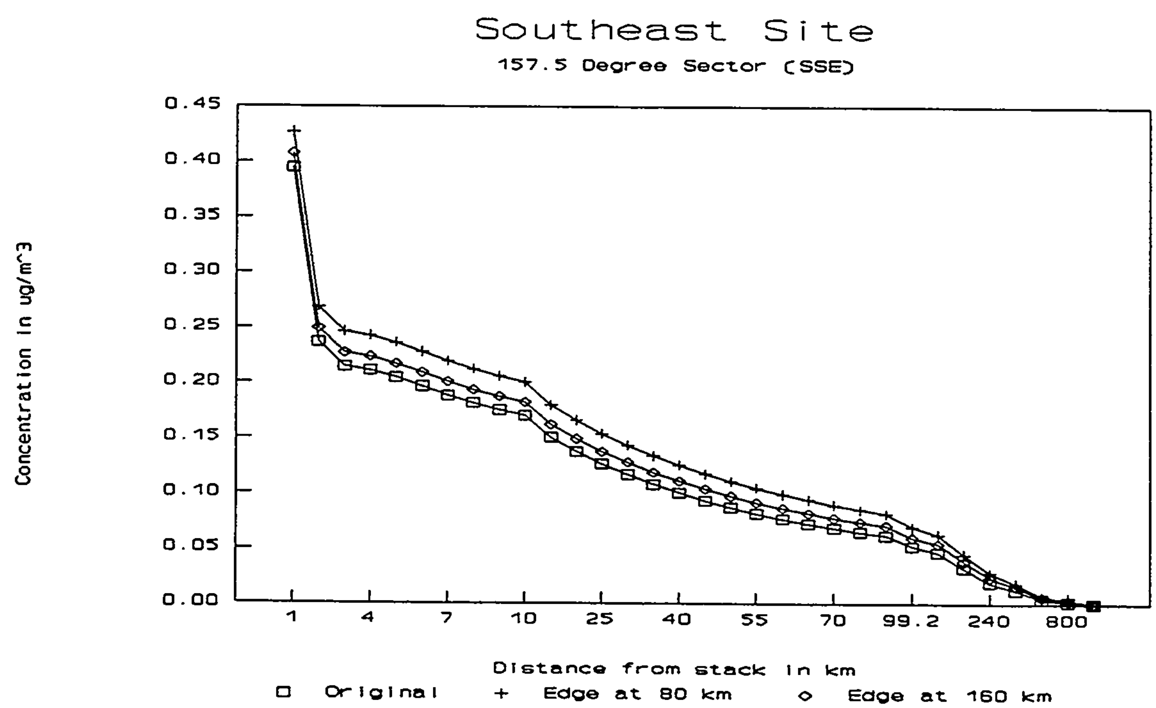

Fig. 15. Southeast Site 157.5 degree sector (SSE)

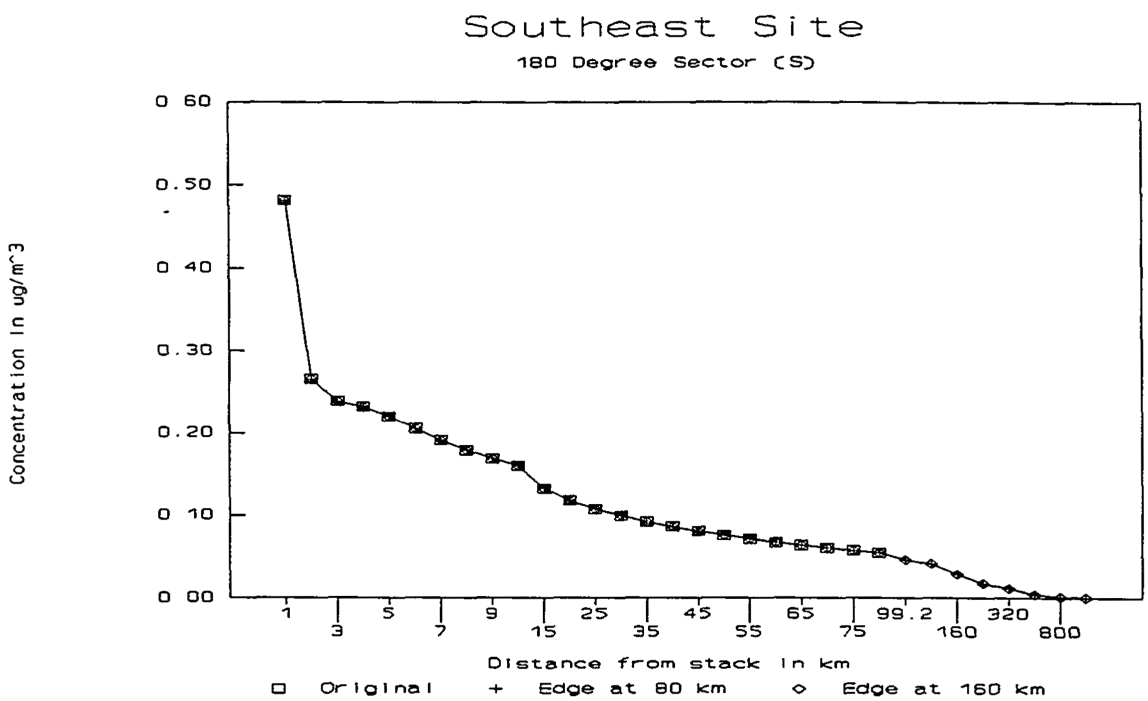

Fig. 16. Southeast Site 180 degree sector (S)

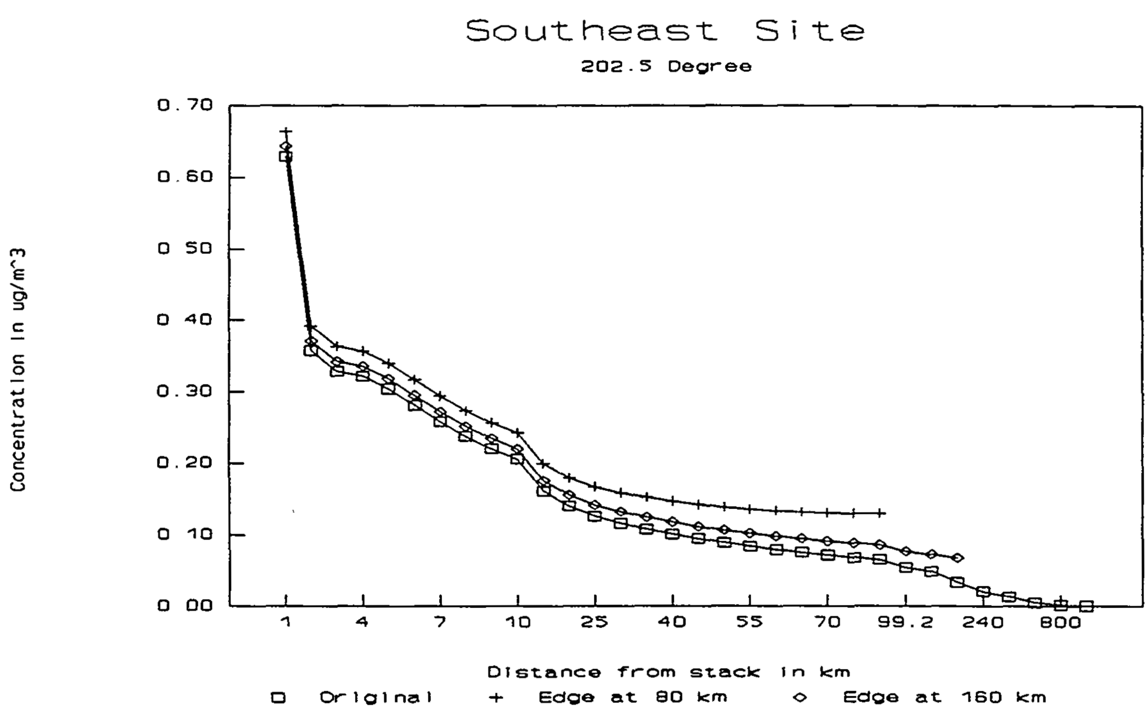

Fig. 17. Southeast Site 202.5 degree

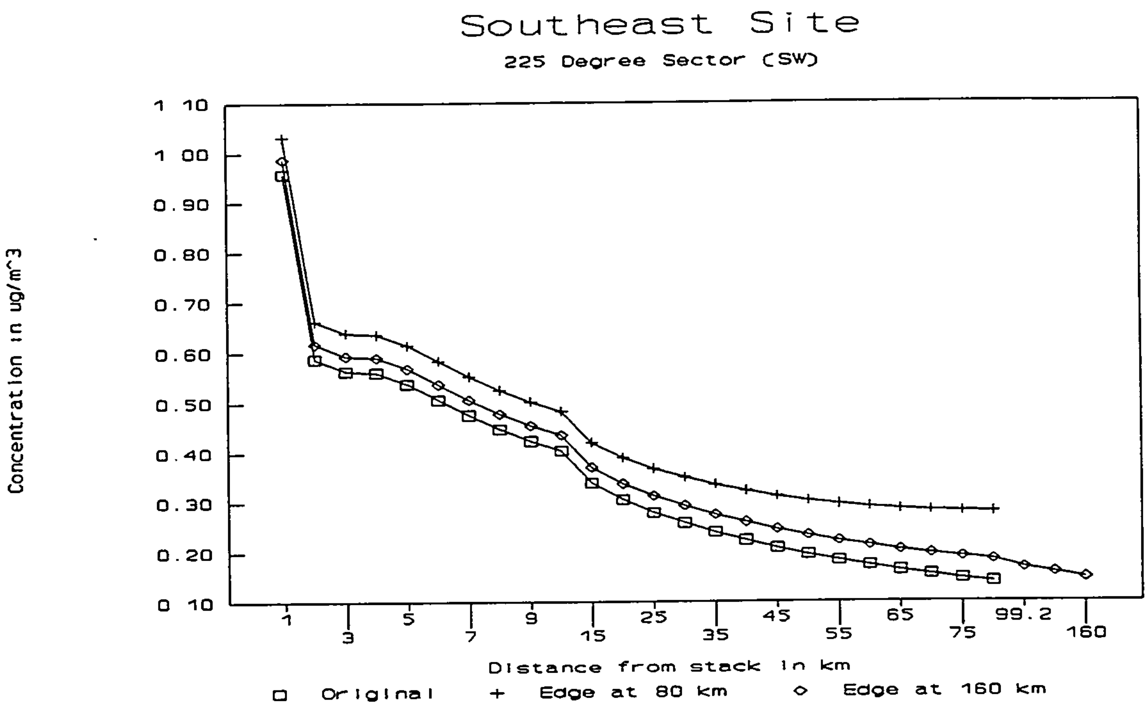

Fig. 18. Southeast Site 225 degree sector (SW)

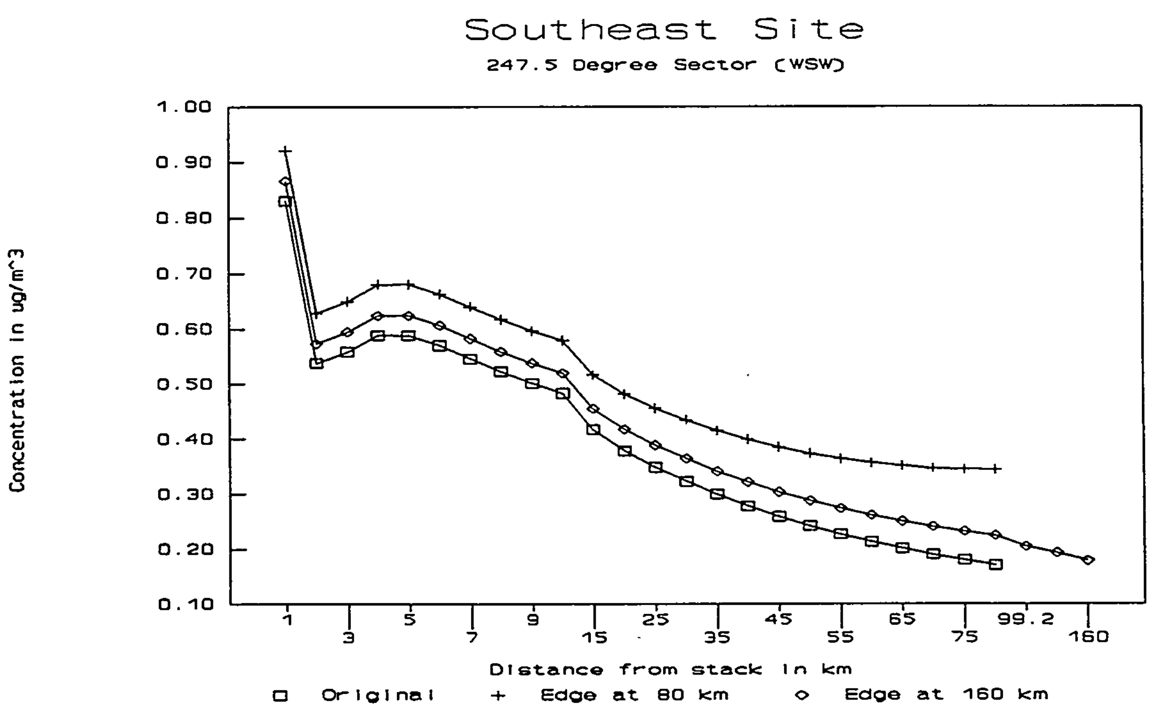

Fig. 19. Southeast Site 247.5 degree sector (WSW)

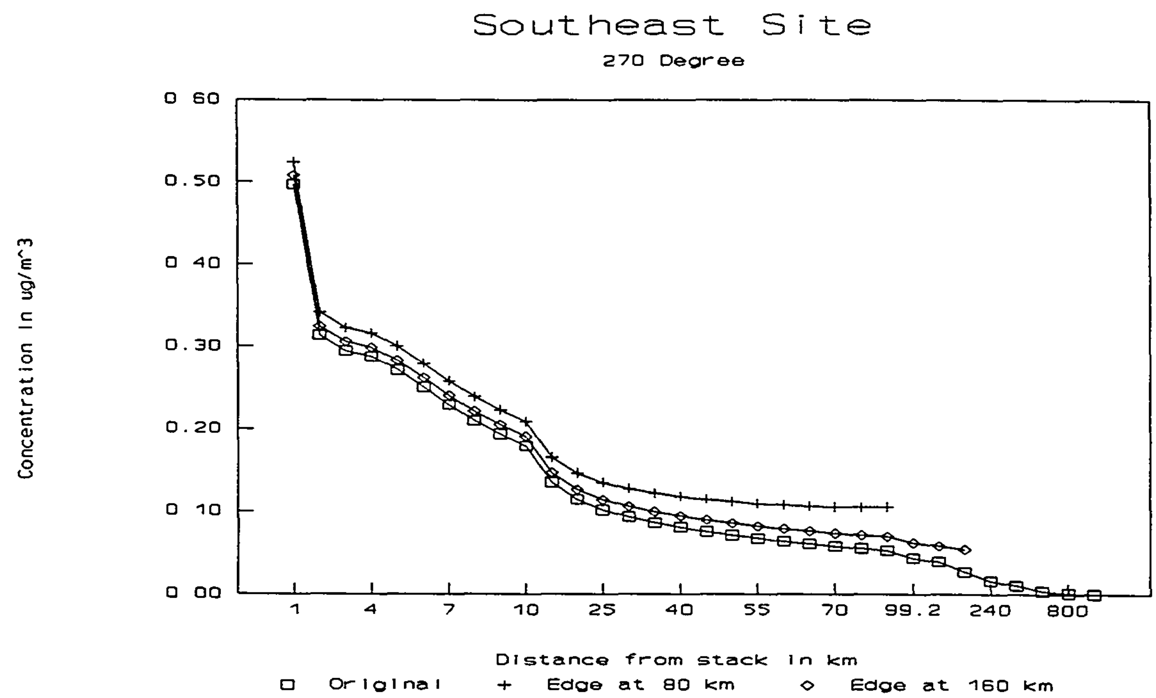

Fig. 20. Southeast Site 270 degree

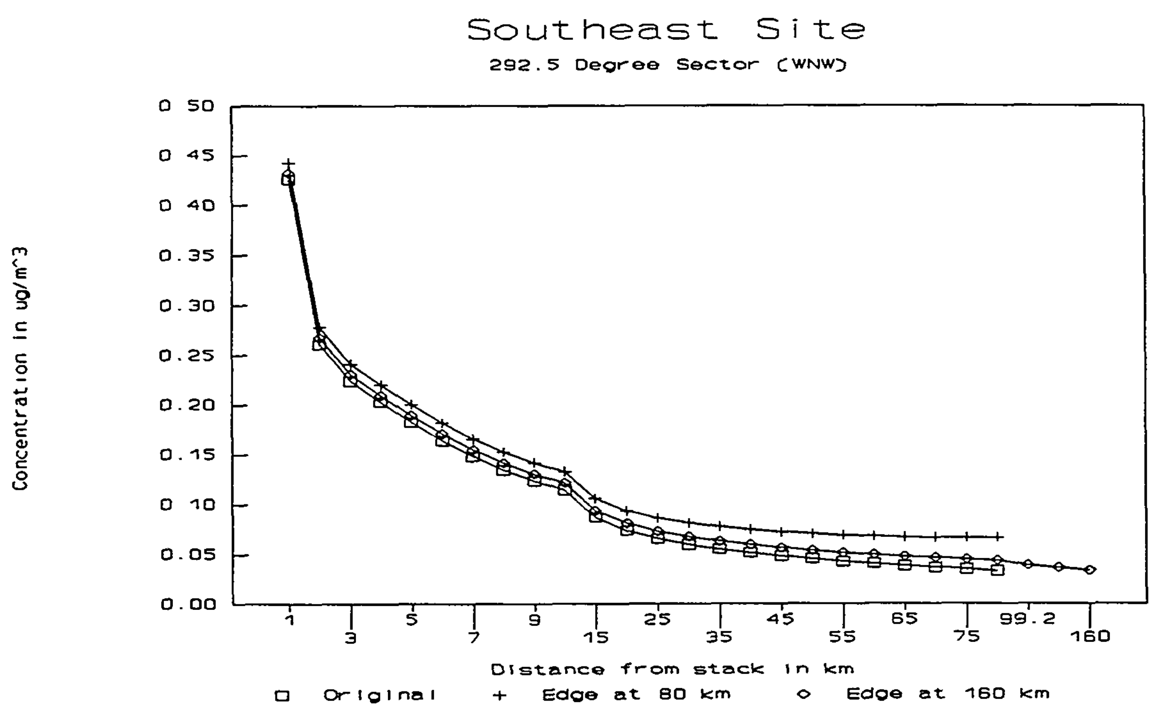

Fig. 21. Southeast Site 292.5 degree sector (WNW)

7. RESULTS AND DISCUSSION

Figures 6 - 21 show the plots of ISCLT concentrations vs. downwind distances. Three curves are shown in each figure. One is for the original ISCLT predictions, the second one is for the sum of the original concentrations and those reflected at the 80 km key-hole range (EDGE80), and the last one is the sum of the original concentrations and those reflected at the 160 km key-hole range (EDGE160). The 360° and 180° sectors were assumed not to be affected by this phenomenon. Thus, the concentration values are identical for the three curves in the two figures for the 180° and 360° sectors.

For the sectors at the 22.5° to 157.5° directions, concentrations are reasonably close to those of the original ones at the distance of 160 km or greater for both EDGE80 and EDGE160. EDGE80 produces higher concentrations than EDGE160 as compared with the original ones within a 160 km range from the stack. This can be attributed to the shorter reflection distance from the stack used in EDGE80 than that in the EDGE160 calculations.

The concentration beyond 15 km decreases exponentially, as seen in Figs. 6 - 21. Comparing the concentration values for the sectors at 202.5° to 337.5°, similar results can be found, i.e., EDGE80 results in higher concentration than EDGE160. Since these sectors were closer to the virtual stack than those in the eastern half of the circle, larger reflected concentrations were added to the original ones than those in the eastern half of the circle.

Figures 22 - 37 show the plots of ISCLT concentrations as functions of downwind distance for the southwest (SW) site. The curves at 360° and 180° are identical for the reason given previously for the southeast site. For the 22.5° sector, the three curves were almost identical to each other. Minute differences among three curves can be found for the 45° - 157.5° sectors. The three concentration curves are close to one another at distances greater than 160 km for all of the above discussed sectors. EDGE80 produced appreciably larger concentrations, both on an absolute basis and on a percentage basis, than EDGE160 and than the original ISCLT results for the sectors of 202.5°, 225°, and 247.5°. The percentage concentration differences are smaller for the rest of the sectors in the western half of the circle at the SW site, although the absolute differences for EDGE80 in the 270° and 292.5° sectors are still fairly large.

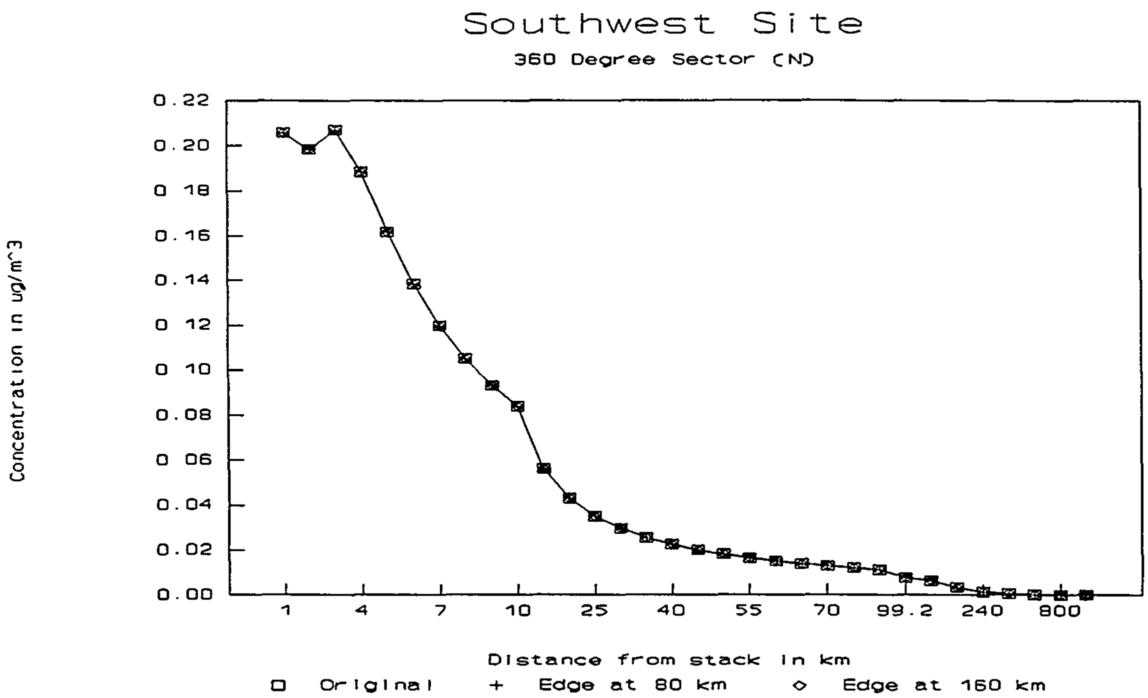

Fig. 22. Southwest Site 360 degree sector (N)

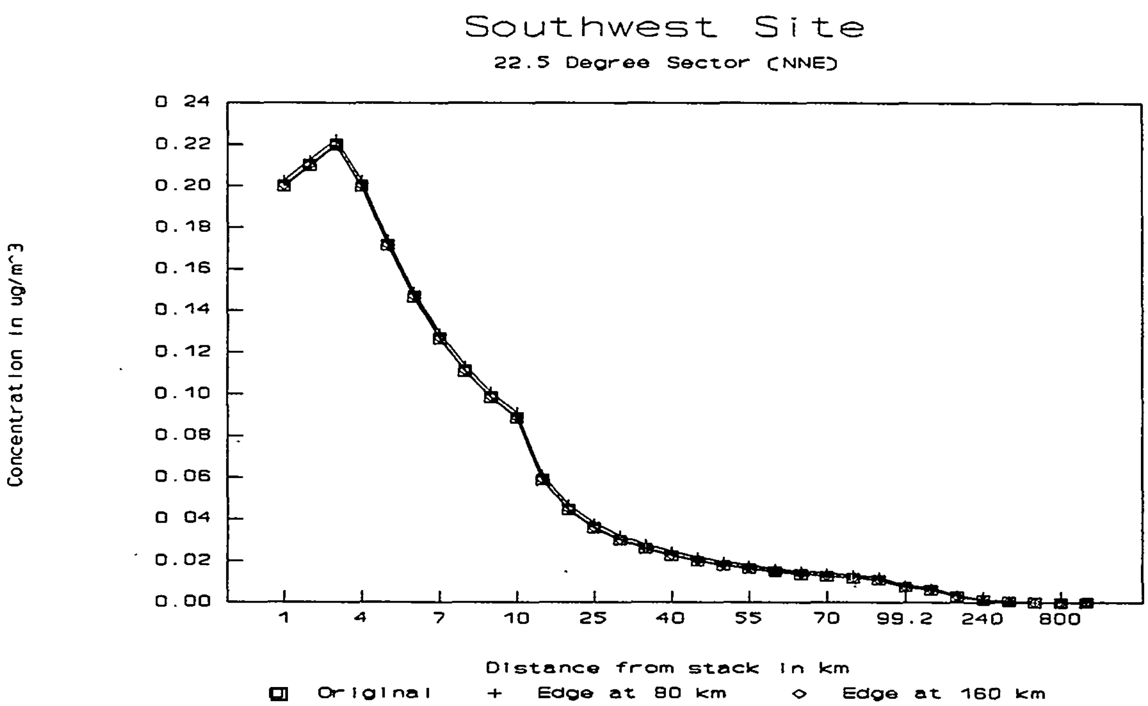

Fig. 23. Southwest Site 22.5 degree sector (NNE)

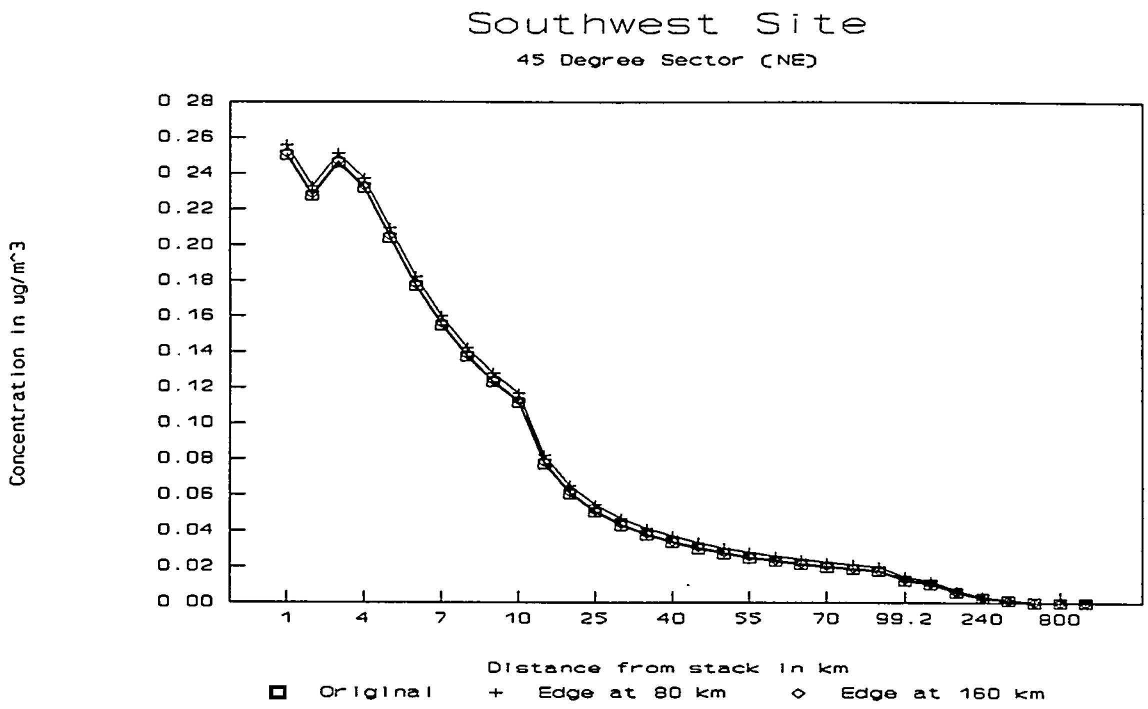

Fig. 24. Southwest Site 45 degree sector (NE)

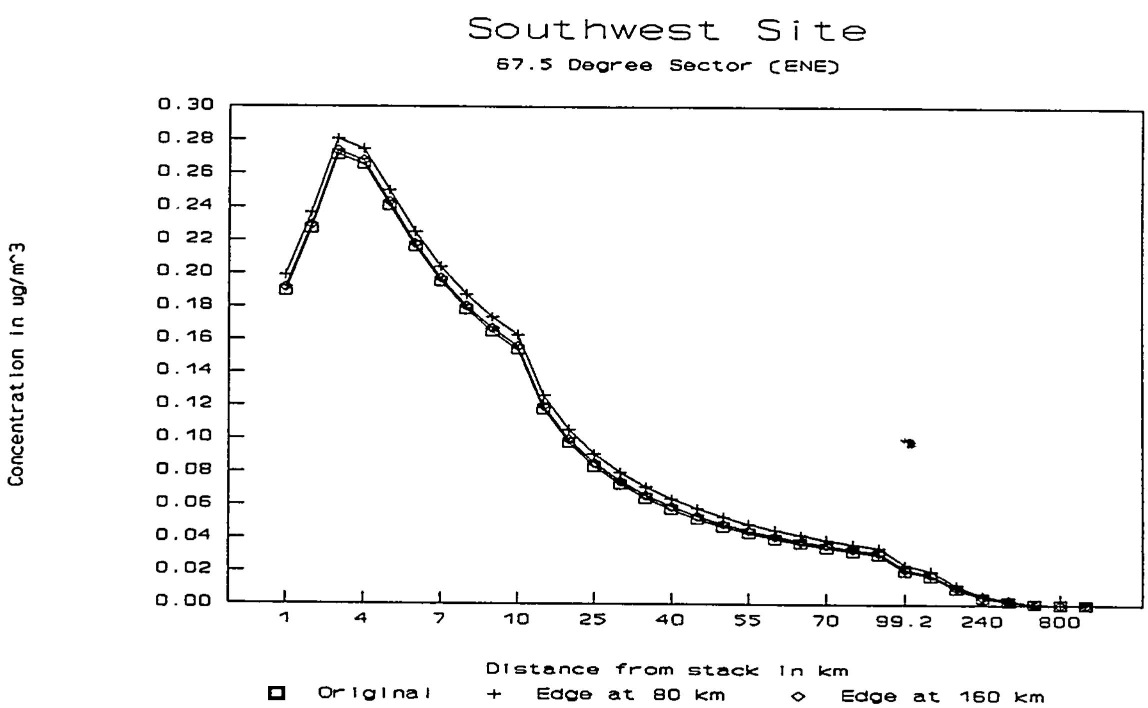

Fig. 25. Southwest Site 67.5 degree sector (ENE)

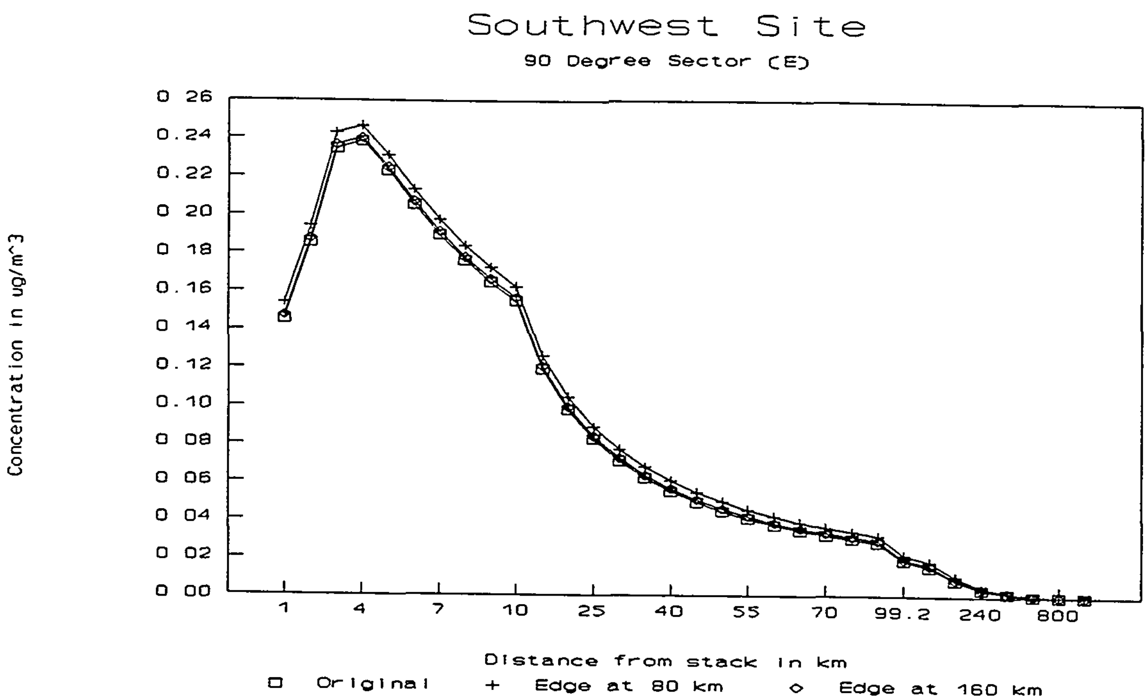

Fig. 26. Southwest Site 90 degree sector (E)

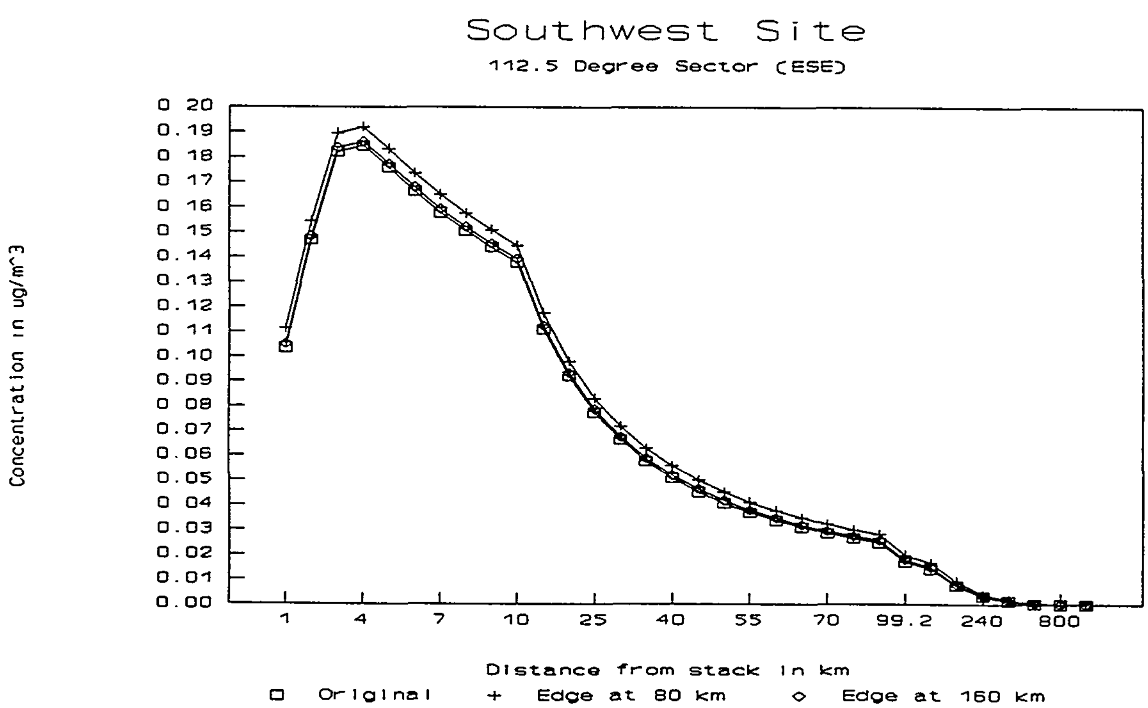

Fig. 27. Southwest Site 112.5 degree sector (ESE)

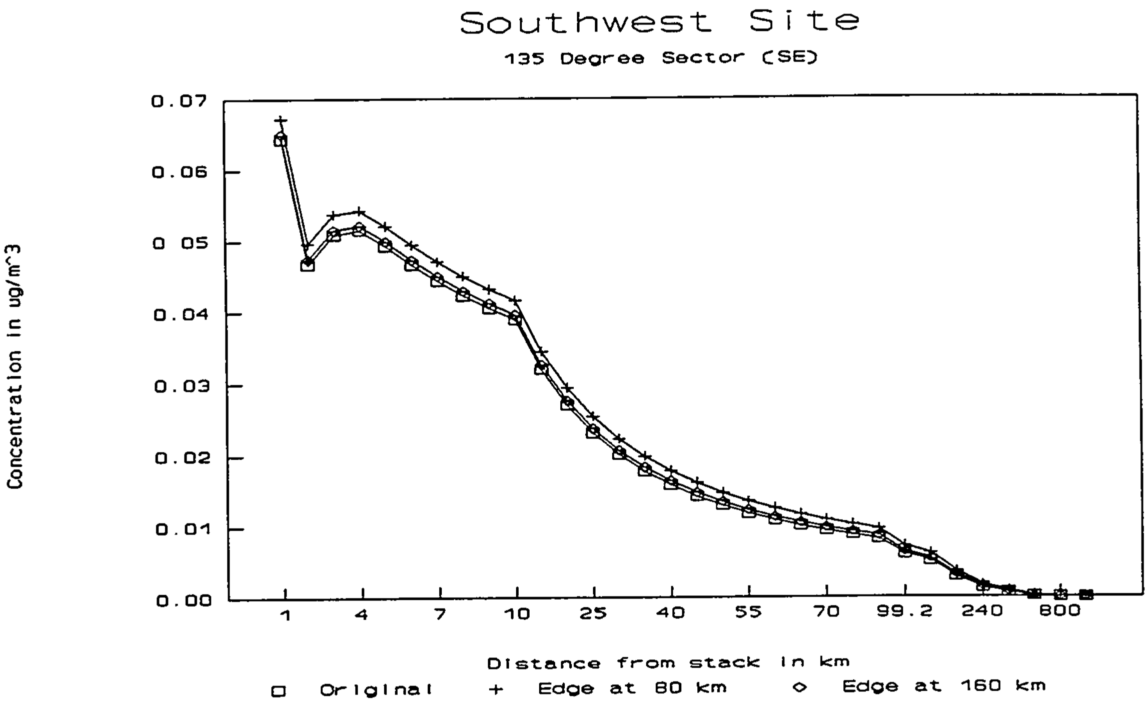

Fig. 28. Southwest Site 135 degree sector (SE)

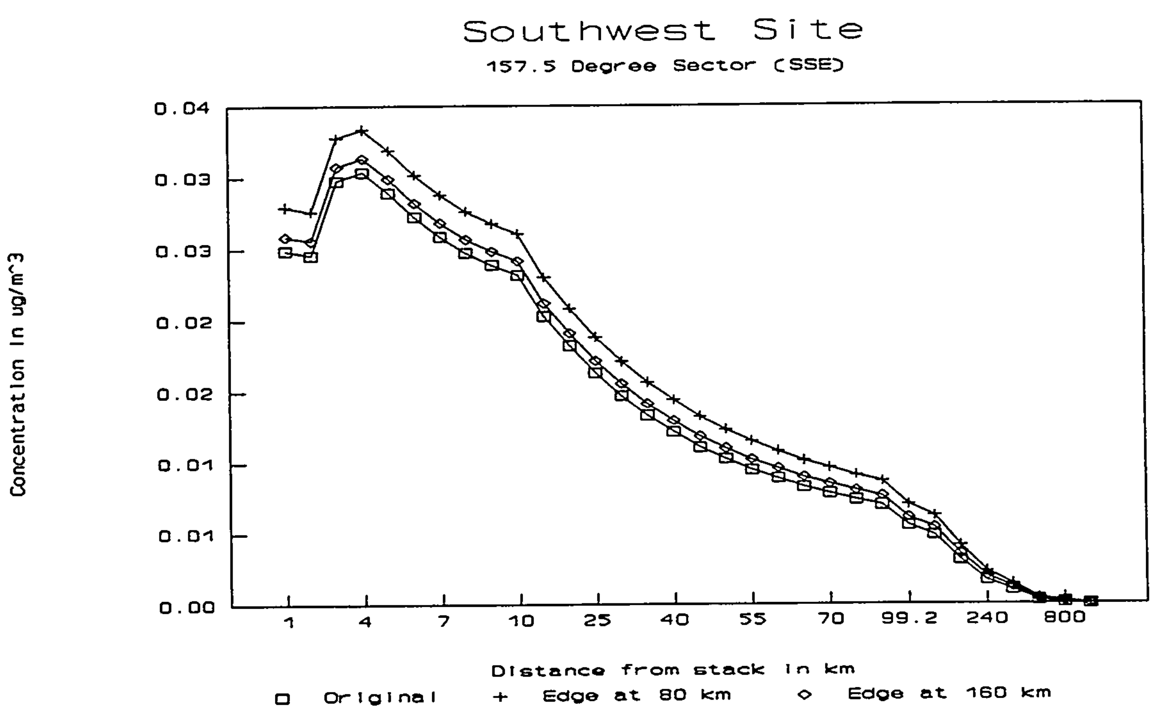

Fig. 29. Southwest Site 157.5 degree sector (SSE)

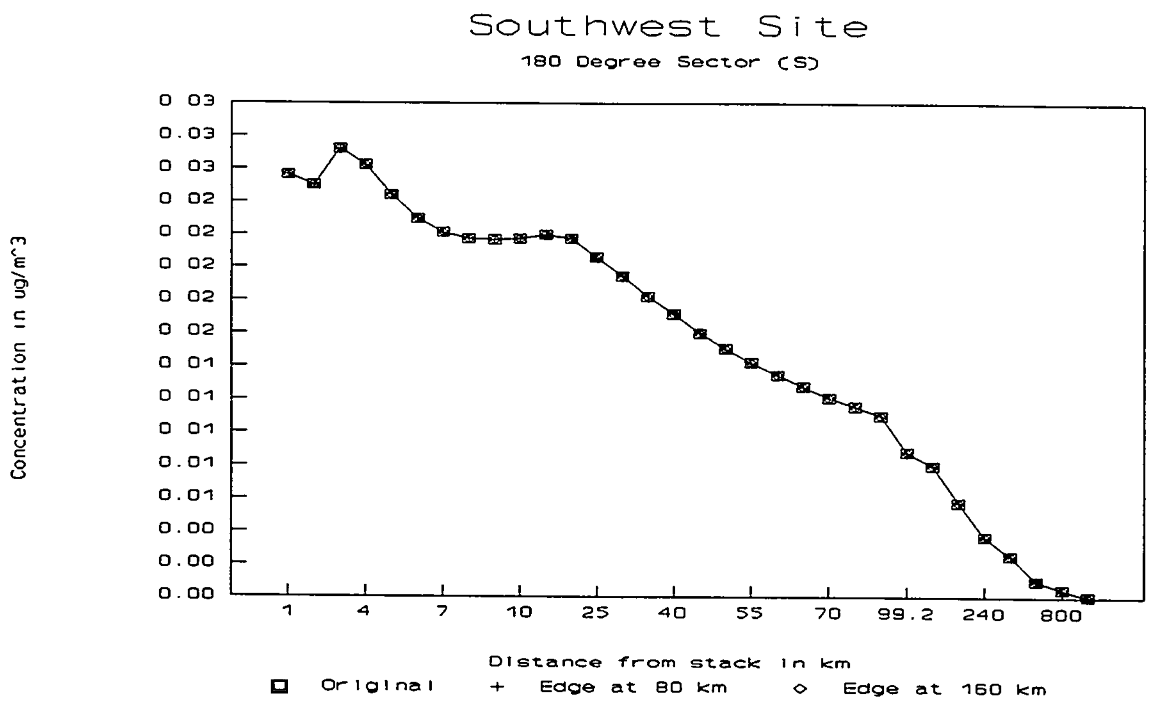

Fig. 30. Southwest Site 180 degree sector (S)

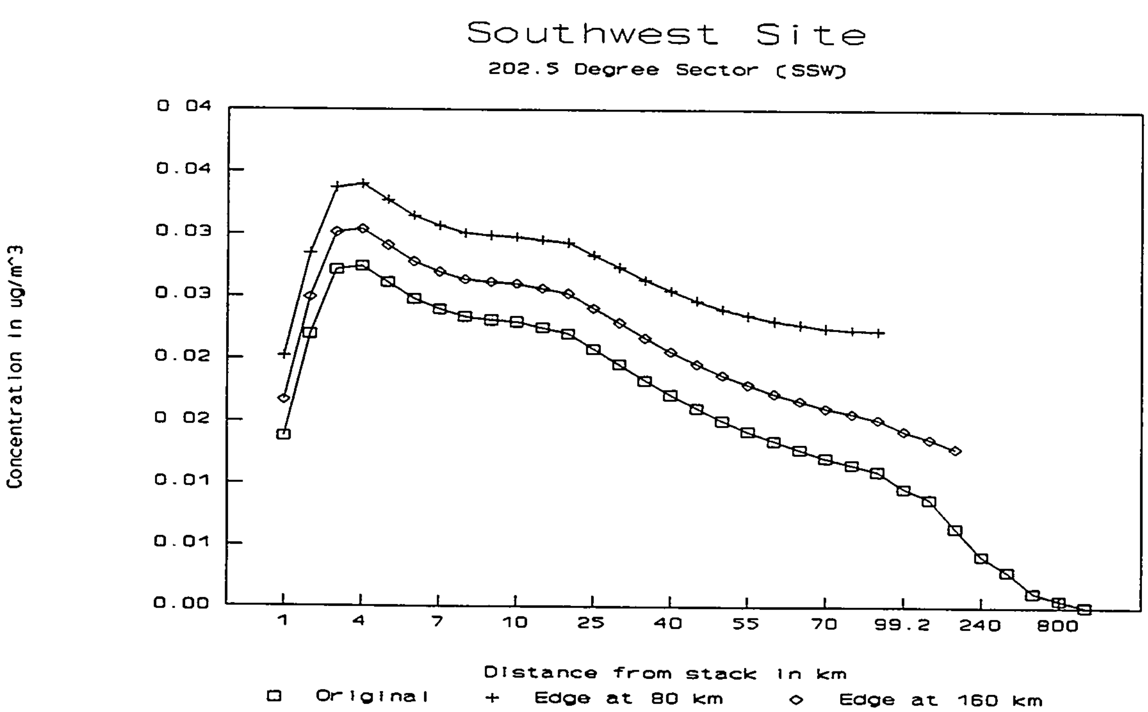

Fig. 31. Southwest Site 202.5 degree sector (SSW)

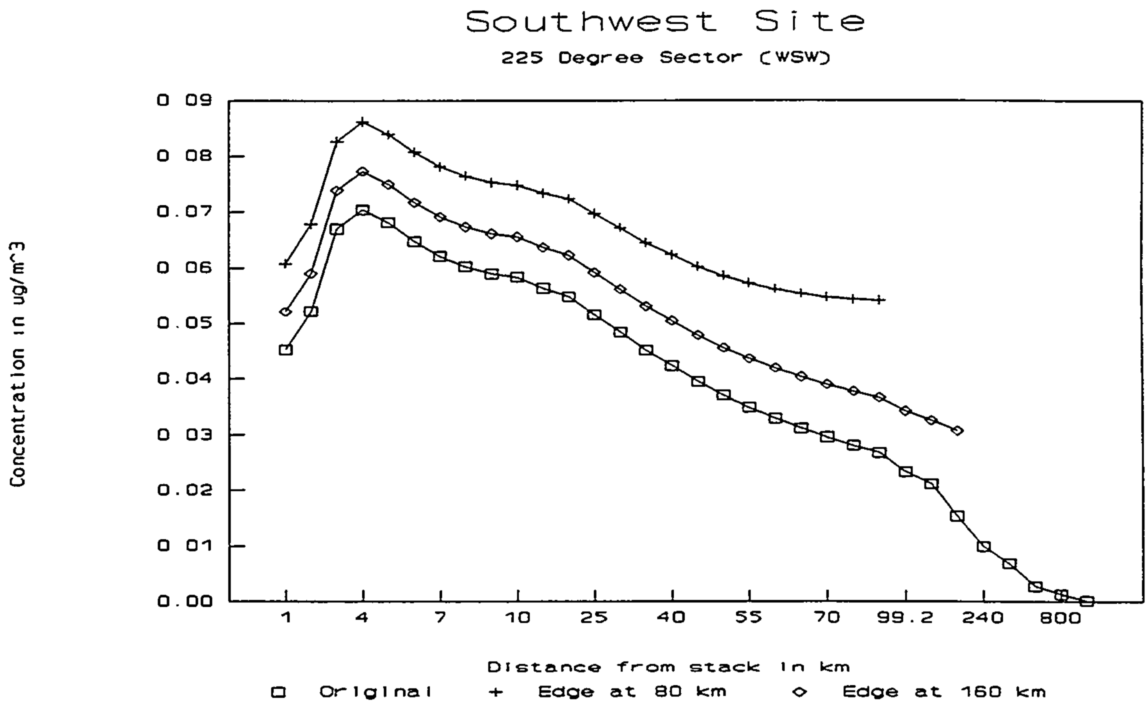

Fig. 32. Southwest Site 225 degree sector (WSW)

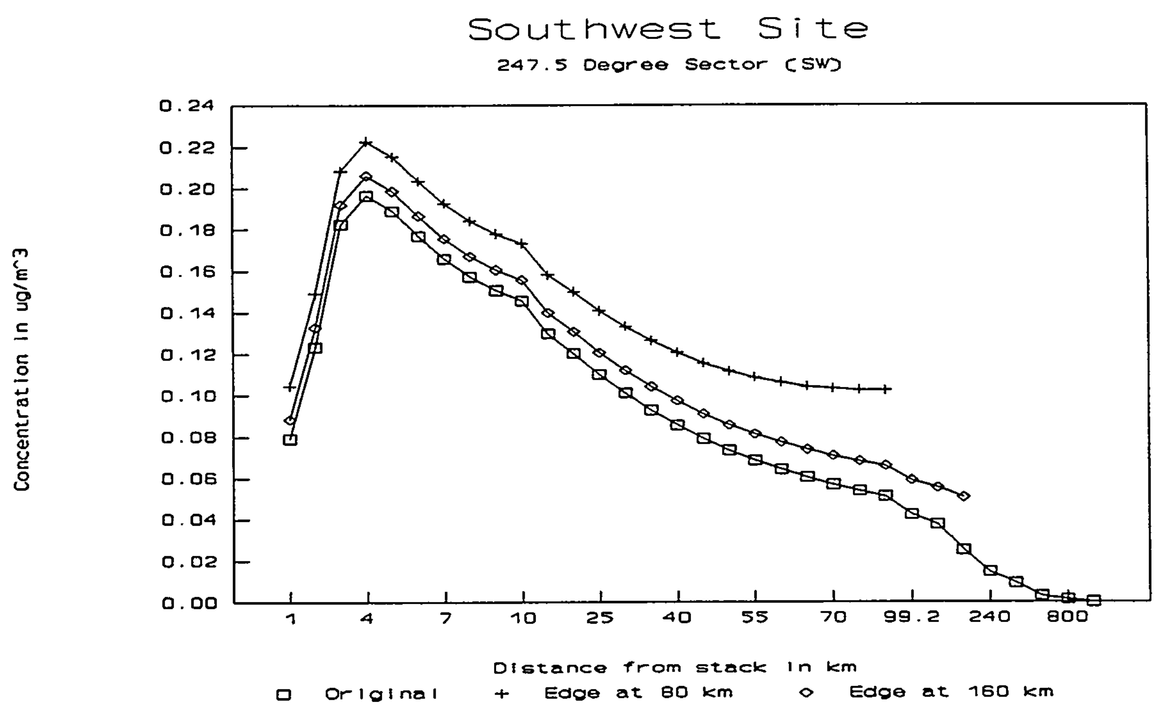

Fig. 33. Southwest Site 247.5 degree sector (SW)

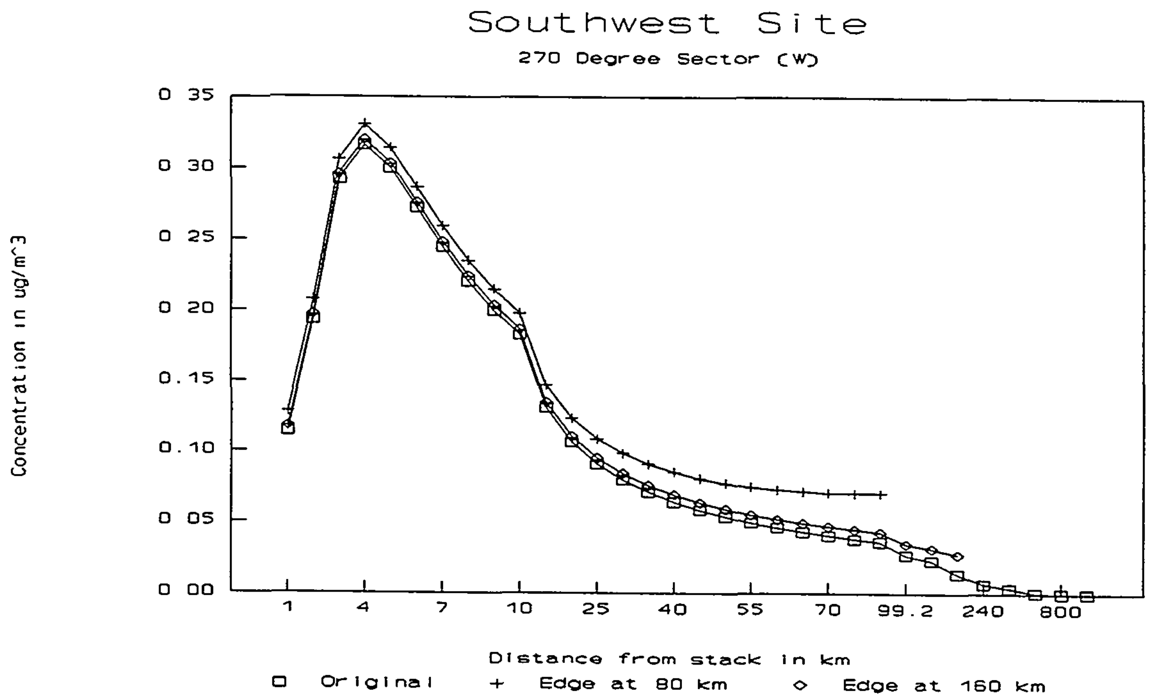

Fig. 34. Southwest Site 270 degree sector (W)

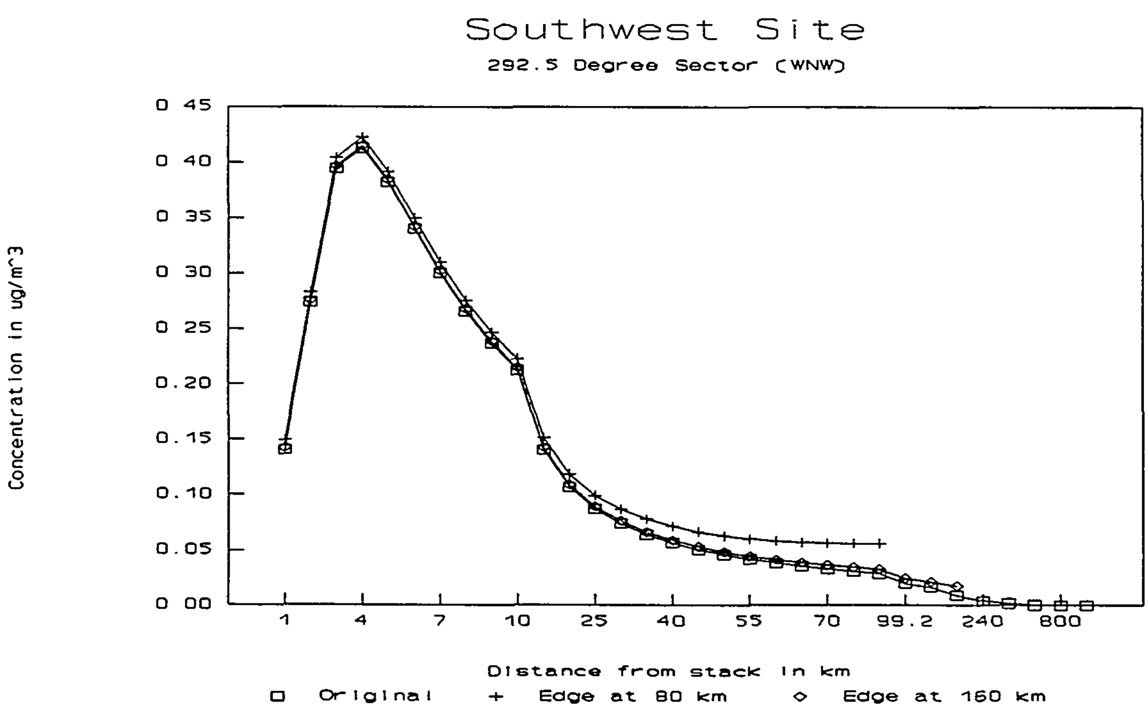

Fig. 35. Southwest Site 292.5 degree sector (WNW)

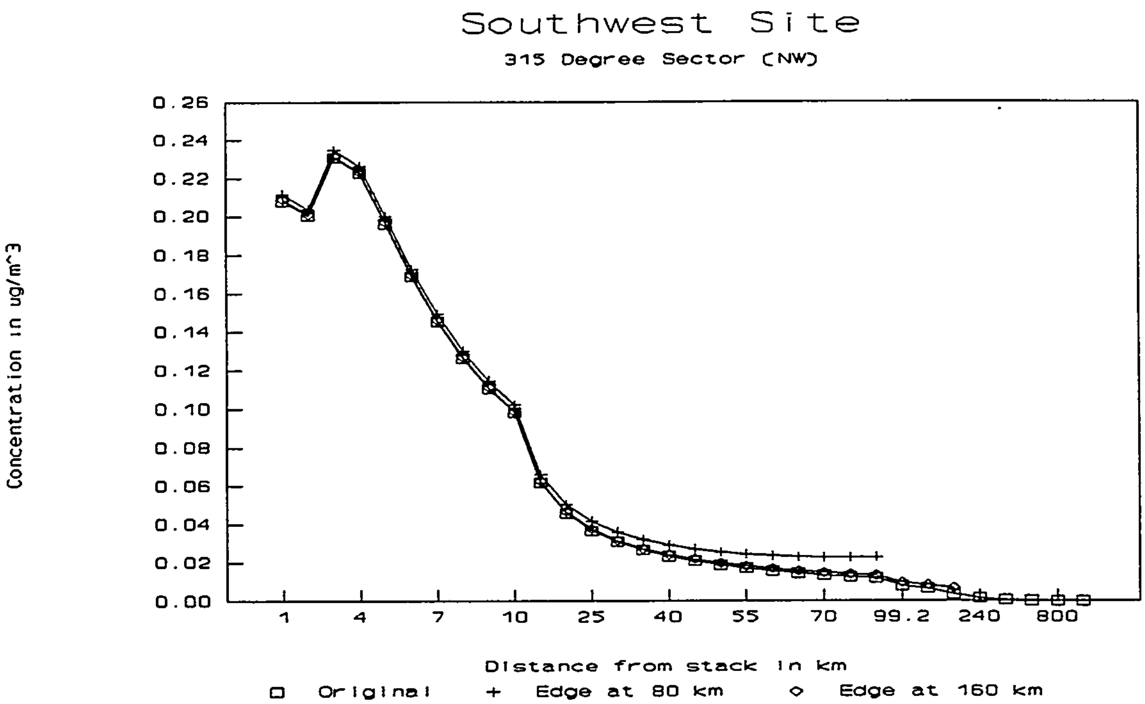

Fig. 36. Southwest Site 315 degree sector (NW)

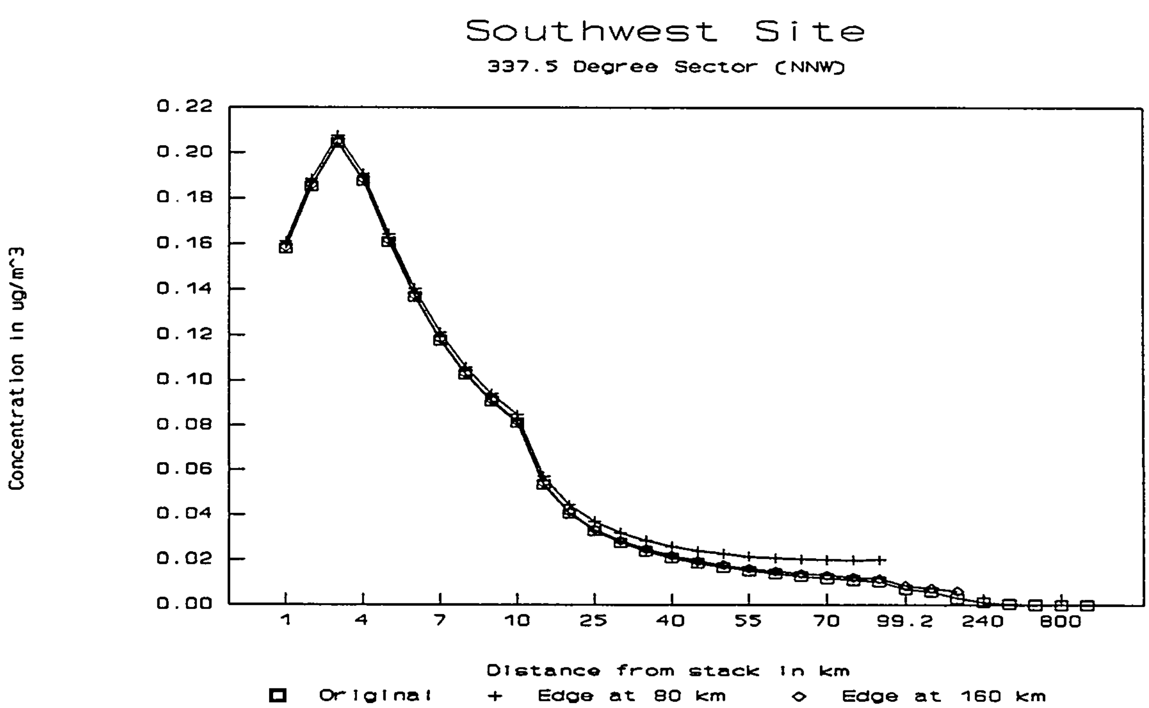

Fig. 37. Southwest Site 337.5 degree sector (NNW)

8. SUMMARY

Statistical models were derived to estimate ambient concentrations of four criteria pollutants at distances greater than 80 km from the stack of a coal-fired power plant. The models indicate that, beyond 15 km, the log-transformed concentration was strongly correlated with the square root of downwind distance. A statistical method was developed and applied to account for the hypothesized reversal of the direction of pollutant transport due to changes in the wind flow pattern. Results of this analysis indicate that such changes in wind direction do not appreciably affect the annual average concentrations beyond 160 km. Some differences were found between the initial extrapolations and the revised concentrations (i.e., the sum of initial and reflected concentrations) in the 80 to 160 km range. The largest of these differences, on a percentage basis, were in the southwest sector and adjacent sectors at the western site. An appreciable effect was found for many of the sectors within 80 km at both sites.

Results of this analysis indicate that such changes in wind direction do not appreciably affect the annual average concentrations beyond 160 km.

Table 2. Regression parameters for calibration models

Southeast Site			Southwest Site		
	Intercept	Slope		Intercept	Slope
SO2			SO2		
360.0	-1.3160	-0.2031	360.0	-1.7923	-0.3079
337.5	-1.7342	-0.2091	337.5	-1.8177	-0.3159
315.0	-1.9039	-0.2011	315.0	-1.7072	-0 3139
292.5	-1.8050	-0.1796	292.5	-0.9155	-0 3003
270.0	-1.3828	-0.1753	270.0	-1.1398	-0 2489
247.5	-0.1671	-0.1777	247.5	-1.2675	-0.1904
225.0	-0.4014	-0.1745	225 0	-2.2031	-0.1559
202.5	-1.1858	-0.1734	202.5	-3.1374	-0.1509
180.0	-1.3635	-0.1718	180.0	-3.1688	-0.1570
157.5	-1.1845	-0.1776	157.5	-3.0473	-0 2159
135.0	-0.6523	-0.1933	135 0	-2.4294	-0.2685
112.5	-0.7631	-0.1841	112.5	-1.0840	-0.2988
90.0	-0.6779	-0.1912	90.0	-1.0767	-0 2914
67.5	-0.2155	-0.1954	67.5	-1.1455	-0 2763
45.0	0.1447	-0 2064	45.0	-1.5397	-0.2875
22.5	-0.3692	-0.2139	22.5	-1.7062	-0.3212
NO2			NO2		
360.0	-0.8050	-0.2031	360.0	-0.7910	-0.3079
337.5	-1.2232	-0.2091	337.5	-0.8165	-0.3159
315.0	-1.3929	-0 2011	315 0	-0.7060	-0.3139
292.5	-1.2940	-0.1796	292.5	-0.0858	-0.3003
270.0	-0.8718	-0.1753	270.0	-0.1386	-0.2489
247.5	0.3440	-0.1777	247.5	-0 2663	-0.1904
225.0	0.1096	-0.1745	225.0	-1.2019	-0 1559
202.5	-0.6748	-0.1734	202.5	-2.1362	-0.1509
180.0	-0.8524	-0.1718	180.0	-2.1675	-0.1570
157.5	-0.6734	-0.1776	157.5	-2.0461	-0.2159
135.0	-0.1412	-0.1933	135.0	-1.4282	-0.2685
112.5	-0.2520	-0.1841	112.5	-0.0828	-0.2988
90.0	-0.1669	-0.1912	90.0	-0.0755	-0.2914
67.5	0.2956	-0.1954	67.5	-0.1442	-0.2763
45.0	0.6558	-0 2064	45.0	-0.5385	-0.2875
22.5	0.1419	-0.2139	22.5	-0.7050	-0.3212

Table 2. Regression parameters for calibration models

Southeast Site			Southwest Site		
TSP			TSP		
360.0	-3.7787	-0.2031	360.0	-3.9435	-0.3079
337.5	-4.1969	-0 2091	337.5	-3.9690	-0.3159
315 0	-4.3666	-0 2011	315.0	-3.8584	-0.3139
292.5	-4.2677	-0.1796	292.5	-3 0667	-0 3003
270.0	-3.8455	-0.1753	270.0	-3.2910	-0.2489
247.5	-2.6297	-0.1777	247.5	-3.4188	-0.1904
225 0	-2.8641	-0.1745	225.0	-4.3544	-0.1559
202.5	-3.6485	-0.1734	202.5	-5.2887	-0.1509
180.0	-3.8261	-0.1718	180.0	-5.3200	-0.1570
157.5	-3 6471	-0.1776	157.5	-5.1985	-0.2159
135.0	-3.1149	-0.1933	135.0	-4.5806	-0.2685
112.5	-3.2258	-0.1841	112.5	-3.2352	-0 2988
90 0	-3.1406	-0.1912	90 0	-3.2279	-0.2914
67.5	-2.6782	-0.1954	67.5	-3 2967	-0 2763
45.0	-2.3179	-0.2064	45.0	-3.6910	-0.2875
22 5	-2.8318	-0 2139	22.5	-3.8574	-0.3212
PM-10			PM-10		
360.0	-4.1877	-0.2031	360 0	-4 3378	-0.3079
337.5	-4.6059	-0 2091	337.5	-4 3633	-0.3159
315 0	-4.7756	-0.2011	315.0	-4.2527	-0.3139
292.5	-4 6767	-0.1796	292.5	-3.4610	-0.3003
270 0	-4.2545	-0.1753	270.0	-3.6853	-0.2489
247.5	-3.0388	-0.1777	247.5	-3 8131	-0.1904
225.0	-3.2731	-0.1745	225.0	-4.7486	-0.1559
202.5	-4.0575	-0.1734	202.5	-5.6830	-0.1509
180.0	-4 2352	-0.1718	180 0	-5.7143	-0.1570
157.5	-4.0562	-0.1776	157.5	-5.5928	-0 2159
135 0	-3.5240	-0.1933	135.0	-4.9749	-0.2685
112 5	-3.6348	-0.1841	112.5	-3.6295	-0.2988
90.0	-3.5496	-0.1912	90.0	-3 6222	-0.2914
67.5	-3.0872	-0.1954	67.5	-3.6910	-0 2763
45 0	-2.7269	-0.2064	45.0	-4 0853	-0.2875
22.5	-3.2409	-0.2139	22.5	-4.2517	-0.3212

Table 3. ***Joint frequencies of occurrence of the wind speed and wind* direction by season and Pasquill stability category for the SE and the SW sites.**

	SE	SW
1-A	4.89%	0.35%
1-B	4.18%	4.79%
1-C	7.75%	11.65%
1-D	63.14%	34.27%
1-E	10.41%	18.09%
1-F	9.63%	30.84%
TOTAL	100%	100%
2-A	8.27%	2.67%
2-B	4.52%	7.45%
2-C	10.29%	12.19%
2-D	57.08%	41.29%
2-E	8.88%	16.63%
2-F	10.96%	19.79%
TOTAL	100%	100%
3-A	9.66%	5.59%
3-B	7.44%	13.29%
3-C	16.45%	17.61%
3-D	41.52%	27.80%
3-E	11.66%	16.25%
3-F	13.27%	19.45%
TOTAL	100%	100%
4-A	7.05%	1.29%
4-B	6.36%	9.24%
4-C	9.86%	13.71%
4-D	50.72%	27.65%
4-E	13.42%	19.50%
4-F	12.58%	28.61%
TOTAL	100%	100%

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Lamb, R. G. (1983). "A Regional Scale (1,000 km) Model of Photochemical Air Pollution, Part I - Theoretical Formulation, Part II - Input Processor Network Design, and Part III - Tests of Numerical Algorithms", EPA-600/3-83-035, EPA-600/3-84-085, and EPA-600/3-85-037, U.S. Environmental Protection Agency, Research Triangle Park, NC.

Scire, J. S. et al. (1984). "User's Guide to the MESOPUFF II Model and Related Processor Programs", EPA-600/8-84-013, U.S. Environmental Protection Agency, Research Triangle Park, NC.

PAPER 3[1]

OZONE MODELING

1. INTRODUCTION

1.1 POWER PLANT PLUME PHOTOCHEMISTRY

Exhaust gases from power plants that burn fossil fuels contain concentrations of sulfur dioxide (SO_2), nitric oxide (NO), particulate matter, hydrocarbon compounds and trace metals. Estimated emissions from the operation of a hypothetical 500 MW coal-fired power plant are given in Table 1. Ozone is considered a secondary pollutant, since it is not emitted directly into the atmosphere but is formed from other air pollutants, specifically, nitrogen oxides (NO_x) and non-methane organic compounds (NMOC) in the presence of sunlight. (NMOC are sometimes referred to as hydrocarbons, HC, or volatile organic compounds, VOC, and they may or may not include methane.)

Ozone is considered a secondary pollutant, since it is not emitted directly into the atmosphere but is formed from other air pollutants, specifically, nitrogen oxides (NO_x) and non-methane organic compounds (NMOC) in the presence of sunlight.

Additionally, ozone formation is a function of the ratio of NMOC concentrations to NO_x concentrations. Figure 1 is a typical ozone isopleth generated with the Empirical Kinetic Modeling Approach (EKMA) option of the Environmental Protection Agency's (EPA) Ozone Isopleth Plotting Mechanism (OZIPM-4) model. Ozone isopleth diagrams, originally generated with smog chamber data, are more commonly generated with photochemical reaction mechanisms and tested against smog chamber data. The shape of the isopleth curves in Fig. 1 is a function of the region (i.e. background conditions) where ozone concentrations are simulated.

The location of an ozone concentration on the isopleth diagram is defined by the ratio of NMOC and NO_x coordinates of the point, known as the

[1]Based largely on Ph.D. dissertation by C. M. McIlvaine.

Table 1. Controlled emission rates from the operation of the hypothetical 500 MW coal-fired power plant.

Pollutant	Emission rate for operation phase: Tons/GWh	Tons/year	Grams/sec
EASTERN COAL			
NO_x	2.90	9520	273.9
SO_2	1.74	5712	164.3
Hydrocarbon	0.06	210	6.0
TSP	0.15	485	14.0
PM_{10}	0.10	323	9.3
WESTERN COAL			
NO_x	2.20	7236	208.2
SO_x	0.81	2660	76.5
Hydrocarbon	0.09	293	8.4
TSP	0.10	310	8.9
PM_{10}	0.06	207	6.0
BOTH COALS			
Arsenic	2×10^4	0.66	1.9×10^2
Cadmium	3×10^6	0.01	2.9×10^4
Manganese	1.3×10^4	0.44	1.3×10^2
Lead	9×10^5	0.31	8.9×10^3
Selenium	5×10^5	0.17	4.9×10^3

NOTE: NO_x nitrogen oxide
SO_2 sulfur dioxide
TSP total suspended particles
PM_{10} particulate matter less than or equal to 10 microns in diameter

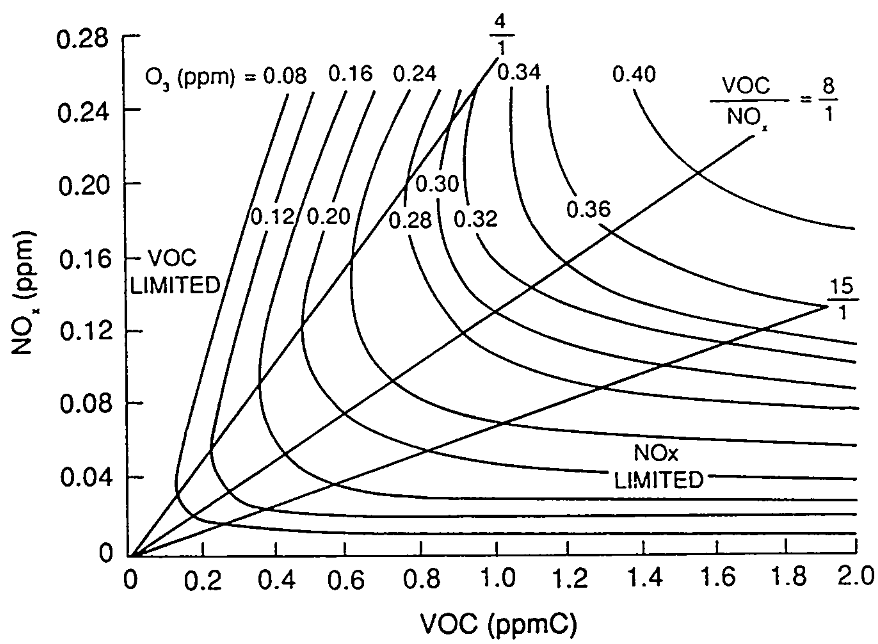

Fig. 1. Typical ozone isopleths generated with the EKMA option of EPA's OZIPM-4 model. The NO_x-limited region is typical of rural and suburban areas and the VOC-limited region is typical of highly polluted urban areas.

***Source:* National Research Council (NRC) (1991).**

NMOC/NO_x ratio (NRC 1991). The diagonal line from the lower left to the upper right corresponds to an NMOC/NO_x ratio of approximately 8/1. This line can be used to define two areas of the graph. Areas to the left of the line have low NMOC/NO_x ratios and are described as NMOC-limited. In these areas, such as highly polluted urban areas characterized by relatively high concentrations of NO_x, the addition of NO_x emissions results in little or no increase in ozone concentrations and may actually result in lower ozone concentrations due to the scavenging of ozone by NO_x emissions (see equation 1 below). The area to the right of the line in Fig. 1 has high NMOC/NO_x ratios and is described as NO_x-limited. Rural areas, such as the Southeast Reference site, and suburbs downwind of cities are often characterized by high NMOC/NO_x ratios. Since the only source of ozone in the troposphere is the photolysis of NO_2 (equations 2 and 3 below), any increase in NO_x emissions in NO_x-limited areas results in higher ozone concentrations (NRC 1991).

While most large power plants are considered significant sources of NO_x emissions, NMOC emissions from power plants are not considered significant and do not typically require control. Since NMOC emissions from power plants are not present in sufficient quantities to provide an optimal hydrocarbon to NO_x ratio within the plume, ozone formation from the emissions of power plants is the result of a complex series of reactions involving NO_x emissions from the plant reacting with ambient concentrations of hydrocarbons, hydrocarbon derivatives and ozone. Ambient hydrocarbons may be from either man-made or natural sources.

Initially, ozone that may be present in the ambient air, during daylight and evening hours, reacts with the NO from the power plant to form nitrogen dioxide (NO_2) and oxygen (O_2), described by the reaction:

$$NO + O_3 \rightarrow NO_2 + O_2 \qquad [1]$$

This reaction causes the characteristic ozone depletion observed near the stack in power plant plumes. Ozone depletion is defined here as ozone concentrations within the power plant plume that are less than those outside the power plant plume. In the presence of sunlight, within the first few tens of kilometers of the plant, the photochemistry within power plant plumes (with low hydrocarbon concentrations) can be described by these three equations (White 1977), known as the NO_2 photolytic cycle:

$$NO + O_3 \rightarrow NO_2 + O_2 \qquad [1]$$

$$NO_2 + h\nu \rightarrow NO + O(^3P) \qquad [2]$$

$$O(^3P) + O_2 + M \rightarrow O_3 + M \qquad [3]$$

where M is any energy-accepting third body, usually nitrogen (N_2) or O_2, and $O(^3P)$ is one of two electronic states of oxygen known as the triplet-P (Seinfeld 1975). NO_2 absorbs ultraviolet energy from the sun which breaks the molecule into NO and a ground state oxygen atom $O(^3P)$. Energy from solar radiation is represented by hv, which is the product of Planck's constant (h) and the frequency of the electromagnetic wave of solar radiation (v). The net effect of these three reactions is conversion of the NO emissions to NO_2 with no increase in ozone concentrations.

The net generation of ozone in power plant plumes can only occur in the presence of reactions which compete with the ozone depletion reaction [1]. Further downwind, as the plume disperses, ambient air containing pollutants from other sources, most importantly reactive hydrocarbons, becomes entrained into the plume. Reactive hydrocarbons in the ambient air participate in a complex series of oxidation reactions which result in the formation of highly reactive radicals.

An extremely important intermediate compound in this series of reactions is the hydrocarbon derivative, formaldehyde. Formaldehyde is also emitted directly from such sources as automobiles, forest fires, manufacturing, printing, and spray painting (Graedel 1978). Formaldehyde reacts in the presence of sunlight to form the highly reactive hydroperoxy radical (HO_2) by the reactions (Carlier et al. 1986):

$$HCHO + hv \rightarrow H\cdot + HCO\cdot \qquad [4]$$

$$HCO\cdot + O_2 \rightarrow HO_2\cdot + CO \qquad [5]$$

Ozone depletion is slowed by the reaction of NO with the hydroperoxy radical (HO_2):

$$HO_2\cdot + NO \rightarrow OH\cdot + NO_2 \qquad [6]$$

as the ozone generating reactions [2] and [3] continue in the plume. Eventually, the ozone concentration within the plume may exceed ambient (i.e. outside the plume) levels.

The formation of ozone is controlled by a combination of conditions, including ambient ozone concentrations which provide the mechanism necessary for the initial conversion of NO to NO_2, reactive hydrocarbon concentrations of the ambient air mass, and the rate of entrainment of ambient air within the plume. These conditions, as well as sufficient photochemical activity, determine whether ozone levels in the plume will eventually exceed ambient levels to form the widely documented ozone 'bulge' (Keifer 1977; Meagher et al. 1981; Luria et al. 1983; Gillani and Wilson 1980; Davis 1974).

To summarize, the major factors in the formation of excess ozone in power plant plumes are:

1. NO_x emissions from the plant,

2. ambient ozone concentrations,

3. reactive hydrocarbons,

4. favorable ratio of ambient hydrocarbons to plume NO_x,

5. atmospheric mixing, and

6. sufficient photochemical activity (sunlight and temperature).

1.2 MODELING OBJECTIVES

The coal fuel cycle analysis requires that an estimate be made of ozone concentrations that occur in the vicinity of a coal-fired power plant located at the Southeast Reference site, due to emissions of nitrogen oxides (NO_x) and non-methane organic compounds (NMOC) from the plant. A discussion of the probable impact from emissions from the plant located at the Southwest Reference site is given in Section 5. Estimates of the peak daily ozone concentrations, due to the plant, for each day of the ozone season, are needed for the health effects analysis. The crops and vegetation effects analysis requires an estimate of the seasonal 9 a.m. to 9 p.m. average ozone concentrations due to the plant. These modeling requirements present a unique challenge, since all the currently available computer models which simulate ozone formation are designed to predict hourly and instantaneous ozone concentrations, over a period of several days at most. These predictions are primarily for comparison to the National Ambient Air Quality Standard (NAAQS) of 120 ppb (one-hour average) not to be exceeded more than once per year.

Modeling studies in the U. S. have, so far, concentrated on assessing ozone concentrations under episodic conditions over periods from one month to one year. Recently, attention has been given to the long-term ozone problem over periods from one month to one year. Simpson (1992) reports that De Leeuw et al. (1990) and Hov and Flatoy (1991) have used trajectory models to calculate long-term ozone concentrations in the Netherlands and Norway and Christensen et al. (1991) has developed a long term Eulerian model. Simpson (1992) proposes three reasons for this trend: (1) the most commonly applied approach to ozone assessments is the episodic study which generates results for a period of a few days; it is not clear how these results are generally applicable to other time periods; (2) in order to evaluate the performance of photochemical models, it is important to assess a wide range of meteorological conditions including low ozone conditions; and (3) vegetation damage is thought to occur over the growing season (typically several months), so that models must be able to evaluate how changes in emissions and meteorology affect average and peak ambient ozone concentrations over a similar time period.

Similarly, human health effects studies show that longer term exposures to lower concentrations may have adverse effects. Studies in the past have looked at short-term exposures to high ozone concentrations. The National

Research Council reports that controlled human studies and field health studies have indicated the need to consider a period longer than one hour for exposure to ozone (NRC 1991). Clearly, there is a need for an ozone model capable of predicting ozone concentrations over periods of several weeks to several months.

The most readily available computerized models which may be used to simulate ozone formation from power plants are the Reactive Plume Model (RPM), the Urban Airshed Model (UAM) and the Ozone Isopleth Plotting Mechanism Model (OZIPM-4) developed by Systems Applications Incorporated (SAI) (EPA 1991; EPA 1989; SAI 1989; SAI 1987). The RPM and OZIPM-4 models are trajectory models and the UAM model is a grid based model. The RPM and UAM models typically require extensive input data and require the use of either a mainframe or minicomputer. At the inception of this study, the OZIPM-4 model was the only personal computer based model. All three models incorporate the Carbon Bond IV (CB-4) chemical kinetic mechanism which contains 81 chemical and photochemical reactions used to simulate the interactions of 34 species that influence the formation of ozone (NAPAP 1990; Gery et al. 1989).

For this analysis, the OZIPM-4 model was incorporated into a new model (Mapping Area-wide Predictions of Ozone, MAP-O_3) which can be used to predict ozone concentrations: (1) throughout a study area, providing more spatial resolution than trajectory models and (2) over longer time periods (e.g., several months to the ozone season), providing temporal resolution not found in other ozone models. The OZIPM-4 model was selected for this analysis for several reasons: (1) the model incorporates the same chemical mechanism as other ozone models (CB-4); (2) the relatively short run-time of the OZIPM-4 model makes it possible to predict seasonal ozone concentrations that would not be feasible with the other models (a one-day simulation with the OZIPM-4 model takes less than one minute on most personal computers whereas single-day simulations with the UAM model can take one day and longer on a mainframe computer); (3) the implementation of a personal computer model requires significantly less commitment of resources than do models which require either mainframe computers or minicomputer work stations; (4) preliminary modeling using OZIPM-4 has shown it to be sufficiently robust to predict ozone depletion within the plume, where ozone concentrations are below ambient concentrations,

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as well as areas where ozone concentrations are 20 to 30 ppb above ambient concentrations ('ozone bulges'); and (5) OZIPM-4 modeling results to date compare favorably to measurement studies using aircraft flights through power plant plumes (McIlvaine 1994).

The OZIPM-4 model was developed to simulate ozone formation in urban plumes for the purpose of computing VOC emissions reductions necessary to attain the NAAQS for ozone. There is a common misconception that the OZIPM-4 model cannot be used to simulate ozone formation from large point-source emissions. For example, Hogo and Gery (1988) state that the model should not be applied to rural areas or to the development of control strategies for single or small groups of emission sources, unless special attention is given to current limitations and assumptions. However, as discussed below, the OZIPM-4 model can, in fact, be used to simulate ozone concentrations due to NO_x emissions from a single source.

There is a common misconception that the OZIPM-4 model cannot be used to simulate ozone formation from large point-source emissions.

The OZIPM-4 model combines simplified meteorological assumptions with a chemical kinetic mechanism to mathematically simulate physical and chemical processes occurring in the atmosphere. In the OZIPM-4 model, a column of air containing ozone and precursors is transported along a trajectory. Figure 2 is a sketch of the column of air in the OZIPM-4 model that is used to simulate the formation of ozone due to emissions from an urban plume. Typically, in an urban simulation the column is initialized with anthropogenic emissions of NO_x and non-methane organic compounds (NMOC) during the period from 6 to 9 a.m. As the column is transported downwind of the urban area, it may encounter fresh precursor emissions of anthropogenic and/or biogenic origin, that are mixed uniformly within the column. The column extends from the earth's surface through the mixed layer. As the column moves, the height of the column increases, (i.e. its volume increases) and air above the surface layer is mixed in. Concentrations of NMOC species, NO_x and ozone within the column are physically decreased by dilution due to the rise in mixing height and are increased by the addition of fresh emissions from ground level and by entrainment of any pollutants contained in the air aloft (EPA 1989).

Figure 3 is a sketch of the column of air in the OZIPM-4 model used to simulate ozone formation due to emissions from a *large point source plume*. Initially, the column height is defined by the depth of the plume. Within less

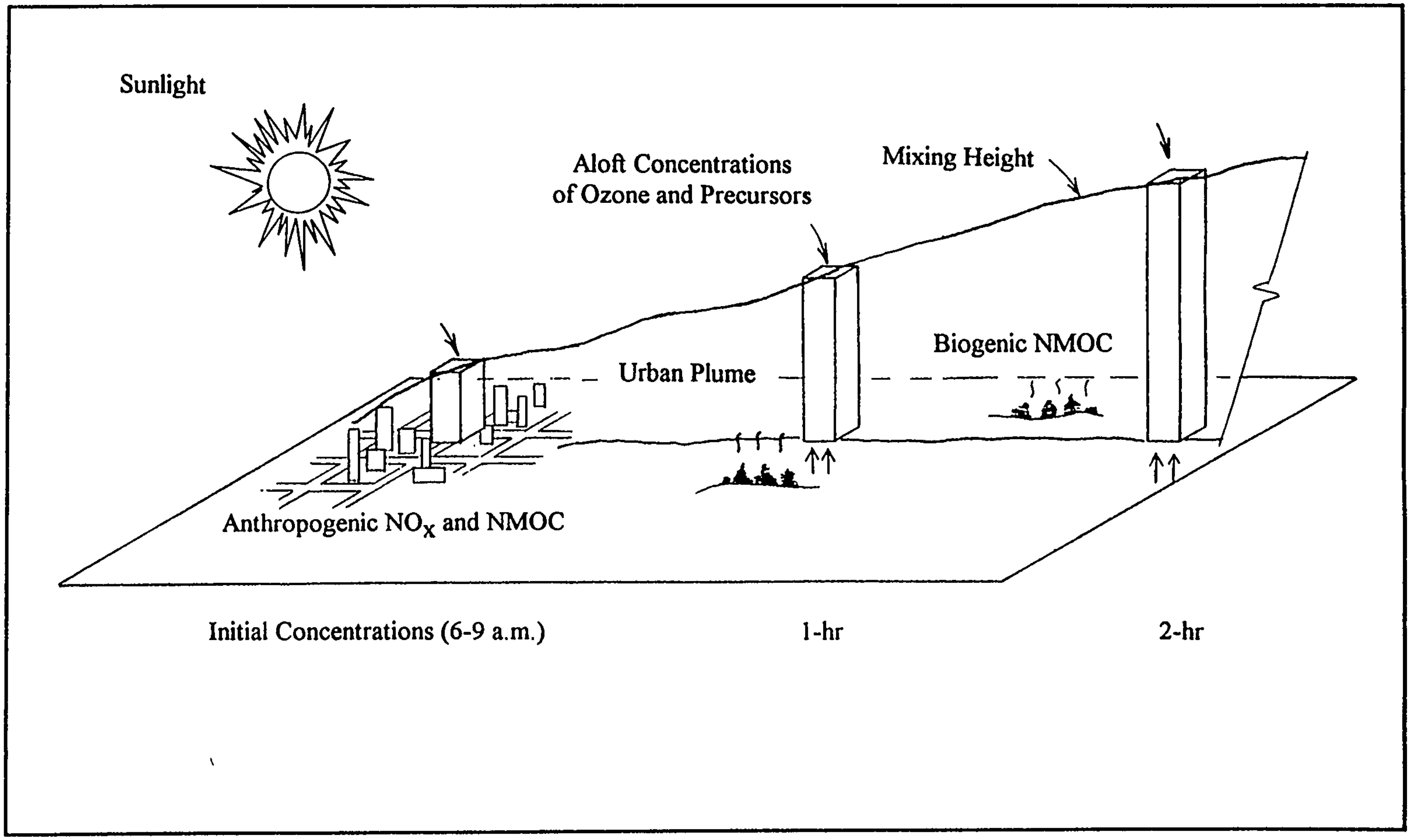

Fig. 2. Sketch of the column of air in the OZIPM-4 model that is used to simulate the formation of ozone due to emissions from an urban plume
Source: **C. M. McIlvaine, Ph.D. dissertation (1994).**

than one hour of travel time on sunny summer days, the plume (i.e. the column) mixes to the ground due to vertical mixing caused by solar heating. After this time the fate of the column is identical to that of one which originates in an urban area (i.e. encountering additional precursor emissions either at ground level or aloft, as it travels downwind). Refer to Section 3.5 and 3.6 for a more detailed discussion of the relationship between the plume and the mixing height as it is simulated in the OZIPM-4 model. McIlvaine (1994) provides a detailed discussion of the use of the OZIPM-4 model to simulate ozone formation in power plant plumes as well as a model validation exercise in which ozone concentrations predicted with the OZIPM-4 model are compared to ozone measurements made in the plume of a large power plant.

2. CONCEPTUAL DISCUSSION OF THE MAP-O_3 MODELING METHODOLOGY

This section is intended to provide a general description of the MAP-O_3 ozone model and modeling methodology that were developed for the fuel cycle analysis. The MAP-O_3 modeling methodology was applied to the coal-fired power plant located at the Southeast Reference site. Figure 4 is a flowchart of the modeling methodology used for this study. The entire modeling methodology is referred to as the MAP-O_3 modeling methodology. This includes the adaptation of the OZIPM-4 model to predict ozone concentrations due to emissions from power plants, as well as the actual development of the new MAP-O_3 model. The MAP-O_3 model predicts area-wide ozone concentrations over the ozone season, by combining ozone concentrations predicted with the OZIPM-4 model with plume trajectories calculated from wind speed and direction measurements. The MAP-O_3 model is also used to predict seasonal average ozone concentrations, as well as daily peak ozone concentrations over the ozone season throughout the study area.

As discussed in the introduction, the effect of power plant NO_x emissions on ozone concentrations is a complex function of meteorological conditions, hydrocarbon concentrations (due to manmade and/or natural hydrocarbon emissions), as well as, ambient concentrations of ozone and ozone precursors. Since the various combinations of these conditions is unique for each day, the task of predicting ozone concentrations over a period of several months is complex and time-consuming. One alternative to modeling each unique day of the ozone season is to model a few days which represent the range of conditions expected to occur over the time period of interest. This approach was chosen for this analysis.

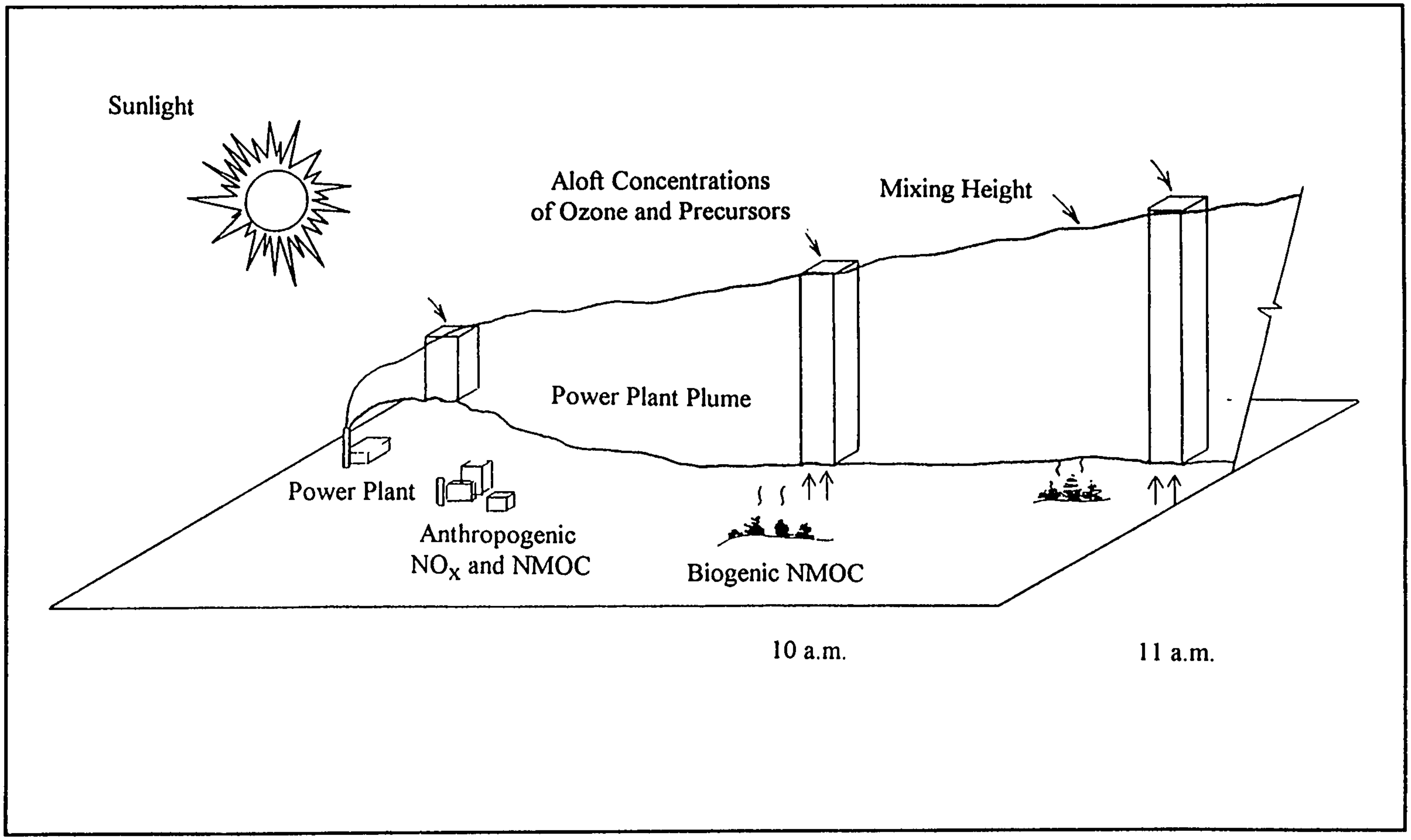

Fig. 3. Sketch of the column of air in the OZIPM-4 model that is used to simulate the formatin of ozone due to emissions from a large point source plume
Source: **C. M. McIlvaine, Ph.D. dissertation (1994)**

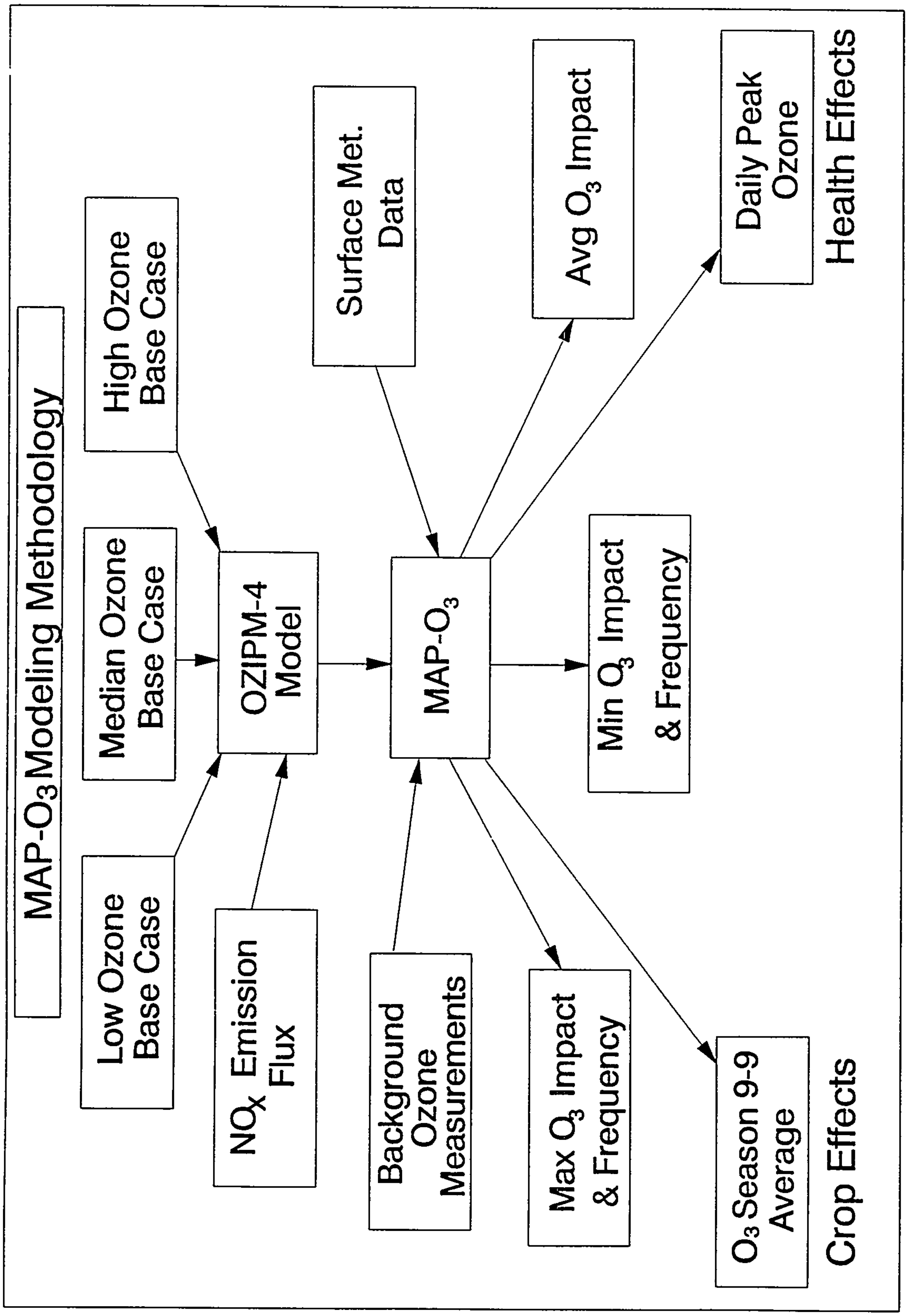

Fig. 4. Flowchart for the MAP-0_3 modeling methodology

A range of parameters that are characteristic of conditions which result in low, median and high ozone concentrations were identified from a case analysis of ambient ozone monitoring data and the corresponding meteorological observations. These parameters were used in the EPA OZIPM-4 model to predict existing ozone concentrations at the Southeast Reference site (without the power plant) for three composite base case days. These three base case scenarios were then used in the OZIPM-4 model to predict ozone concentrations expected to occur as the result of the power plant NO_x and NMOC emissions on high, median and low ozone days. The difference between the base case simulations and the plant simulations is the increment of ozone due to the plant emissions under high, median and low ozone conditions.

The difference between the base case simulations and the plant simulations is the increment of ozone due to the plant emissions under high, median and low ozone conditions.

Each day of the ozone season was identified as either a 'high', 'median' or 'low' ozone day according to the peak daily ozone concentration that was measured at a nearby monitoring station on that day. This typing scheme, together with the hourly ozone concentrations due to the plant emissions, predicted for each of three composite ozone days, resulted in predicted hourly ozone concentrations for each hour of each day of the ozone season. A new model developed for this study (MAP-O_3) was used to predict the location of each ozone concentration predicted with the OZIPM-4 model and to calculate the longer-term ozone concentrations needed for this analysis. The MAP-O_3 model calculates the path of the power plant plume (trajectory) from meteorological surface observations of wind speed and direction, for each day of the ozone season. The plume trajectories are combined with the hourly ozone concentrations to provide a map of ozone concentrations occurring in the vicinity of the power plant. The MAP-O_3 model also calculates the peak one-hour ozone concentration for each day of the ozone season and the seasonal average 9 a.m. to 9 p.m. ozone concentration.

Results from the MAP-O_3 model are transferred to an isopleth plotting routine (e.g., SURFER, Deltagraph or others) which generates isopleth maps showing the distribution of ozone concentrations (both above and below ambient ozone concentrations) due to emissions of NO_x and NMOC from the power plant.

3. MAP-O_3 MODELING METHODOLOGY

3.1 METHODOLOGY OVERVIEW

The MAP-O_3 modeling methodology and the assumptions upon which it is based are quite simple, yet the justification of these assumptions is somewhat lengthy. This section describes the methodology and the assumptions upon which it is based, as well as the data used in the modeling. Section 4 presents the modeling results and Section 5 provides a list of the modeling assumptions and limitations as discussed in depth in Section 3. Section 6 provides a discussion of the expected impact from power plant emissions located at the Southwest Reference site.

The first step in the modeling methodology is to develop base case conditions which will be used to simulate ozone concentrations at the Southeast Reference site without the power plant emissions. A description of the EPA OZIPM-4 model used to predict these ozone concentrations is given in Section 3.2. The development of the three base case scenarios is discussed in Section 3.3. Next, the base case modeling results are calibrated against monitored ozone concentrations as discussed in Section 3.4. The assumption of atmospheric mixing in the OZIPM-4 model is discussed in Section 3.5 and Section 3.6 outlines the calculation of emission fluxes of NO_x and NMOC from the power plant. Next, the base case scenarios together with the plant emissions fluxes are used to predict ozone concentrations with the OZIPM-4 model as a function of the plume age and hour that the plume first mixes to the ground as outlined in Section 3.7. Section 3.8 discusses the MAP-O_3 model and how it is used to predict daily peak and seasonal average ozone concentrations throughout the study area for the ozone season. Steps in the MAP-O_3 modeling methodology are outlined here. Refer to Fig. 4 for a flowchart of the MAP-O_3 modeling methodology.

1. Develop three base case scenarios to simulate the range of conditions occurring over the ozone season. (Section 3.3)

    a. Obtain measured ozone concentrations and meteorological observations (e.g., wind speed, direction, temperature, relative humidity, sky cover, mixing height and stability class) in the vicinity of the power plant for the ozone season.

    b. Use the measured ozone concentrations to identify 1) the ten days in the ozone season with the highest peak daily ozone concentrations, 2) the ten days in the ozone season with median peak daily ozone concentrations, and 3) the ten days in the ozone season with the lowest ozone concentrations. Prepare composite

10-day average meteorological conditions that correspond to high, median and low ozone conditions.

c. Use the three 10-day average composite meteorological conditions, together with estimates of biogenic emissions, anthropogenic emissions, solar radiation and boundary conditions (of ozone and ozone precursors) to simulate three base case scenarios (i.e. simulate the formation of ozone under conditions of high, median and low ozone formation in the study area without the addition of power plant emissions).

2. Calibrate the base case simulations by comparing predicted ozone concentrations with monitored ozone concentrations under the same conditions, and make any necessary adjustments to boundary conditions. (Section 3.4)

3. Calculate NO_x and NMOC emissions fluxes from the power plant. (Section 3.6)

4. Using the three base case scenarios developed in step (1) and the calculated NO_x and NMOC emissions fluxes from the power plant, predict ozone concentrations with the OZIPM-4 model as a function of the time that the plume first mixes to the ground and the age of the plume. (Section 3.7)

5. Use the MAP-O_3 model, together with measured peak daily ozone concentrations and surface observations of wind speed and direction, to map ozone concentrations throughout the study area for each hour of each day of the ozone season. (Section 3.8)

a. Read a file of measured peak daily ozone concentrations to determine whether each day is a high, median or low ozone day.

b. Using surface observations of wind speed and direction, determine the path of the power plant plume (trajectory) for each hour of each day.

c. Combine the predicted trajectories of the plume with predicted ozone concentrations from OZIPM-4 according to whether each day is designated as a high, median or low ozone day.

d. Calculate daily peak and seasonal average ozone concentrations at each location in the polar receptor grid.

e. Convert results to Cartesian coordinates for import to graphing routines.

3.2 OZIPM-4 MODEL

The OZIPM-4 model is a photochemical trajectory model which combines simplified meteorological assumptions with a chemical kinetic mechanism to mathematically simulate physical and chemical processes occurring in the atmosphere. In the OZIPM-4 model, a column of air containing ozone and precursors is transported along a trajectory. As the column moves, it encounters fresh precursor emissions that are mixed uniformly within the column. The column extends from the earth's surface to the mixing height. The mixing height is defined as the top of a surface-based layer of air (known as the mixed layer or the surface layer) which is well mixed due to mechanical and/or thermal turbulence. Typical values of the mixing height for the east Tennessee area range from 450 meters for the mean annual morning mixing height to 1,600 meters for the mean annual afternoon mixing height, (Holtzworth 1972). The air within the column is assumed to be uniformly mixed at all times (EPA 1989).

At the beginning of a simulation, the column contains some specified initial concentration of NMOC and NO_x. As the column moves along the trajectory, the height of the column can change because of diurnal variations in mixing height. As the height of the column increases, its volume increases, and air above the surface layer is mixed in. Pollutants above the mixed layer are described as "transported above the surface layer" or "transported aloft." Any ozone or ozone precursors above the mixed layer that are entrained into the column as it expands are rapidly mixed throughout the column (EPA 1989).

Concentrations of NMOC species, NO, NO_2 and O_3 within the column are physically decreased by dilution due to the rise in mixing height, and physically increased by the addition of fresh emissions from ground level and by entrainment of pollutants transported aloft. All species react chemically according to the Carbon Bond IV (CB-4) kinetic mechanism (EPA 1989). Certain photolysis rates within that mechanism are functions of the intensity and spectral distribution of sunlight, and they vary diurnally according to time of year and location.

Input parameters that can be specified by the user include:

- Latitude and Longitude;
- Time zone;

- Date;
- Morning and afternoon mixing heights;
- Hourly temperature variation;
- Hourly atmospheric moisture estimates;
- Concentrations of NMOC, NO_x, CO, ozone and up to ten other species in the air above the mixed layer due to transport aloft;
- Concentrations of NMOC, NO_x, CO, ozone and up to ten other species transported in the surface layer;
- VOC, CO and NO_x emissions at each hour;
- Organic reactivity; and
- Biogenic emission rates and speciation.

Three types of output can be requested from the model. For the purposes of this study, OZIPM-4 was used to estimate ozone concentrations as a function of time for a single set of precursor conditions. Use of the OZIPM-4 model to simulate the formation of ozone from a power plant plume is discussed in Sections 3.5 and 3.6. The model's abilities to predict zones of ozone depletion and zones of ozone formation are discussed in Section 3.7.

3.3 BASE CASE SCENARIOS

Simulating ozone formation from power plant emissions involves two steps. First, a computer simulation is run using a set of input conditions that characterizes the study area (base case) and then the same set of input conditions is run again with the addition of the plant emissions (plant case). The difference in ozone concentrations between the two runs is the increment of the ozone concentration that is due to the plant emissions.

The magnitude of the impact of NO_x emissions from the plant on ozone concentrations is a function of the area (background conditions) in which the plant is located. Factors which affect the impact of the plant emissions include meteorological conditions, biogenic NMOC emissions, anthropogenic NMOC and NO_x emissions, and ambient precursor concentrations in and above the surface layer. These factors vary throughout the day, as well as throughout the ozone season.

Typically, ozone assessment studies are designed to determine the worst-case impact on ozone concentrations and for this reason most studies are limited to model inputs which are characteristic of high ozone days (i.e. low wind speeds, high temperatures, high precursor concentrations). In order to evaluate the performance of the OZIPM-4 model, and to simulate ozone formation over the entire ozone season, a wide range of background conditions was used to develop model inputs. Meteorological observations and ozone monitoring data were used to derive the model inputs (i.e. temperature, mixing height, biogenic emissions, and precursor concentrations transported at the surface and aloft) which were used to predict low, median and high base case ozone concentrations, (i.e. three separate base case scenarios using the high, median and low ozone conditions were developed).

In order to avoid having to simulate ozone formation for each day of the ozone season (153 days) these three base case scenarios were used to simulate the formation of ozone over the ozone season. Each day of the season was defined as either a 'low', 'median', or 'high' ozone day based on the peak one-hour ozone concentration measured at the Knoxville, Rutledge Pike monitor, for each day. Figure 5 is a cumulative frequency distribution of the daily peak one-hour ozone concentrations at the Rutledge Pike monitoring station during the period May 1 to September 30, 1990. The cumulative frequency distribution is plotted as the log of the daily peak ozone concentration versus probit (probits are standard deviations coded by the addition of five to avoid negative values; probit 5.0 corresponds to a cumulative frequency of 50%; probit 6.0 corresponds to a cumulative frequency of 84.13% and probit 3.0 corresponds to a cumulative frequency of 2.27%). The Rutledge Pike monitoring station is designated by EPA as a rural site and is part of the EPA Aerometric Information Retrieval System (AIRS) database. The monitor is located approximately 60 km from the Southeast Reference site and approximately 20 km northeast of Knoxville, TN. Each day in the ozone season was assigned a designation of either low, median or high by assuming that the top third days of the cumulative frequency distribution of ozone concentrations are 'high' days, the middle third of the days are 'median' days and the lower third are 'low' days.

3.3.1 Meteorology

Figure 6 shows the relationship between peak daily temperatures and peak daily one-hour ozone concentrations at the Knoxville, Rutledge Pike monitoring station for each day of the 1990 ozone season (May 1 to September 30). The ten days with the highest daily peak one-hour ozone concentrations are

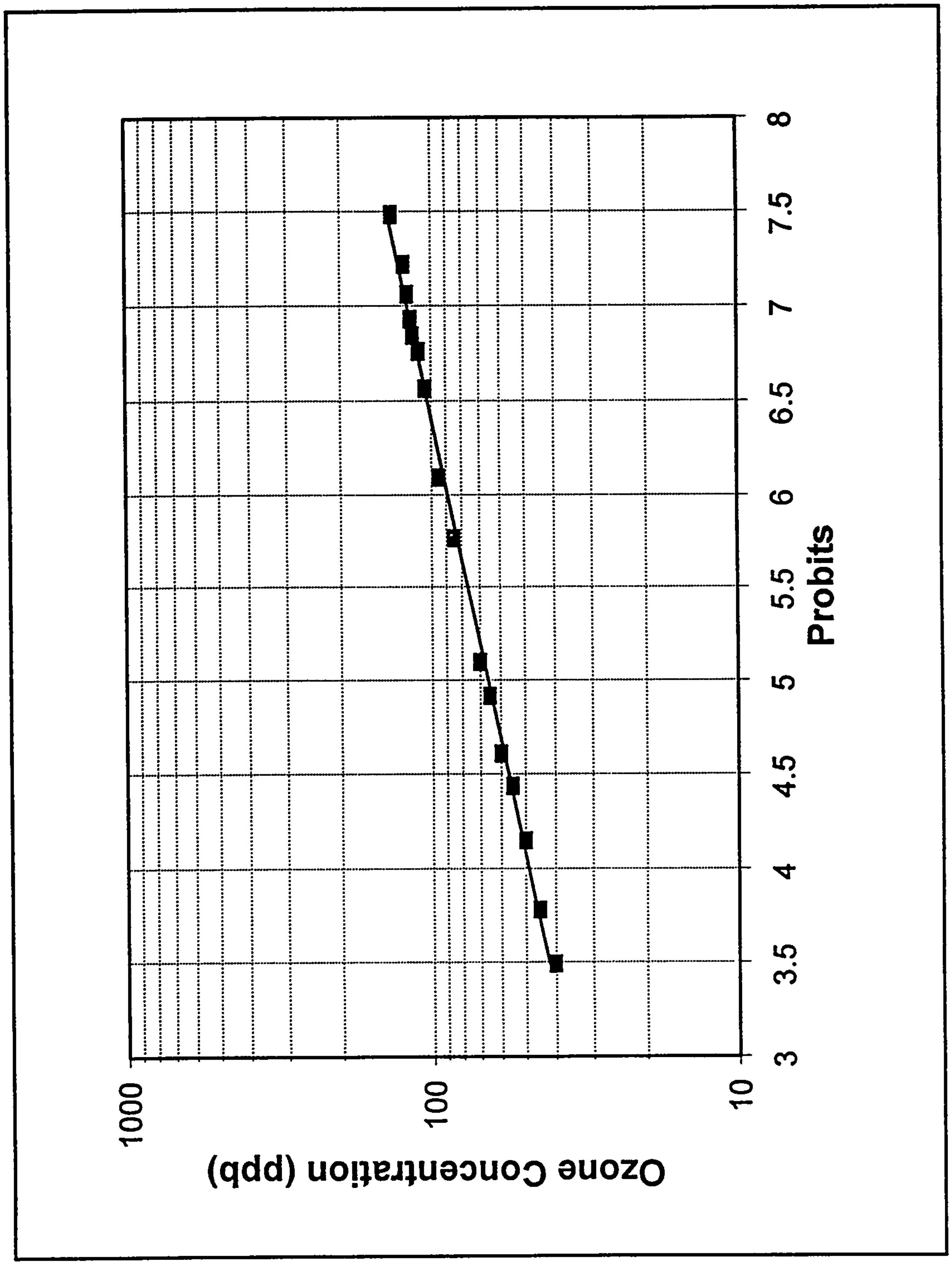

Fig. 5. Monitored daily peak ozone concentration versus probit for the 1990 ozone season at Knoxville, TN

shown in the figure with circles; the ten days with the ten median ozone concentrations are shown with squares and the ten days with the ten lowest ozone concentrations are shown in the figure with triangles. The meteorological surface observations of sky cover, relative humidity, temperature, and wind speed for each day were obtained from the National Climatic Center hourly surface observations data files for the Knoxville, McGhee Tyson Airport for 1990. These values were averaged to develop three composite, 10-day average, low, median and high ozone days. Hourly average observations for each composite day are shown in Tables 2, 3 and 4.

Figure 7 shows the hourly temperature variation for each of the composite days. As shown in Fig. 7 the highest daily one-hour average temperature for the high ozone composite day was 93 °F (34.0 °C), the highest daily one-hour average temperature for the median ozone composite day was 83 °F (28.3 °C) and the highest daily one-hour average temperature for the low ozone composite day was 67 °F (19.4 °C). As we saw in Fig. 6, there is a strong correlation between the highest daily one-hour average temperature and the highest daily one-hour average ozone concentration. The range of temperatures used in developing the input parameters for each of the three base case scenarios should provide a good measure of the model's ability to predict ozone under widely varying conditions of ozone formation.

...there is a strong correlation between the highest daily one-hour average temperature and the highest daily one-hour average ozone concentration.

Also shown in Tables 2, 3 and 4, are the hourly mixing heights calculated from the OZIPM-4 model. The model accepts morning and afternoon mixing height observations and calculates the hourly mixing heights based on an assumed initial time of mixing height rise of 8 a.m. and a time that maximum mixing height is attained of 3 p.m. The morning and afternoon mixing height observations were obtained from the National Climatic Center for Nashville, TN for each day of the three groups of ten days. The median morning mixing height and the median afternoon mixing height were selected for each composite day. These median mixing heights were then input to the OZIPM-4 model, which calculated the hourly mixing heights shown in Tables 2, 3 and 4.

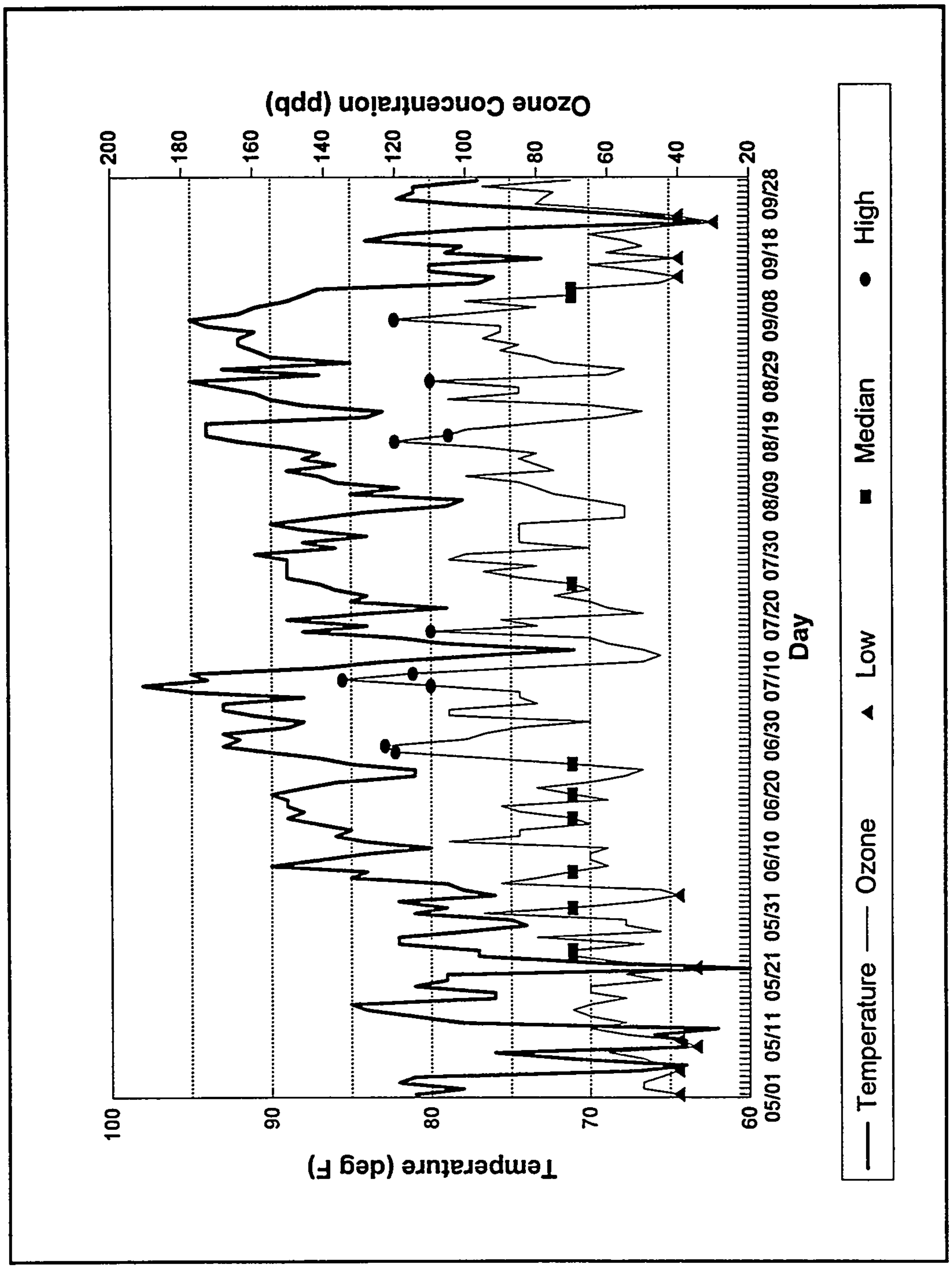

Fig. 6. Relationship between the maximum daily one-hour average temperature and maximum daily one-hour average ozone concentration for the 1990 ozone season at Knoxville, TN

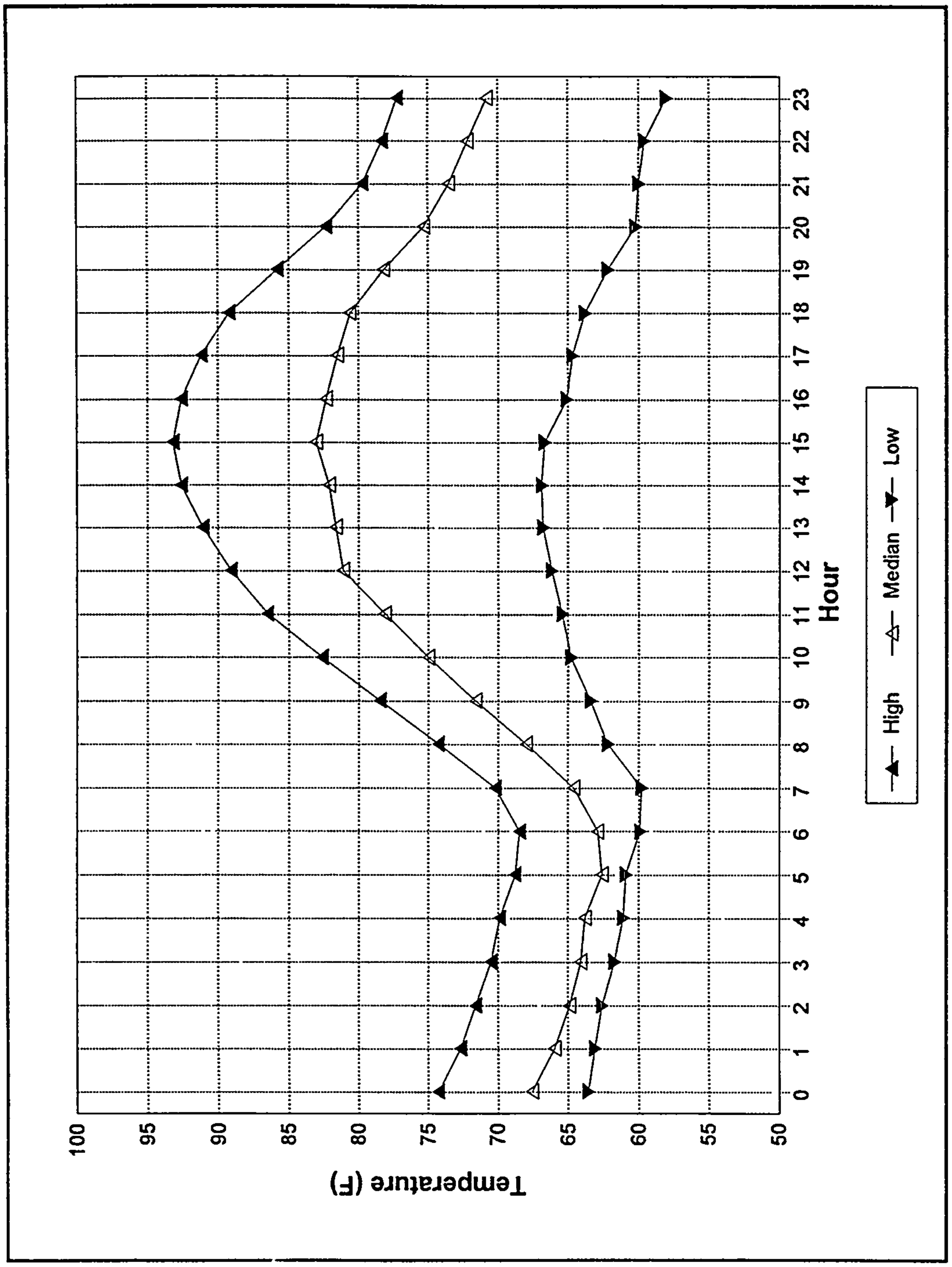

Fig. 7. Ten-day average hourly temperatures for each composite base case day

Table 2. Hourly average observations for the 10-high ozone days during 1990

Begin Hour	Sky Cover	Relative Humidity	10-meter Wind Speed (m/s)	Temp. (C)	Temp. (F)	Mixing Height (m)
0	0.1	0.78	1.0	23.5	74	
1	0.1	0.82	1.1	22.6	73	
2	0.1	0.85	1.2	22.0	72	
3	0.1	0.88	0.5	21.4	71	
4	0.2	0.90	0.9	21.1	70	
5	0.2	0.92	0.8	20.4	69	
6	0.2	0.92	0.6	20.3	69	98
7	0.2	0.88	1.0	21.2	70	98
8	0.2	0.77	1.3	23.5	74	98
9	0.1	0.69	2.2	25.8	79	323
10	0.2	0.62	2.4	28.1	83	685
11	0.1	0.54	2.2	30.3	87	1082
12	0.1	0.49	2.8	31.7	89	1414
13	0.1	0.46	2.8	32.8	91	1650
14	0.1	0.41	2.3	33.7	93	1818
15	0.2	0.39	2.8	34.0	93	1946
16	0.3	0.39	2.3	33.7	93	1948
17	0.2	0.43	2.3	32.9	91	1948
18	0.1	0.46	2.1	31.8	89	1948
19	0.3	0.54	2.5	29.9	86	1948
20	0.3	0.60	2.1	27.9	82	1948
21	0.2	0.67	2.1	26.5	80	1948
22	0.2	0.71	1.4	25.7	78	1948
23	0.2	0.74	1.2	25.1	77	1948

Table 3. Hourly average observations for the 10-median ozone days during 1990

Begin Hour	Sky Cover	Relative Humidity	10-meter Wind Speed (m/s)	Temp. (C)	Temp. (F)	Mixing Height (m)
0	0.2	0.80	1.3	19.8	68	
1	0.2	0.85	1.2	18.9	66	
2	0.2	0.88	1.3	18.3	65	
3	0.3	0.89	1.0	17.8	64	
4	0.3	0.90	1.6	17.7	64	
5	0.3	0.93	1.4	17.0	63	
6	0.4	0.93	1.3	17.2	63	120
7	0.3	0.88	1.3	18.1	65	120
8	0.2	0.83	1.4	20.0	68	120
9	0.2	0.75	1.9	22.0	72	264
10	0.3	0.68	1.8	23.9	75	522
11	0.5	0.60	2.8	25.6	78	820
12	0.4	0.53	3.6	27.3	81	1075
13	0.4	0.51	3.1	27.6	82	1256
14	0.3	0.51	3.1	27.8	82	1383
15	0.3	0.49	2.8	28.3	83	1478
16	0.4	0.51	2.5	27.9	82	1479
17	0.4	0.53	3.0	27.5	82	1479
18	0.4	0.54	3.1	27.0	81	1479
19	0.3	0.61	2.4	25.7	78	1479
20	0.3	0.67	2.4	24.1	75	1479
21	0.4	0.71	2.4	23.1	74	1479
22	0.3	0.73	1.9	22.3	72	1479
23	0.3	0.74	2.2	21.6	71	1479

Table 4. Hourly average observations for the 10-low ozone days during 1990

Begin Hour	Sky Cover	Relative Humidity	10-meter Wind Speed (m/s)	Temp. (C)	Temp. (F)	Mixing Height (m)
0	0.2	0.83	2.8	17.6	64	
1	0.1	0.85	2.6	17.3	63	
2	0.4	0.86	2.3	17.0	63	
3	0.3	0.89	2.1	16.5	62	
4	0.2	0.90	2.0	16.1	61	
5	0.2	0.90	2.0	16.1	61	
6	0.1	0.89	1.8	15.5	60	861
7	0.2	0.90	2.3	15.4	62	861
8	0.2	0.83	2.4	16.8	62	861
9	0.2	0.78	3.1	17.4	65	889
10	0.2	0.72	3.5	18.2	65	966
11	0.3	0.69	3.5	18.6	65	1071
12	0.3	0.67	3.5	19.0	66	1162
13	0.4	0.66	3.7	19.3	67	1222
14	0.1	0.65	3.9	19.4	67	1263
15	0.3	0.66	4.1	19.3	67	1292
16	0.2	0.67	3.8	18.4	65	1292
17	0.2	0.69	3.4	18.2	65	1292
18	0.3	0.69	3.4	17.7	64	1292
19	0.3	0.73	2.7	16.8	62	1292
20	0.2	0.79	2.3	15.7	60	1292
21	0.2	0.79	2.4	15.6	60	1292
22	0.2	0.78	3.1	15.3	60	1292
23	0.2	0.82	2.1	14.4	58	1292

3.3.2 Biogenic Emissions

Vegetation emits hydrocarbons, especially during the day. The rate of emission increases significantly with increasing temperature. These emissions contribute to the formation of ozone and should be included in any OZIPM-4 simulation (EPA 1989). In addition to temperature, biogenic emissions estimates are sensitive to wind speed, relative humidity and cloud cover. Biogenic emissions within the vicinity of the Southeast site were estimated with the EPA Personal Computer Version of the Biogenic Emissions Inventory System (PC-BEIS) computer program. The PC-BEIS computer program allows users to estimate hourly emissions of isoprene, a-pinene, other monoterpenes and unidentified hydrocarbons for any county in the contiguous United States. These emissions estimates are adjusted for user input meteorological parameters (Pierce and Baugues 1991). The hourly observations of sky cover, temperature, relative humidity and wind speed for each composite base case day (Tables 2, 3 and 4) were used as inputs to the model. Hourly emissions fluxes of isoprene, a-pinene, and monoterpene were calculated with the PC-BEIS program for each composite day. These values were then used as inputs to the OZIPM-4 model. Table 5 shows the total daily biogenic emissions for each composite base case day. Figure 8 shows the hourly emissions fluxes of isoprene for each composite day.

3.3.3 Anthropogenic Emissions

Anthropogenic emissions are any man-made emissions of NO_x or NMOC within the vicinity of the plant. These emissions were estimated from an existing emission inventory for middle and west Tennessee. Anthropogenic emissions estimates for five counties in middle and west Tennessee for 1987 were obtained from the "Comprehensive Emission Inventory for Precursors of Ozone in the Nashville and Memphis Tennessee Areas," 1990 prepared by the University of Tennessee. These emissions estimates include all categories of anthropogenic emissions, i.e. mobile and stationary point and area sources and are based on the latest methods recommended by the U.S. EPA for developing comprehensive ozone precursor inventories.

Thirteen counties were identified within a 50 kilometer radius of the Southeast Reference site. These counties are considered representative of the entire study area. Since current emissions estimates for the 13 counties are not available, the emissions estimates from the five middle and west Tennessee counties were used to develop average emissions per capita factors for NMOC and NO_x.

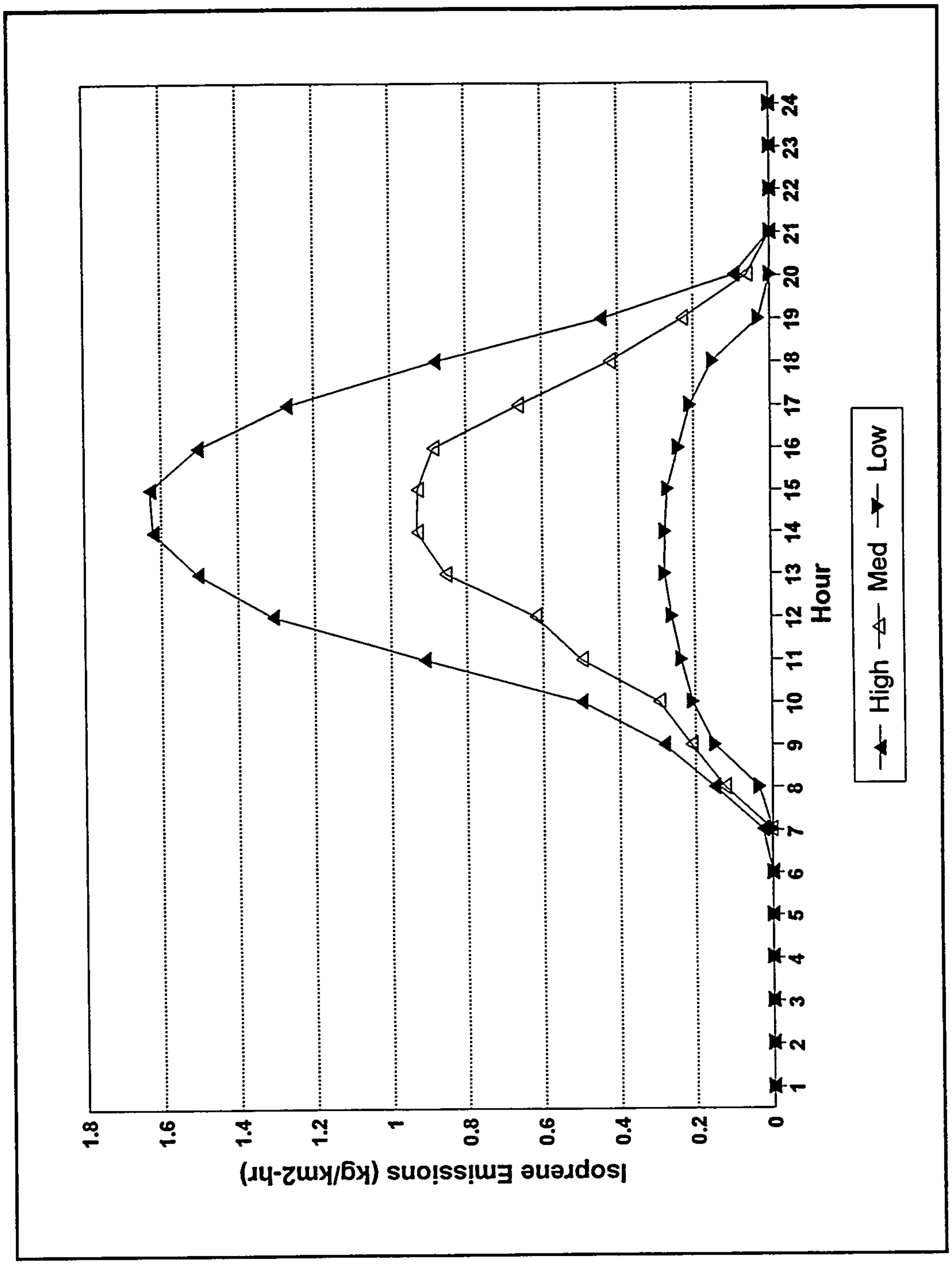

Fig. 8. Hourly isoprene emission flux for each composite base case day

Table 5. Total daily biogenic NMOC emissions at the Southeast Reference site

Biogenic Emissions (kg/km^2-day)			
Base Case	Isoprene	Alpha-pinene	Monoterpene
High	11.6	1.59	1.77
Median	6.44	1.20	1.29
Low	2.32	0.739	0.759

These factors were then applied to the 1980 population in each of the 13 counties within the vicinity of the Southeast Reference site. See Table 6. (Population data for 1990 was unavailable at the time emissions were calculated, however, the average increase in population in the 13 counties was 4.7% from 1980 to 1990.) The area of each county was used to calculate emissions densities (kilograms per square kilometer per hour) that are required for the OZIPM-4 model. The average emission densities of NMOC and NO_x (1.311 and 0.936 kg/km^2-hr) were calculated for the 13 counties and these emission densities were used in the OZIPM-4 model, assuming a uniform distribution of anthropogenic emissions throughout the day and throughout the study area. While this method of calculating anthropogenic emissions of NMOC and NO_x is fairly crude, it is used only to model the relative change in ozone concentrations, with and without the power plant emissions.

3.3.4 Solar Radiation

Reactions for several species (e.g. formaldehyde and NO_2) in the chemical kinetic mechanism (Carbon Bond IV) in the OZIPM-4 model are photochemical reactions whose rates are dependent on solar intensity. The rate of absorption of radiation by a particular molecule depends on both the wavelength distribution and the intensity of radiation. Because radiation in the lower atmosphere is attenuated by absorption, scattering and reflection, the solar intensity distribution at ground level varies as a function of solar angle (Seinfeld 1975). Solar angle and therefore solar intensity, is a function of time of day, latitude and time of year. This variation in solar intensity explains the seasonal and diurnal variation characteristic of many atmospheric photochemical reactions.

Table 6. Average emission densities for 13-counties in East Tennessee based on average per capita emissions for five counties in Middle and West Tennessee

County	Area (sq. miles)	1980 Population	Countywide Emissions (tons/day) NMOC	Countywide Emissions (tons/day) NO_x	Countywide Emissions (kg/capita-day) NMOC	Countywide Emissions (kg/capita-day) NO_x	Grid Emissions (kg/km^2-hr)** NMOC	Grid Emissions (kg/km^2-hr)** NO_x
Middle and West Tennessee:								
Cheatham	307	25,412	8.85	5.43	0.316	0.194	0.84	0.52
Dickson	491	30,037	13.25	7.4	0.400	0.223	0.79	0.44
Robertson	476	37,021	15.97	13.7	0.391	0.336	0.98	0.84
Davidson	526	477,811	156	117	0.296	0.222	8.66	6.49
Shelby	786	777,113	232	176	0.271	0.205	8.62	6.54
Average kg/capita-day:					0.335	0.236		
East Tennessee:								
Anderson	354	67,346					2.10	1.48
Blount	566	77,770					1.48	1.04
Campbell	496	34,923					0.76	0.53
Cumberland	684	28,676					0.45	0.32
Knox	526	319,694					6.55	4.62
Loudon	247	28,553					0.98	0.84
McMinn	432	41,878					1.04	0.74
Meigs	217	7,431					0.37	0.26
Monroe	653	28,700					0.47	0.33
Morgan	523	16,604					0.34	0.24
Rhea	337	24,235					0.77	0.55
Roane	395	48,425					1.32	0.93
Scott	533	19,259					0.39	0.27
						Average	1.311	0.936

NMOC - Non-methane Organic Compounds
NO_x - Nitrogen Oxides
** 12 hrs/day

The OZIPM-4 model has no provision for cloud cover, however solar radiation is calculated as a function of latitude, time of day and day of the year. In selecting the calendar day to be modeled for each composite base case scenario, the relative location of high, median and low ozone calendar days during the ozone season was noted from Fig. 6. High ozone days tended to occur during late-June, July and August, while the median days tended to occur in May, early-June and early-September and low ozone days tended to occur in May and September. Figure 9 shows the annual variation in solar radiation reaching the earth's surface at 40° north latitude (as well as, the average number of reports of ozone concentrations 120 ppb in New York and Boston during the period 1983 to 1985). The calendar day to be modeled for each base case was selected using both Figs. 6 and 9. The high ozone day (Fig. 6) nearest the peak solar radiation seen in Fig. 9 was June 28; the low ozone day (Fig. 6) with the lowest solar radiation according to Fig. 9 was September 24 and the median ozone day approximately midway between the highest and lowest solar radiation days selected was July 25. These calendar days were input to the OZIPM-4 model for each of the three base case scenarios in order to simulate a range of solar radiation conditions representative of low, median and high ozone days. The Southeast Reference site latitude and longitude of 35.89 and 84.38, respectively, were also used as inputs to the OZIPM-4 model.

3.3.5 Boundary Conditions

In addition to calendar day, anthropogenic and biogenic emissions and meteorological inputs, the OZIPM-4 model requires inputs that describe the initial concentration of species within the column of air, as well as the concentrations of species above the column (aloft) and surface concentrations of species that may be transported into the column.

Initial Precursor Concentrations

The OZIPM-4 model is sensitive to the initial concentrations of NMOC and NO_x that are present at the surface at the start of a simulation. These concentrations are intended to represent the NMOC and NO_x that are initially present within the mixed layer at the start of the model simulation (normally 8 a.m.). Because of the difficulties associated with estimating these concentrations, as well as the model sensitivity to them, the model simulation was started at 6 a.m. and the initial concentrations were assumed to be essentially zero at this time.

The OZIPM-4 model also requires initial values for biogenic species. Since biogenic emissions are sensitive to ambient temperature and sunlight, they are not expected in large quantities until later in the day (Baugues 1991).

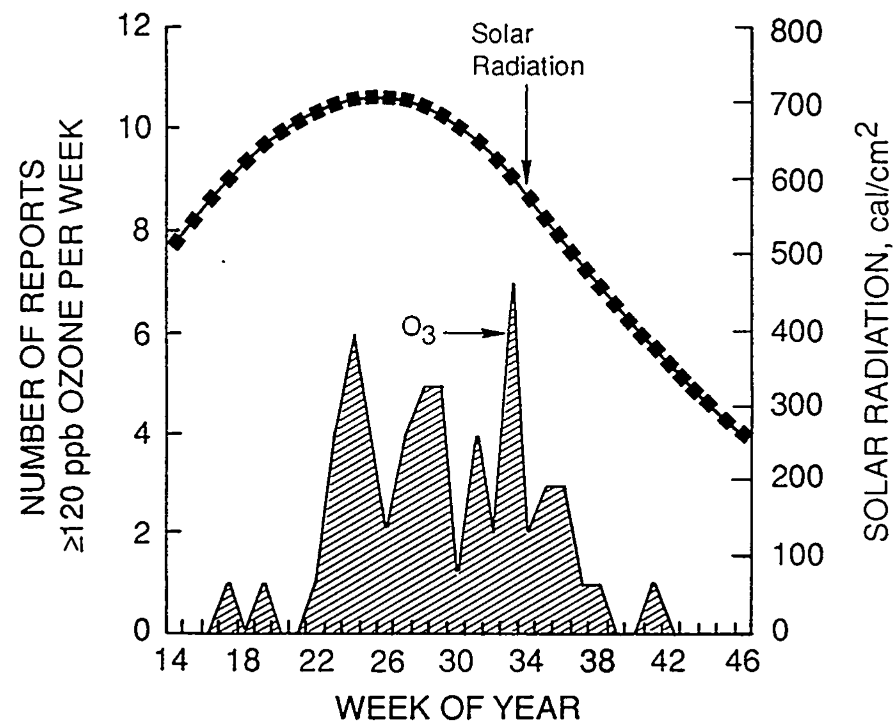

Fig. 9. The average number of reports of ozone concentrations $\geq$ 120 ppb at the combined cities of New York and Boston from 1983 to 1985 (1 April=week 14, 1 May=week 18, June=week 22, 1 July=week 27, 1 August=week 31, 1 September=week 35, 1 October=week 40, 1 November=week 44). A repsentation of the annual variation in solar radiation reaching the earth's surface at 40° N latitude (units, calories/cm$_2$-day) is shown. Average over 1983 - 1985.

Source: **National Research Council (NRC) (1991).**

Therefore, the initial concentration for all three biogenic species was set equal to zero.

Ozone Transport

Two possible mechanisms by which ozone is transported into an area are:

1) Advection (horizontal transport) of ozone along the earth's surface

2) Advection of ozone aloft (typically at night and during early morning hours above the mixed layer, with downward mixing when the height of the mixing layer begins to increase).

Ozone transported at the surface is subject to reactions and scavenging by other species emitted during the night (e.g. nitric oxide). While the scavenging of ozone at the surface may be the predominant mechanism of depletion in urban areas with numerous mobile NO_x sources, nocturnal ozone depletion in three rural areas was shown to be controlled by dry deposition (Kelly 1984).

The primary impact of ozone transport near the surface is expected to be the more rapid conversion of NO to NO_2. Several studies have shown that ozone transported along the surface tends to be minimal. For these reasons, the initial value for ozone transported at the surface (at 6 a.m.) was set equal to zero (EPA 1989).

Because of nighttime atmospheric stability, ozone transported aloft does not come into contact with scavengers emitted at night. Therefore, overnight advection of ozone aloft is the more significant mechanism of transport from one area to another (EPA 1989).

Measurements of ozone aloft were made using aircraft over six cities during the summers of 1985 and 1986. Ninety percent of the ozone aloft values from this study fall between approximately 25 to 60 ppb (EPA 1989). EPA (1991) recommends the use of 40 ppb ozone aloft for UAM simulations of urban areas. For this study area, the ozone aloft value of 40 ppb was used for the high ozone base case. The selection of this value assumes that ozone aloft values on high ozone days in a mostly rural area could be expected to be as high as ozone aloft values seen in some urban areas. An ozone aloft value of zero ppb was used for the median and low ozone base cases. A discussion of this selection of ozone aloft values is provided in Section 3.4.

Precursor Transport

Precursor pollutants can be transported in both the surface layer and aloft. Since the surface layer outside urban areas is relatively shallow, long-range transport of precursors may not be significant. Recent measurements of NMOC aloft over six cities indicates that most NMOC values aloft fall within a range of 0-50 ppb with a median of 30 ppb (EPA 1989) (Baugues 1987). For this study, 22.2 ppb NMOC aloft (EPA 1991) and zero NMOC at the surface were used for the high ozone base case. For modeling ozone concentrations in urban areas and areas downwind of urban areas, EPA recommends using 2 ppb NO_x aloft and zero NO_x at the surface. These values were used for the high ozone base case. The NO_x concentration of 2 ppb accounts for any natural NO_x emissions which, in rural areas, is between 0.2 and 10 ppb (NRC 1991). For both the median and low ozone base cases, NMOC and NO_2 concentrations aloft and at the surface were set equal to zero. A discussion of the precursor aloft concentrations for the median and low base cases is provided in the next section.

3.4 OZIPM-4 MODEL CALIBRATION

In order to assess the reasonableness of the range of inputs developed for each of the base case scenarios, the OZIPM-4 model was calibrated against monitoring data. Daily peak one-hour modeled ozone concentrations for each of the three base case scenarios were compared to daily peak one-hour measured ozone concentrations. The nearest monitoring station, Rutledge Pike, Knoxville, is located approximately 20 km northeast of Knoxville, TN. Since this monitor is located in a predominantly downwind direction of Knoxville during the daytime hours, and since the monitor is approximately 60 km from the Southeast Reference site, the OZIPM-4 model was calibrated using emissions that were characteristic of the Knoxville area (model calibration run). Reasonable agreement was obtained between Knoxville modeled ozone concentrations and Knoxville monitored ozone concentrations, after which, the model was rerun with emissions characteristic of the Southeast Reference site.

For the model calibration run, the hourly emissions flux for Knox county, shown in Table 6, for NMOC and NO_x of 6.55 and 4.62 kg/km^2-hr, respectively, were assumed to occur for twelve hours a day throughout the study area. Biogenic emissions for Knox county were calculated using the PC-BEIS program with the meteorological inputs from the three composite days described above. All other inputs described in the Base Case Scenarios Section, were identical with the exception of aloft ozone and precursor concentrations which will be discussed separately here. The EPA recommended values of aloft concentrations of ozone, NMOC and NO_2 of 40 ppb, 22.2 ppb and 2 ppb (EPA 1991), respectively, were used as model inputs for the high ozone day scenario.

It is assumed that aloft ozone and precursor concentrations on high ozone days in a predominantly rural area could be expected to be as high as concentrations seen aloft in some cities. Therefore these values were selected for the high ozone base case. Results from the Knoxville model calibration runs (discussed below) were used to support the selection of zero aloft concentrations of ozone and precursors for the median and low ozone conditions at the Southeast Reference site.

3.4.1 Knoxville Results

Figure 10 shows the results of the model calibration run for Knoxville. Each I-beam represents the monitored ozone concentrations for each of the ten-day groups. The upper line on each beam represents the highest peak daily ozone concentration, the middle line represents the ten-day average daily peak ozone concentration and the lower line represents the lowest peak daily ozone concentration. The peak daily measured one-hour ozone concentrations for the ten highest ozone days ranged from 105 ppb to 135 ppb with an average of 117 ppb. The peak daily measured one-hour ozone concentration for the ten median ozone days was 70 ppb for each day. The peak daily measured one-hour ozone concentrations for the ten lowest ozone days ranged from 30 ppb to 40 ppb with an average of 38 ppb. The upper line (solid triangle markers) on Fig. 8 shows the modeled peak daily one-hour ozone concentration for each base case scenario for the Knoxville area, using the EPA recommended aloft ozone and precursor values described above (40 ppb, 22.2 ppb, 2 ppb). The lower line (open triangle markers) shows the modeled peak daily one-hour ozone concentration for each base case scenario for the Knoxville area, using zero ozone and precursor concentrations aloft. These two lines represents the upper and lower bound of ozone concentrations predicted within the range of aloft concentrations.

Figure 11 shows the selection of aloft values based on the best agreement with the monitored values. The bold line (labeled H-0-0) represents the ozone concentrations resulting from the aloft concentrations selected for each base case condition. Since the EPA recommended values (40 ppb, 22.2 ppb, 2 ppb) showed good agreement with the highest of the ten high ozone days, these values were selected to simulate a high ozone day. The modeled daily peak one-hour ozone concentration for high ozone conditions was 134 ppb compared with the measured high ozone ten-day average daily peak concentration of 117 ppb (during which time the daily peak concentration ranged from 105 to 135 ppb). The median ozone monitored concentration was modeled more accurately with the aloft ozone and precursor concentrations set to zero. The modeled daily peak one-hour ozone concentration under median ozone conditions was 67 ppb compared to the measured median ozone ten-day average daily peak concentration of 70 ppb. The low ozone modeled value showed the

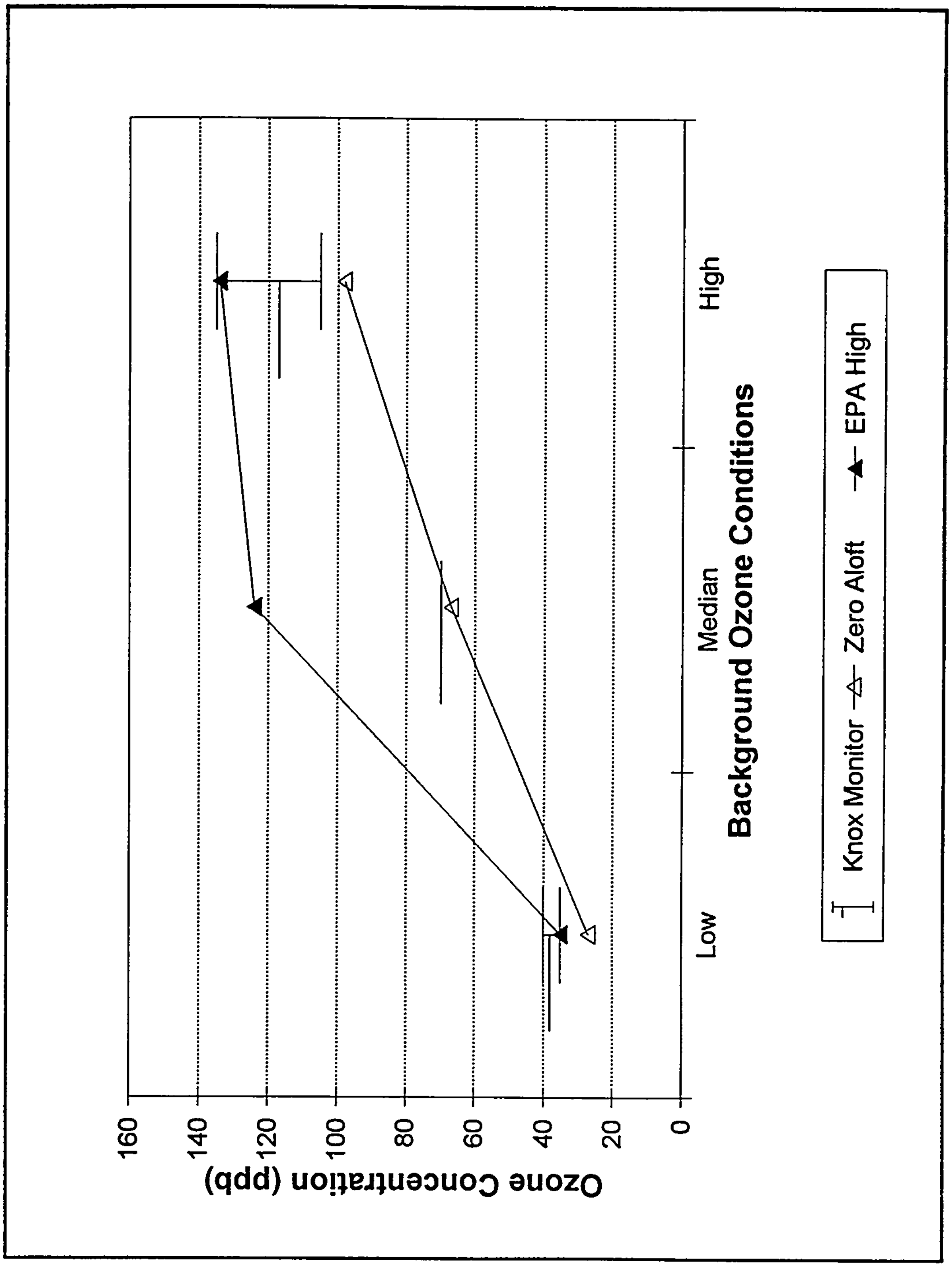

Fig. 10. Modeled and measured ozone concentrations for three base case conditions for the Knoxville calibration simulations

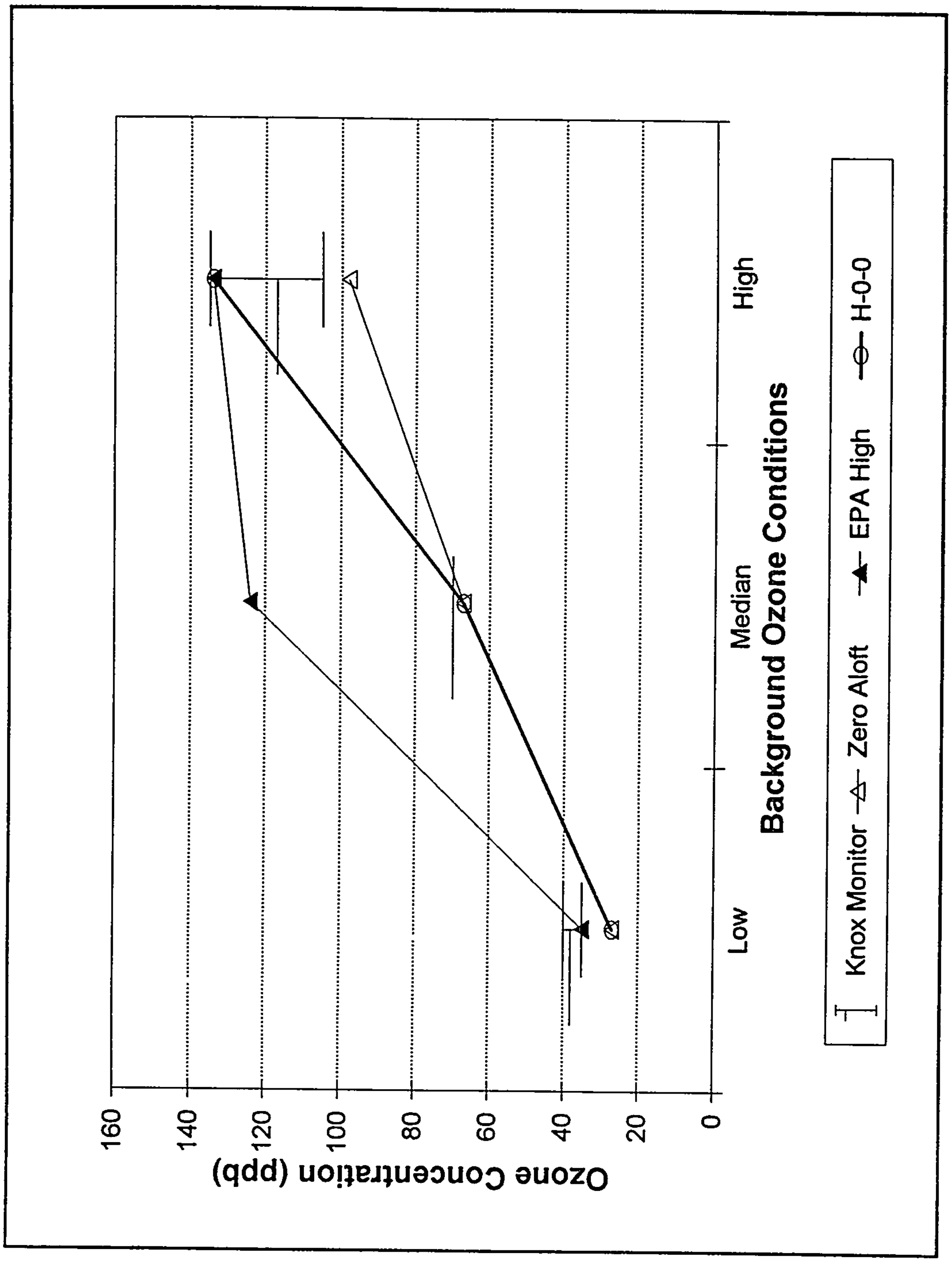

Fig. 11. Modeled and measured ozone concentrations showing selected aloft conditions for the Knoxville calibration simulations

least sensitivity to aloft ozone and precursor concentration values, and for this reason, was also set to zero. The modeled daily peak one-hour ozone concentration under low ozone conditions was 27 ppb compared to the measured low ozone ten-day average daily peak concentration of 38 ppb.

3.4.2 Southeast Reference Site Results

Using the aloft ozone and precursor concentrations described above, and anthropogenic and biogenic emissions for the Southeast Reference site, the OZIPM-4 was run for the three base case conditions at the Southeast Reference site. Peak daily one-hour ozone concentrations for each of the three base case conditions are shown in Fig. 12 for the Southeast Reference site and Knoxville. As seen in Fig. 12, the largest difference between sites is seen for the high ozone base case with a peak one-hour ozone concentration at Knoxville of 134 ppb and a peak one-hour ozone concentration at the Southeast Reference site of 88 ppb. The corresponding concentrations for the median base case scenario were 67 ppb at Knoxville and 56 ppb at the Southeast Reference site. For the low ozone base case, the peak one-hour ozone concentration at the Southeast Reference site was 45 ppb and the peak one-hour ozone concentration at Knoxville was 27 ppb. The higher ozone concentrations predicted at the rural Southeast Reference site for the low ozone base case may be due to the higher biogenic emissions estimated at the rural site. Whatever the reason, the concentrations are too low to be of much significance in this application.

3.5 STABILITY CLASS AND ATMOSPHERIC MIXING

Stability class (which is a measure of atmospheric mixing) was computed with the EPA PCRAMMET program using the meteorological surface observations file described above for each day of the three groups of ten days. The modal stability class for each hour was selected for each composite day. Stability class is not used as an input to any of the models used in this analysis, however it is an important factor in supporting the assumptions made in using the OZIPM-4 model to estimate impacts from large point sources. As seen in Tables 7 and 8, the most frequently occurring stability class during daytime hours under high and median ozone days is stability 2 (or B stability - moderately unstable). Under these conditions of atmospheric mixing, the plume from an elevated point source, such as a power plant stack, would be expected to mix to the ground within a relatively short downwind distances (i.e. relatively short travel times). An estimate of the distance to plume touchdown was made with the EPA SCREEN model. Using stability B and a wind speed of 2.1 meters/second, (24-hr average, 10-meter wind speed under median ozone conditions) plume touchdown occurred less than 1,600 meters from the stack (approximately 13 minutes travel time). Similar results were obtained for high

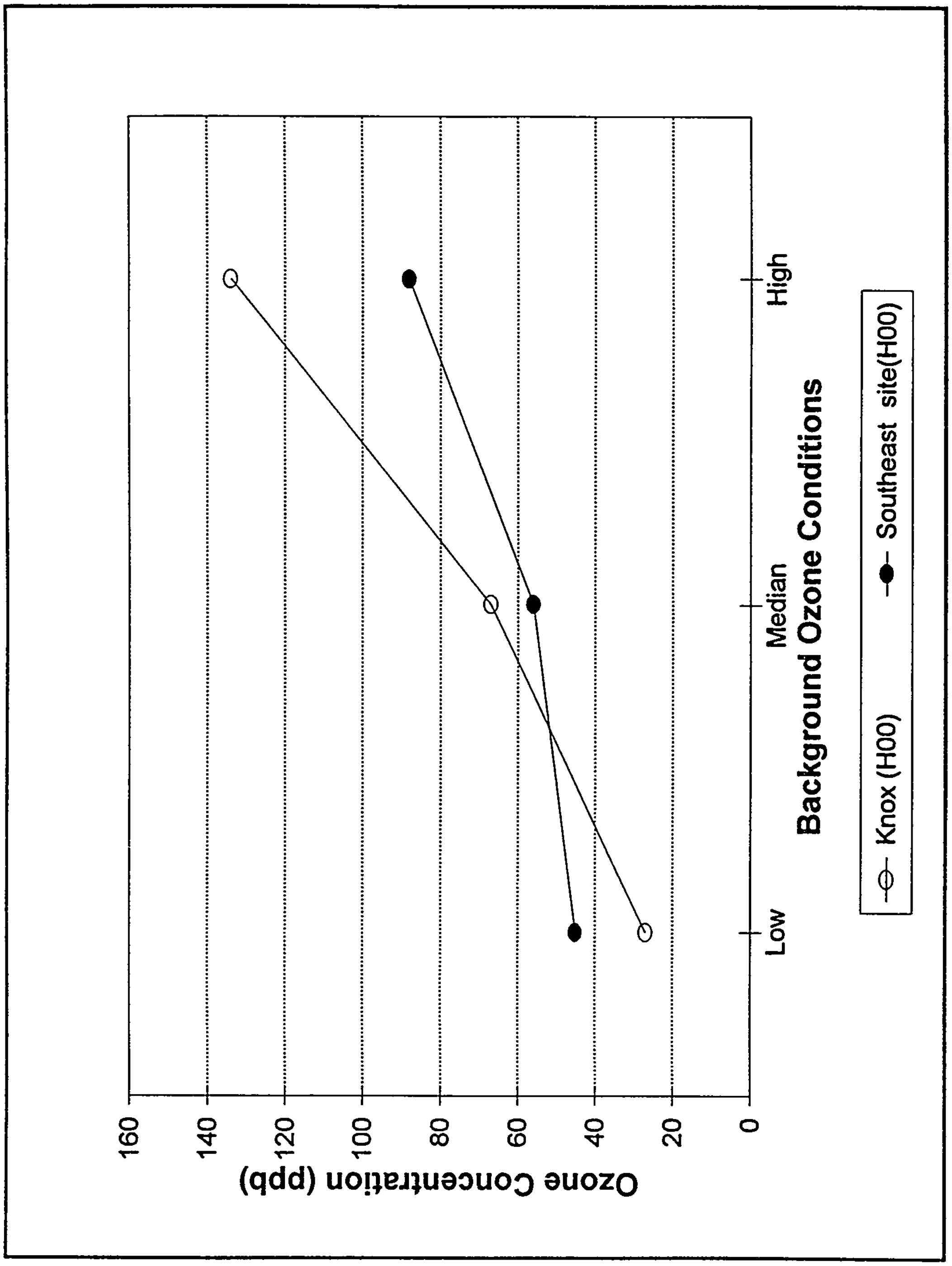

Fig. 12. Modeled ozone concentrations for Knoxville and the southeast site for three base case conditions using selected aloft conditions

ozone conditions of stability B and wind speed of 1.7 m/s (24-hr average, 10-meter wind speed under high ozone conditions) with the plume touching down within 1,800 meters (approximately 18 minutes travel time). Based on these SCREEN model results, it is apparent that the OZIPM-4 assumption of complete mixing is not inconsistent with plume mixing behavior during summer daytime conditions.

3.6 NO_x EMISSIONS FLUX

Once the base case simulations for the Southeast Reference site have been run, the power plant emissions are entered in the OZIPM-4 model in the form of an hourly emissions flux. Unlike Gaussian dispersion models which accept emissions from point sources as an emission rate (e.g. grams/second), the OZIPM-4 model accepts emissions of NO_x and NMOC as an emissions flux in units of kilograms per square kilometer per hour (kg/km^2-hr). Both the OZIPM-4 model and Gaussian type models predict pollutant concentrations, typically in units of grams per cubic meter, (g/m^3) or ppb. As discussed in Section 3.2, the simulated column of air in the OZIPM-4 model is assumed to extend from the earth's surface through the mixed layer and the air within the column is assumed to be uniformly mixed at all times. As the column of air passes over the power plant, the column is 'initialized' with a quantity of NO_x and NMOC emissions from the plant.

In the OZIPM-4 model, the column of air is transported at some wind speed (u) along a trajectory (Lagrangian coordinate system). Output from the model is in the form of pollutant concentrations that occur, within the column, after some period of time (travel time or downwind distance assuming some wind speed). In order to use the OZIPM-4 model to calculate ozone concentrations due to a point source, an emissions flux must be calculated and entered into the model, that will result in a concentration within the column (i.e. the plume) equal to that which would occur from the plant emissions after traveling downwind for one hour. The one hour time period is chosen because that is the normal temporal resolution achieved with the OZIPM-4 model. That is, OZIPM-4 is typically used to calculate (instantaneous or average) ozone concentrations, hour by hour. Therefore, all input conditions such as emissions are one-hour averages.

If the height of the column is the mixing height (L), the concentration of pollutant within the column of air in the OZIPM-4 model after one hour of travel time is given by:

Table 7. Hourly average wind speeds for the 10-high ozone days during 1990 at Rutledge Pike, Knoxville used to calculate emissions fluxes

Begin Hour	10-meter Wind Speed* (m/s)	Stack Top Wind Speed* (m/s)	Stability Class	Adjusted** NO_x Flux (kg/km^2-hr)	Adjusted** NMOC Flux (kg/km^2-hr)
0	1.0	4.6	7	2.6	0.04
1	1.1	5.0	7	2.9	0.04
2	1.2	5.5	7	3.2	0.05
3	0.5	2.1	7	3.6	0.05
4	0.9	3.9	7	4.1	0.06
5	0.8	3.4	7	4.8	0.07
6	0.6	2.5	6	5.7	0.09
7	1.0	2.5	5	7.2	0.11
8	1.3	2.0	4	9.6	0.15
9	2.2	2.9	3	14.3	0.22
10	2.4	2.9	2	28.7	0.44
11	2.2	2.6	2	28.7	0.44
12	2.8	3.4	1	28.7	0.44
13	2.8	3.4	1	28.7	0.44
14	2.3	2.7	1	28.7	0.44
15	2.8	3.4	2	28.7	0.44
16	2.3	2.8	2	28.7	0.44
17	2.3	3.0	3	28.7	0.44
18	2.1	2.6	2	28.7	0.44
19	2.5	3.3	3	28.7	0.44
20	2.1	3.2	4	28.7	0.44
21	2.1	5.4	5	28.7	0.44
22	1.4	6.2	6	28.7	0.44
23	1.2	5.5	6	28.7	0.44

*24-hr average of stack height and surface wind speed: 2.6
**Flux based on the 24-hr average stack height and surface wind speed; flux for hours 0 to 9 is adjusted for spreading of plume

Table 8. Hourly average wind speeds for the 10-median ozone days during 1990 at Rutledge Pike, Knoxville used to calculate emissions fluxes

Begin Hour	10-meter Wind Speed* (m/s)	Stack Top Wind Speed* (m/s)	Stability Class	Adjusted** NO_x Flux (kg/km^2-hr)	Adjusted** NMOC Flux (kg/km^2-hr)
0	1.3	5.7	6	1.5	0.02
1	1.2	5.5	7	1.6	0.02
2	1.3	5.7	6	1.8	0.03
3	1.0	4.3	6	2.0	0.03
4	1.6	7.1	7	2.3	0.04
5	1.4	6.2	6	2.7	0.04
6	1.3	3.5	5	3.2	0.05
7	1.3	1.9	4	4.0	0.06
8	1.4	2.1	4	5.4	0.08
9	1.9	2.5	3	8.0	0.12
10	1.8	2.2	2	16.1	0.25
11	2.8	3.7	3	16.1	0.25
12	3.6	4.3	2	16.1	0.25
13	3.1	3.7	2	16.1	0.25
14	3.1	3.7	2	16.1	0.25
15	2.8	3.4	2	16.1	0.25
16	2.5	3.2	3	16.1	0.25
17	3.0	4.0	3	16.1	0.25
18	3.1	4.7	4	16.1	0.25
19	2.4	3.6	4	16.1	0.25
20	2.4	6.2	5	16.1	0.25
21	2.4	10.5	6	16.1	0.25
22	1.9	8.5	6	16.1	0.25
23	2.2	9.8	6	16.1	0.25

*24-hr average of stack height and surface wind speed: 3.5

**Flux based on the 24-hr average stack height and surface wind speed; flux for hours 0 to 9 is adjusted for spreading of plume

$$C_c = \frac{F}{L} t_d (0.001) \qquad (1)$$

where,

C_c = the concentration in the column of air and has units of g/m^3,

F = the emissions flux which has units of kg/km^2-hr,

L = the mixing height which has units of meters and

t_d = the duration of emissions in hours (this value will always be one hour then the OZIPM-4 model is used to simulate a point source emission).

If we convert the source configuration to a polar system, in order to better account for the dispersion of pollutants from a point source, the concentration after one hour of travel time within a 22.5° pie shaped wedge (i.e. one of 16 sectors) with depth L is given by:

$$C_p = \frac{Q}{\frac{2}{16}\pi L u x} \qquad (2)$$

where,

C_p = the concentration which has units of g/m^3,

Q = the emission rate of pollutant from the plant in units of g/s,

u = the wind speed which has units of m/s,

L = the mixing height which has units of meters,

$(2/16)\pi$ = one sixteenth of the circumference of a circle and

x = the distance traveled in one hour, in meters.

The distance traveled in one hour, x, can be expressed as,

$$x = ut_t \left[\frac{3600\ \text{sec}}{1\ \text{hour}}\right] \tag{3}$$

where u is the wind speed in m/s and t_t is the travel time of the plume in hours. Then C_p is given by:

$$C_p = \frac{Q}{\frac{2}{16}\pi L u^2 t_t} \frac{1}{3600} \tag{4}$$

In order to determine the flux that will result in this concentration, we set C_p equal to C_c and solve for the flux, F:

$$F = \frac{8Q}{\pi u^2 t_t t_d}(0.2778) \tag{5}$$

where,

F has units of kg/km^2-hr,

Q has units of g/s,

u has units of m/s and

t_t and t_d have units of hours.

This is the emissions flux that will result in a NO_x concentration in the power plant plume, after one hour of travel time (i.e. one hour of dispersion) from the stack. This method of calculating flux is not appropriate for time periods less than one hour. This calculation assumes no chemical conversion during the first hour. During this time, NO_x concentrations from the plant are expected to be predominantly NO and very high (relative to ambient). Any chemical reactions occurring would most likely be the conversion of some NO to NO_2 by ambient ozone. After this time, NO_x concentrations in the column are expected to be dominated by photochemical reactions and vertical mixing of the atmosphere, as it is subsequently simulated by the OZIPM-4 model.

The emissions flux calculated with this method is a function of the pollutant emission rate (Q), duration of the emission, (t_d), travel time of the plume, (t_t) and the wind speed, (u). The NO_x emission rate for the coal-fired power plant at the Southeast Reference site is 274 g/s and the total hydrocarbon emission rate is 6 g/s. Non-methane hydrocarbons are assumed to be 70% of total hydrocarbons (EPA 1985), so 4.2 g/s NMOC was used to calculate NMOC emissions fluxes. Duration of the emission (t_d) is always one hour for the OZIPM-4 simulations in the MAP-O_3 model, since the column of air receives emissions, in units of kg/km^2-hr, from the stack as it is transported over the power plant plume.

The travel time of the plume (t_t) is the number of hours that the plume travels before mixing to the ground. Prior to 10 a.m., under typical summertime conditions, the mixing height (which may be thought of as a lid which prevents further vertical mixing) is still below the effective stack height. (The effective stack height is the combined height of the stack and the height that the plume has risen due to effects of momentum and buoyancy.) Until the mixing height exceeds the effective stack height, the plume is essentially trapped above the mixed layer and may be transported some distance before the mixing height rises sufficiently to allow the plume to be mixed to the ground. Due to the effects of the mixing height on plume mixing, it is assumed that no plume is mixed to the ground prior to 10 a.m. Any plume which originates between 10 a.m. and 8 p.m. is assumed to mix to the ground within an hour of travel time. Plumes which originate prior to this time are assumed to be transported aloft until 10 a.m., after which time solar heating is sufficient to produce vertical mixing. Since sunlight and temperature are not sufficient to promote photochemical activity during early morning hours, the most likely effect from early morning emissions is to increase concentrations of NO_x aloft until such time, as they are mixed to the ground and can react with NMOC emissions.

The flux calculation for hours prior to 10 a.m. is adjusted to account for the fact that the plume has undergone additional dispersion prior to mixing to the ground. To account for the additional dispersion which occurs in plumes which originate prior to 10 a.m., the flux for each of these hours is defined as a function of the 10 a.m. flux. Plumes which have traveled two hours (dispersed two hours) are assumed to have half the flux of a plume which has traveled one hour ($F_{9a.m.} = F_{10a.m.} / 2$) and plumes which have traveled three hours are assumed to have one third the flux of a plume which has traveled one hour ($F_{8a.m.} = F_{10a.m.} / 3$) and so on. The flux for hours prior to 10 a.m. is calculated with Equation 5 with t_t = travel time of the plume prior to mixing to the ground (i.e. the number of hours prior to 10 a.m. plus one hour to account for 10 - 11 a.m.).

Tables 7, 8 and 9 show the wind speed data used in calculating the NO_x and NMOC emissions flux for the 500 MW coal-fired power plant under low, median and high ozone conditions. The 10-meter wind speeds are the 10-day average observations described earlier for each composite day. Since wind speed varies with height (wind speeds at the earth's surface are slower due to frictional effects of surface roughness), the stack top wind speed was calculated from the 10-meter wind speed using the stability class and the power law expression (Wark and Warner 1981):

$$\frac{u}{u_1} = \left[\frac{z}{z_1}\right]^p$$

where, u is the wind speed at altitude z,

u_1 is the wind speed at altitude z_1 and

p is the positive exponent which is a function of stability class.

Default rural wind profile exponents from the Industrial Source Complex (ISC) Dispersion Model User's Guide were used (EPA 1986). The stack height of the coal-fired power plant at the Southeast Reference site is 150 meters.

In calculating the emissions flux, a 24-hour average representative wind speed was developed for each composite base case scenario. The combined 24-hour average of both the 10-meter and the stack top wind speeds was computed for the flux calculation. This average wind speed was selected to dampen some of the hourly variability seen in both wind speeds and to account for the fact that the actual wind speed is, in fact, unknown and may actually be higher than the surface wind speed and lower than the calculated stack top wind speed. The average wind speeds for the high, median and low ozone conditions were 2.6, 3.5 and 4.2 m/s, respectively.

The 24-hour, average wind speeds described here were used to calculate the emissions flux for the plant under high, median and low ozone conditions during the hours from midnight to 9 p.m. Due to the uncertainty regarding the location of the mixing height, with respect to the plume, during the evening hours (9 p.m. to midnight) and to the fact that emissions from the plant during this time are not expected to have an appreciable impact on ozone concentrations during the following day, ozone concentrations were not predicted for plumes which originate between 9 p.m. and 11 p.m.

Table 9. Hourly average wind speeds for the 10-low ozone days during 1990 at Rutledge Pike, Knoxville used to calculate emissions fluxes

Begin Hour	10-meter Wind Speed* (m/s)	Stack Top Wind Speed* (m/s)	Stability Class	Adjusted** NO_x Flux (kg/km^2-hr)	Adjusted** NMOC Flux (kg/km^2-hr)
0	2.8	7.2	5	1.0	0.02
1	2.6	11.6	6	1.1	0.02
2	2.3	5.8	5	1.2	0.02
3	2.1	9.4	6	1.4	0.02
4	2.0	5.0	5	1.6	0.02
5	2.0	8.9	6	1.8	0.03
6	1.8	2.7	4	2.2	0.03
7	2.3	3.5	4	2.8	0.04
8	2.4	3.6	4	3.7	0.06
9	3.1	4.6	4	5.5	0.08
10	3.5	5.2	4	11.1	0.17
11	3.5	5.2	4	11.1	11.10
12	3.5	5.2	4	11.1	0.17
13	3.7	5.6	4	11.1	0.17
14	3.9	5.9	4	11.1	0.17
15	4.1	6.1	4	11.1	0.17
16	3.8	5.6	4	11.1	0.17
17	3.4	5.1	4	11.1	0.17
18	3.4	5.1	4	11.1	0.17
19	2.7	4.1	4	11.1	0.17
20	2.3	3.4	4	11.1	0.17
21	2.4	3.6	4	11.1	0.17
22	3.1	4.7	4	11.1	0.17
23	2.1	5.3	5	11.1	0.17

*24-hr average of stack height and surface wind speed: 3.5
**Flux based on the 24-hr average stack height and surface wind speed; flux for hours 0 to 9 is adjusted for spreading of plume

The NO_x and NMOC emissions flux for each plume birth hour are shown in Tables 7, 8 and 9. These values were input to the OZIPM-4 model in order to predict the ozone concentrations expected to occur as the result of power plant plumes that originate at certain hours and travel for some period of time.

3.7 BIRTH HOUR AND PLUME AGE RESULTS

As previously discussed, in a Lagrangian coordinate system, the plume is defined by both the time that it originated from the stack (birth hour) and its travel time or age (plume age). Ground-level ozone concentrations were predicted with the OZIPM-4 model for plumes that originated between the hours of midnight and 8 p.m. (begin hour 0:00 to 20:00). Each plume was followed until 9 p.m. For example, in a plume which was born at midnight, ozone concentrations were predicted for each hour, for a total travel time of 21 hours. In a plume which was born at 1 a.m., ozone concentrations were predicted for each hour for a total travel time of 20 hours and so on, until ozone concentrations were predicted for only one hour of travel time, in a plume which was born at 8 p.m.

Results for these OZIPM-4 simulations are shown in Tables 10, 11 and 12, and Figs. 13 through 18. Since the emissions flux for plumes born prior to 10 a.m. is zero until the plume mixes to the ground, ground-level ozone concentrations are also zero for these plumes prior to 10 a.m.

3.7.1 High Ozone Plant Case

Figures 13 and 14 show results for the OZIPM-4 model power plant simulations under high ozone conditions. Figure 13 shows results for plumes which originate between the hours of midnight and 9 a.m. Ozone depletion is seen in all these plumes during the first hour(s) of travel time. The plume which originated at 9 a.m. showed the greatest depletion (-8.5 ppb). Concentrations remained below ambient levels for three hours; after which time the ozone concentration increased to a maximum of 10 ppb. The plume which originated between midnight and 1 a.m. was transported aloft for ten hours, during which time it underwent additional horizontal dispersion and then mixed to the ground between 10 a.m. and 11 a.m. The ozone concentration during the initial hour (after mixing to the ground) was -1.1 ppb and increased to 2.6 ppb several hours later. These results show that plumes which originate during the early morning hours do not cause either large increases in ozone concentrations or large depletions of ozone. The range of ozone concentrations for plumes emitted during this time period ranged from -8.5 ppb to 10 ppb.

Hourly Ozone Concentration (ppb)																					
	Plume Age																				
Birth Hour	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21
0:00 - 1:00	0	0	0	0	0	0	0	0	0	0	-1.1	0.47	2.1	2.6	2.6	2.7	2.7	2.6	2.6	2.6	2.6
1:00 - 2:00	0	0	0	0	0	0	0	0	0	-1.3	0.4	2.2	2.8	2.9	2.9	2.9	2.9	2.8	2.8	2.8	
2:00 - 3:00	0	0	0	0	0	0	0	0	-1.50	.32	2.4	3.0	3.1	3.2	3.2	3.1	3.1	3.1	3.1		
3:00 - 4:00	0	0	0	0	0	0	0	-1.7	0.16	2.5	3.3	3.5	3.6	3.6	3.5	3.5	3.4	3.4			
4:00 - 5:00	0	0	0	0	0	0	-2.0	-.08	2.6	3.6	3.8	3.9	3.9	3.9	3.8	3.8	3.8				
5:00 - 6:00	0	0	0	0	0	-2.4	-.50	2.8	4.0	4.3	4.5	4.5	4.4	4.4	4.4	4.4					
6:00 - 7:00	0	0	0	0	-2.9	-1.1	2.9	4.5	4.9	5.2	5.2	5.1	5.1	5.1	5.1						
7:00 - 8:00	0	0	0	-3.7	-2.4	2.8	5.1	5.8	6.2	6.2	6.2	6.2	6.1	6.1							
8:00 - 9:00	0	0	-5.0	-4.6	2.0	5.7	6.9	7.5	7.7	7.7	7.7	7.6	7.6								
9:00 - 10:00	0	-7.2	-8.5	-1.73	5.5	8.2	9.5	10.0	10.1	10.2	10.1	10.1									
10:00 - 11:00	-12.6	-16.7	-15.5	-7.2	4.2	10.6	13.5	14.6	15.1	15.2	15.2										
11:00 - 12:00	-8.7	-7.7	0.68	10.2	15.0	17.1	17.8	18.1	18.2	18.1											

Table 10. Incremental ozone concentrations (ppb) due to the Southeast Reference site coal-fired power plant NO_x emissions under high ozone conditions for various birth hours and plume ages (OZIPM-4 Model)

Birth Hour	Hourly Ozone Concentration (ppb)																				
	Plume Age																				
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21
12:00 - 13:00	-2.1	5.1	14.6	19.8	21.9	22.5	22.7	22.7	22.6												
13:00 - 14:00	1.8	11.9	20.3	24.1	25.4	25.9	25.9	25.9													
14:00 - 15:00	3.0	13.6	21.4	24.6	25.9	26.1	26.0														
15:00 -1 6:00	2.8	11.9	17.9	20.5	21.2	21.1															
16:00 - 17:00	1.1	6.6	9.6	10.5	10.3																
17:00 - 18:00	-1.5	0.69	1.2	0.93																	
18:00 - 19:00	-4.2	-3.9	-4.3																		
19:00 - 20:00	-6.2	-6.6																			
20:00 - 21:00	-7.4																				

Table 10. (cont.) Incremental ozone concentrations (ppb) due to the Southeast Reference site coal-fired power plant NO_x emissions under high ozone conditions for various birth hours and plume ages (OZIPM-4 Model)

	Hourly Ozone Concentration (ppb)																				
	Plume Age																				
Birth Hour	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21
0:00 - 1:00	0	0	0	0	0	0	0	0	0	0	-.64	-1.3	-0.41	1.2	1.9	2.2	2.3	2.3	2.3	2.2	2.2
1:00 - 2:00	0	0	0	0	0	0	0	0	0	-.68	-1.4	-0.49	1.2	2.0	2.4	2.5	2.5	2.5	2.4	2.4	
2:00 - 3:00	0	0	0	0	0	0	0	0	-0.77	-1.6	-0.70	1.2	2.1	2.6	2.7	2.7	2.7	2.7	2.7		
3:00 - 4:00	0	0	0	0	0	0	0	-0.84	-1.8	-.89	1.2	2.3	2.8	3.0	3.0	3.0	2.9	2.9			
4:00 - 5:00	0	0	0	0	0	0	-0.97	-2.1	-1.3	1.2	2.5	3.1	3.3	3.3	3.3	3.3	3.3				
5:00 - 6:00	0	0	0	0	0	-1.1	-2.5	-1.8	1.0	2.7	3.4	3.7	3.8	3.7	3.7	3.7					
6:00 - 7:00	0	0	0	0	-1.3	-3.0	-2.4	0.74	2.9	3.9	4.2	4.3	4.3	4.3	4.3						
7:00 - 8:00	0	0	0	-1.6	-3.7	-3.6	-0.05	2.9	4.3	4.8	5.0	5.0	4.9	4.9							
8:00 - 9:00	0	0	-2.0	-4.8	-5.5	-2.0	2.3	4.6	5.6	5.8	5.9	5.9	5.9								
9:00 - 10:00	0	-2.6	-6.2	-8.4	-6.3	-0.74	3.5	5.6	6.4	6.7	6,7	6.7									
10:00 - 11:00	-3.8	-9.0	-13.6	-15.3	-14.1	-11.7	-8.3	-5.9	-4.7	-4.4	-4.5										
11:00 - 12:00	-5.6	-10.7	-12.6	-11.4	-8.5	-4.5	-1.8	-0.60	-0.35	-0.41											

Table 11. Incremental ozone concentrations (ppb) due to the Southeast Reference site coal-fired power plant NO_x emissions under median ozone conditions for various birth hours and plume ages (OZIPM-4 Model)

	Hourly Ozone Concentration (ppb)																				
	Plume Age																				
Birth Hour	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21
12:00 - 13:00	-4.9	-7.0	-5.5	-1.7	2.7	5.2	6.2	6.5	6.4												
13:00 - 14:00	-1.9	-0.61	3.3	7.7	10.0	10.9	11.1	11.0													
14:00 - 15:00	0.01	2.9	6.8	9.3	10.4	10.6	10.6														
15:00 -1 6:00	0.30	2.9	4.8	5.7	5.8	5.6															
16:00 - 17:00	-0.61	0.31	0.63	0.49	0.19																
17:00 - 18:00	-2.4	-2.4	-2.8	-3.2																	
18:00 - 19:00	-3.7	-4.2	-4.6																		
19:00 - 20:00	-4.7	-5.2																			
20:00 - 21:00	-5.41																				

Table 11. (cont.) Incremental ozone concentrations (ppb) due to the Southeast Reference site coal-fired power plant NO_x emissions under median ozone conditions for various birth hours and plume ages (OZIPM-4 Model)

Birth Hour	Hourly Ozone Concentration (ppb) — Plume Age 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21
0:00 - 1:00		0	0	0	0	0	0	0	0	0	-0.19	-0.55	-.80	-0.77	-0.45	-0.13	0.01	-0.01	-0.04	-0.05	-.06
1:00 - 2:00	0	0	0	0	0	0	0	0	0	-0.21	-0.62	-.90	-.88	-0.55	-0.21	-0.05	-0.07	-0.11	-0.12	-0.12	
2:00 - 3:00	0	0	0	0	0	0	0	0	-.23	-0.68	-1.0	-1.0	-.66	-0.20	-0.11	-0.14	-0.17	-0.18	-0.19		
3:00 - 4:00	0	0	0	0	0	0	0	-0.27	-0.79	-1.2	-1.2	-.88	-.39	-0.27	-0.29	-.34	-0.35	-0.36			
4:00 - 5:00	0	0	0	0	0	0	-0.31	-.92	-1.4	-1.5	-1.1	-0.70	-.48	-0.51	-0.56	-0.58	-.60				
5:00 - 6:00	0	0	0	0	0	-0.35	-1.0	-1.6	-1.8	-1.4	-0.88	-0.71	-0.74	-0.80	-0.83	-.84					
6:00 - 7:00	0	0	0	0	-.43	-1.3	-2.0	-2.4	-2.0	-1.6	-1.3	-1.3	-1.4	-1.4	-1.4						
7:00 - 8:00	0	0	0	-0.53	-1.6	-2.7	-3.3	-3.1	-2.6	-2.4	-2.5	-2.6	-2.7	-2.7							
8:00 - 9:00	0	0	-.68	-2.1	-3.6	-4.6	-4.8	-4.6	-4.5	-4.7	-4.9	-5.0	-5.0								
9:00 - 10:00	0	-0.96	-3.0	-5.2	-7.1	-8.2	-8.9	-9.3	-9.9	-10.3	-10.5	-10.6									
10:00 - 11:00	-1.5	-4.8	-8.4	-12.1	-15.3	-18.0	-20.0	-21.6	-23.0	-23.5	-23.6										
11:00 - 12:00	-2.7	-6.5	-10.4	-13.6	-16.3	-18.3	-19.9	-21.2	-21.6	-21.7											

Table 12. Incremental ozone concentrations (ppb) due to the Southeast Reference site coal-fired power plant NO_x emissions under low ozone conditions for various birth hours and plume ages (OZIPM-4 Model)

	Hourly Ozone Concentration (ppb)																				
	Plume Age																				
Birth Hour	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21
12:00 - 13:00	-3.4	-7.3	-10.5	-13.0	-14.9	-16.4	-17.5	-17.9	-18.1												
13:00 - 14:00	-3.6	-6.8	-9.2	-11.0	-12.4	-13.4	-13.8	-14.0													
14:00 - 15:00	-3.2	-5.5	-7.2	-8.6	-9.5	-9.9	-10.1														
15:00 -1 6:00	-3.1	-4.8	-6.2	-7.1	-7.4	-7.7															
16:00 - 17:00	-3.2	-4.6	-5.5	-5.9	-6.2																
17:00 - 18:00	-3.7	-4.6	-5.1	-5.4																	
18:00 - 19:00	-4.0	-4.4	-4.7																		
19:00 - 20:00	-4.2	-4.6																			
20:00 - 21:00	-4.2																				

Table 12. (cont.) Incremental ozone concentrations (ppb) due to the Southeast Reference site coal-fired power plant NO_x emissions under low ozone conditions for various birth hours and plume ages (OZIPM-4 Model)

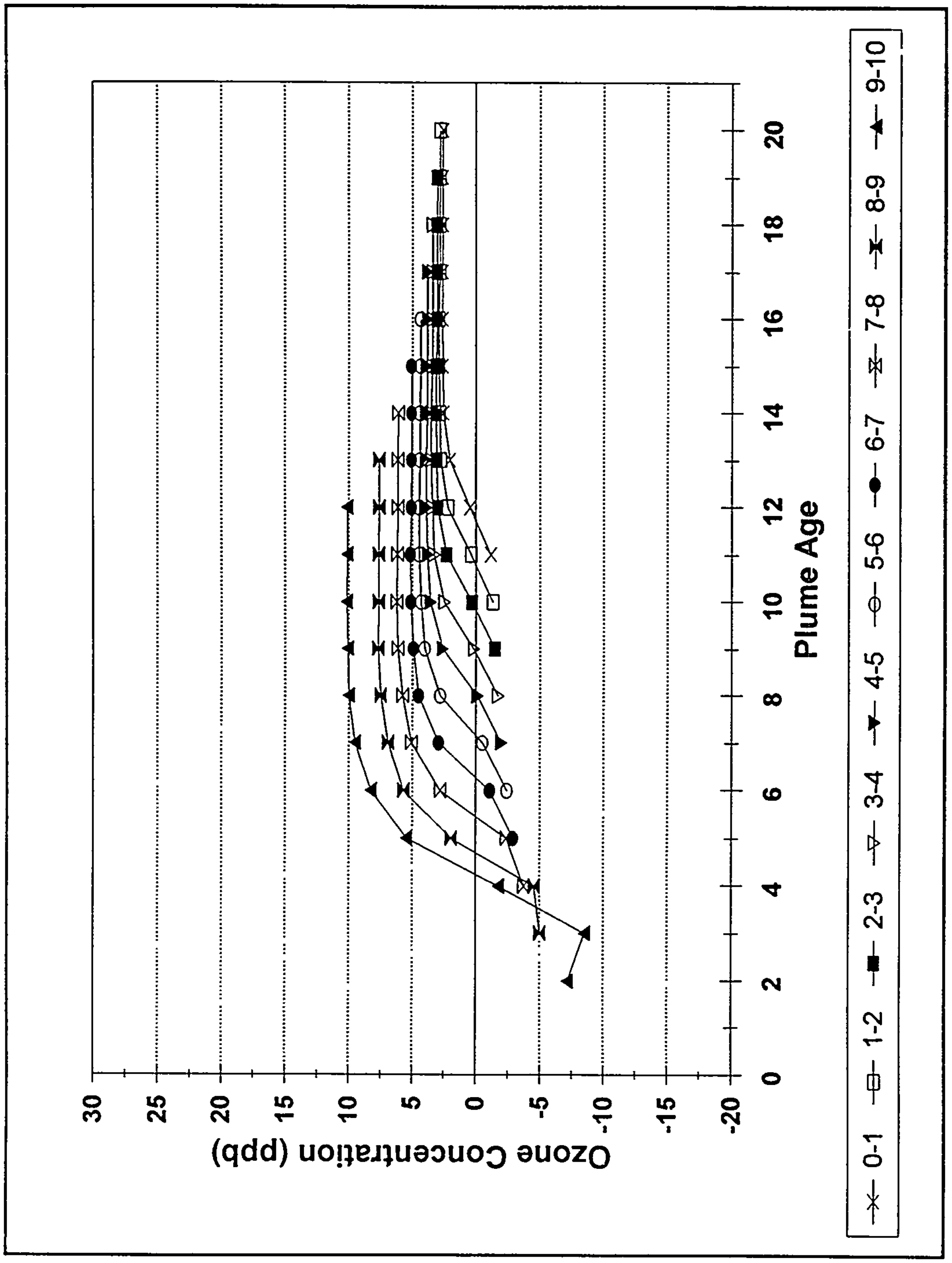

Fig. 13. Ozone concentration versus plume age for plume birth hours 0 to 9 under high ozone conditions

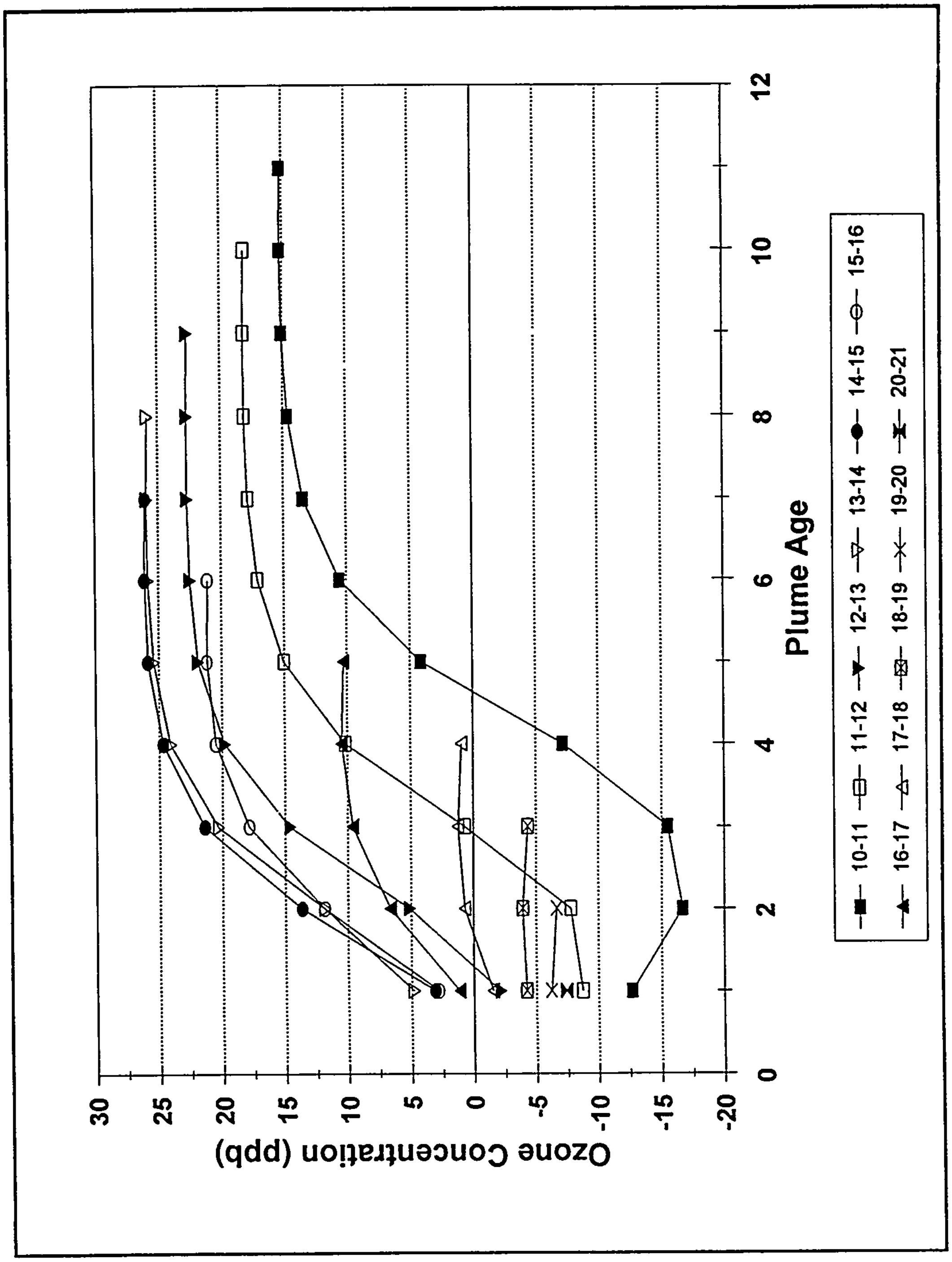

Fig. 14. Ozone concentration versus plume age for plume birth hours 10 to 20 under high ozone conditions

Figure 14 shows results for plumes which originate between the hours of 10 a.m. and 8 p.m. The largest ozone depletion (-17 ppb) is seen after two hours in a plume which originates between 10 and 11 a.m. The largest ozone formation (26 ppb) is seen after five to six hours in plumes which originates between 1 and 3 p.m. Plumes which are emitted between 5 p.m. and 8 p.m. are predominantly depletion plumes.

In general, plumes which are emitted during the hours between 9 a.m. to 11 a.m. show the greatest depletion for the longest period of time. These plumes contain high concentrations of NO_x and are mixed to the ground after minimal dispersion, during the morning hours when sunlight and temperature have not reached sufficient levels to promote the rapid formation of ozone. Plumes emitted during the period from 12 noon to 4 p.m. show relatively rapid increases in ozone. These plumes also contain high concentrations of NO_x, which is quickly mixed to the ground. However, on high ozone days, sunlight and temperature during this time period are sufficient to promote rapid formation of ozone.

3.7.2 Median Ozone Plant Case

Figures 15 and 16 show results for the OZIPM-4 model power plant simulations under median ozone conditions. The trends under median ozone conditions are very similar to the results for high ozone conditions, with less extreme ozone formation and depletion seen under median ozone conditions. Figure 15 shows results for plumes which originate between the hours of midnight and 9 a.m. Ozone depletion is seen in all these plumes during the first hour(s) of travel time. Ozone concentrations for plumes emitted during this time period ranged from -8.4 ppb to 6.7 ppb.

Figure 16 shows results for plumes which originate between the hours of 10 a.m. and 8 p.m. The largest ozone depletion (-15 ppb) is seen after four hours in a plume which originates between 10 and 11 a.m. The largest ozone formation (11 ppb) is seen after six hours in plumes which originate between 1 and 3 p.m. Plumes which are emitted between 4 p.m. and 8 p.m. are predominantly depletion plumes.

3.7.3 Low Ozone Plant Case

Figures 17 and 18 show that the net effect of NO_x and NMOC emissions from the coal-fired power plant during low ozone conditions is the depletion of ambient ozone. All ozone concentrations predicted for plumes born between midnight and 8 p.m. are less than the corresponding predictions for the low ozone base case. The largest depletion (-24 ppb) is seen in a plume which originated at 10 a.m. and traveled for 11 hours. The greatest ozone depletion

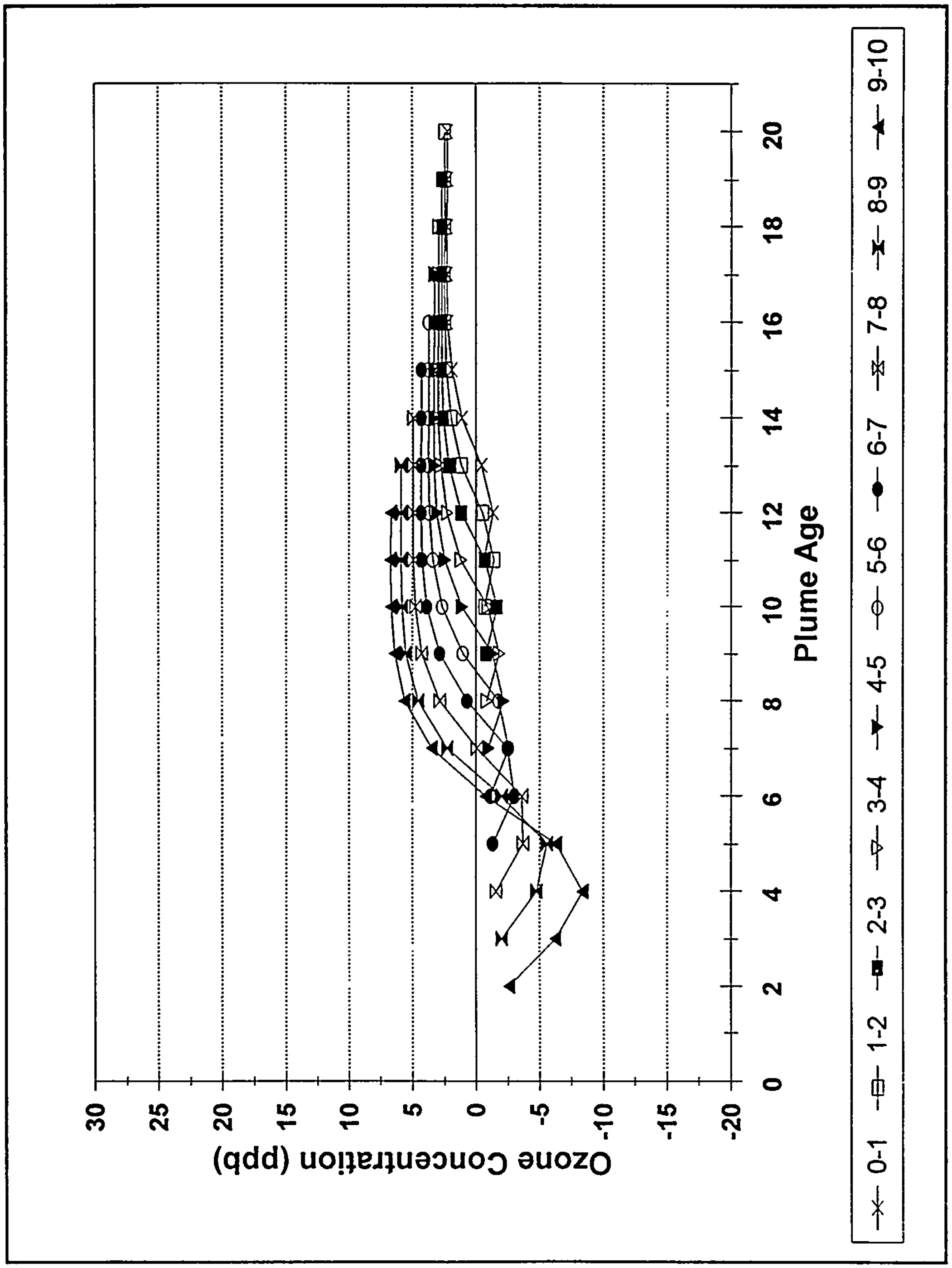

Fig. 15. Ozone concentration versus plume age for plume birth hours 0 to 9 under median ozone conditions

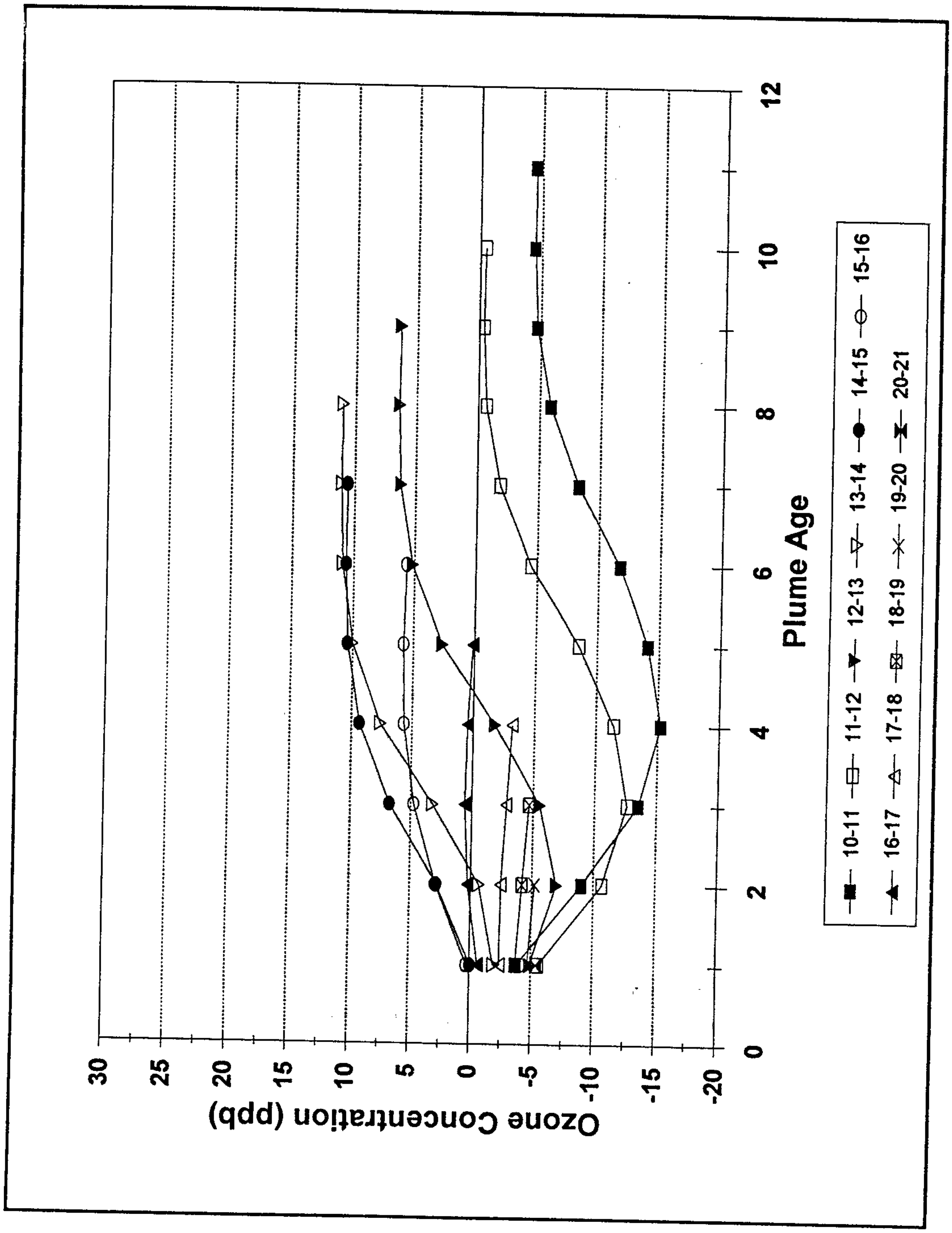

Fig. 16. Ozone concentration versus plume age for plume birth hours 10 to 20 under median ozone conditions

is not seen near the plant on low ozone days, as was predicted in some plumes under median and high ozone conditions and is often reported from plume measurement studies. (Most plume ozone measurement studies are conducted during moderate to high ozone conditions when ambient ozone concentrations are sufficiently high to show significant depletion near the plant, where NO concentrations are high.) In the low ozone base case, the peak daily ozone concentration without plant emissions is 45 ppb and morning ozone concentrations are much less. Not until later in the afternoon, when ambient ozone concentrations have increased, will ozone depletion from the power plant NO_x emissions, be evident under low ozone conditions.

As mentioned earlier, the peak daily one-hour average temperature during the composite low ozone day is 67° F (19.4° C). Apparently, the combination of meteorological inputs comprising the low ozone day are not sufficient to promote the formation of ozone, even several hours downwind of the plant. This is consistent with many plume measurement studies (Keifer 1977; Gillani and Wilson 1980; Davis 1974; Meagher et al. 1981; Luria et al. 1983) which report the formation of ozone concentrations above ambient, within the plumes of power plants only after several hours of travel time on sunny, summer afternoons when temperatures are high.

3.7.4 Discussion

These results from the OZIPM-4 model under low, median and high ozone conditions are generally consistent with plume measurement studies reported by Keifer (1977) for seven power plants in Virginia, West Virginia and Maryland. Keifer divided the plumes he studied into two groups: ozone depletion plumes and ozone formation plumes. Ozone depletion occurred at short downwind distances in both types, but at greater downwind distances, increases in ozone concentrations above ambient levels were seen in the ozone formation plumes. Keifer (1977) reported that on warm, sunny days, at downwind distances beyond the region of ozone depletion, ozone concentrations up to 40 ppb above ambient were observed in seven power plants in Maryland, Virginia and West Virginia. He concluded that the magnitude of the ozone "bulge" depended on the solar flux, temperature and ambient reactive hydrocarbons.

These results from the OZIPM-4 model under low, median and high ozone conditions are generally consistent with plume measurement studies reported by Keifer (1977)...

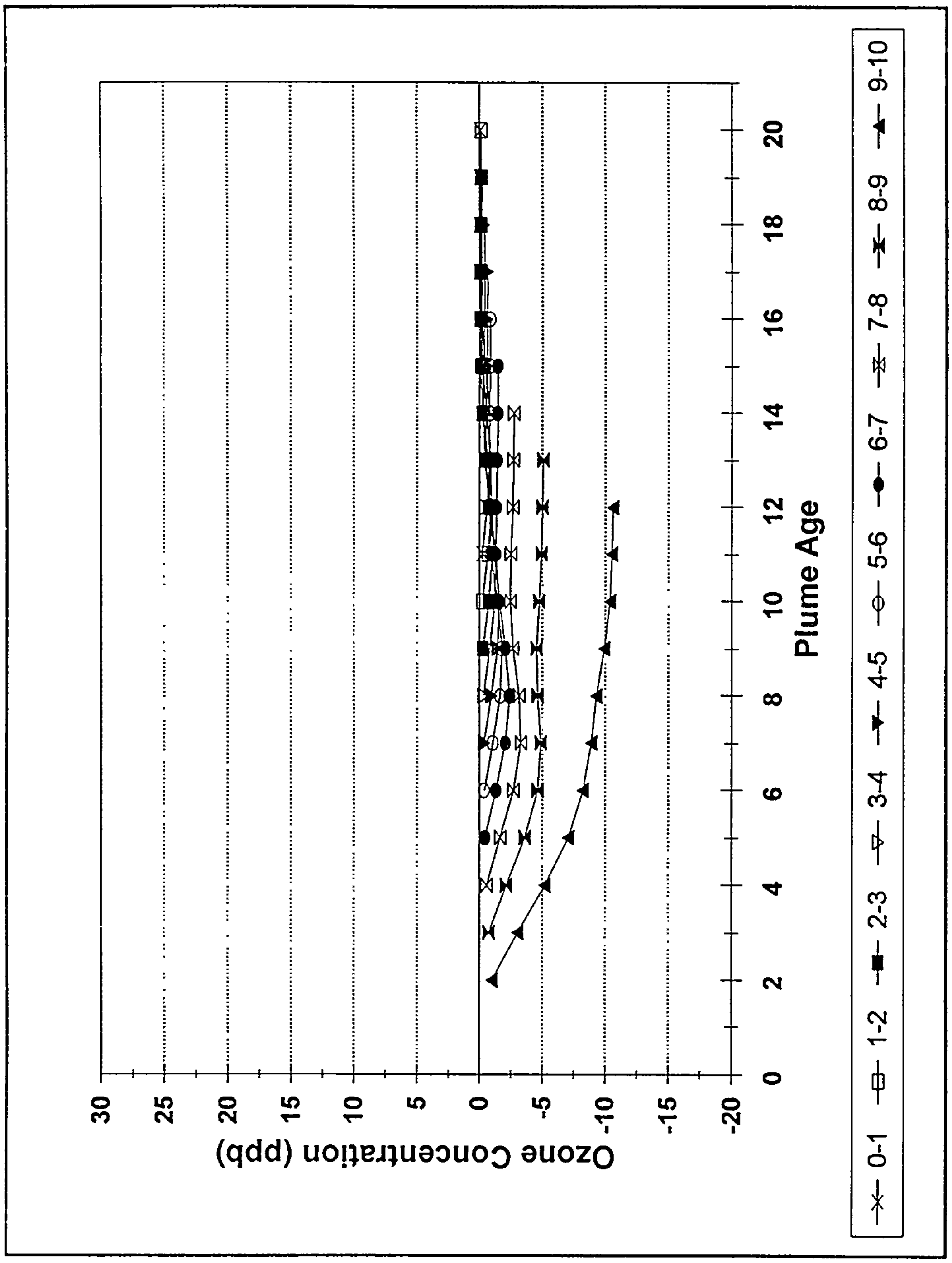

Fig. 17. Ozone concentration versus plume age for plume birth hours 0 to 9 under low ozone conditions

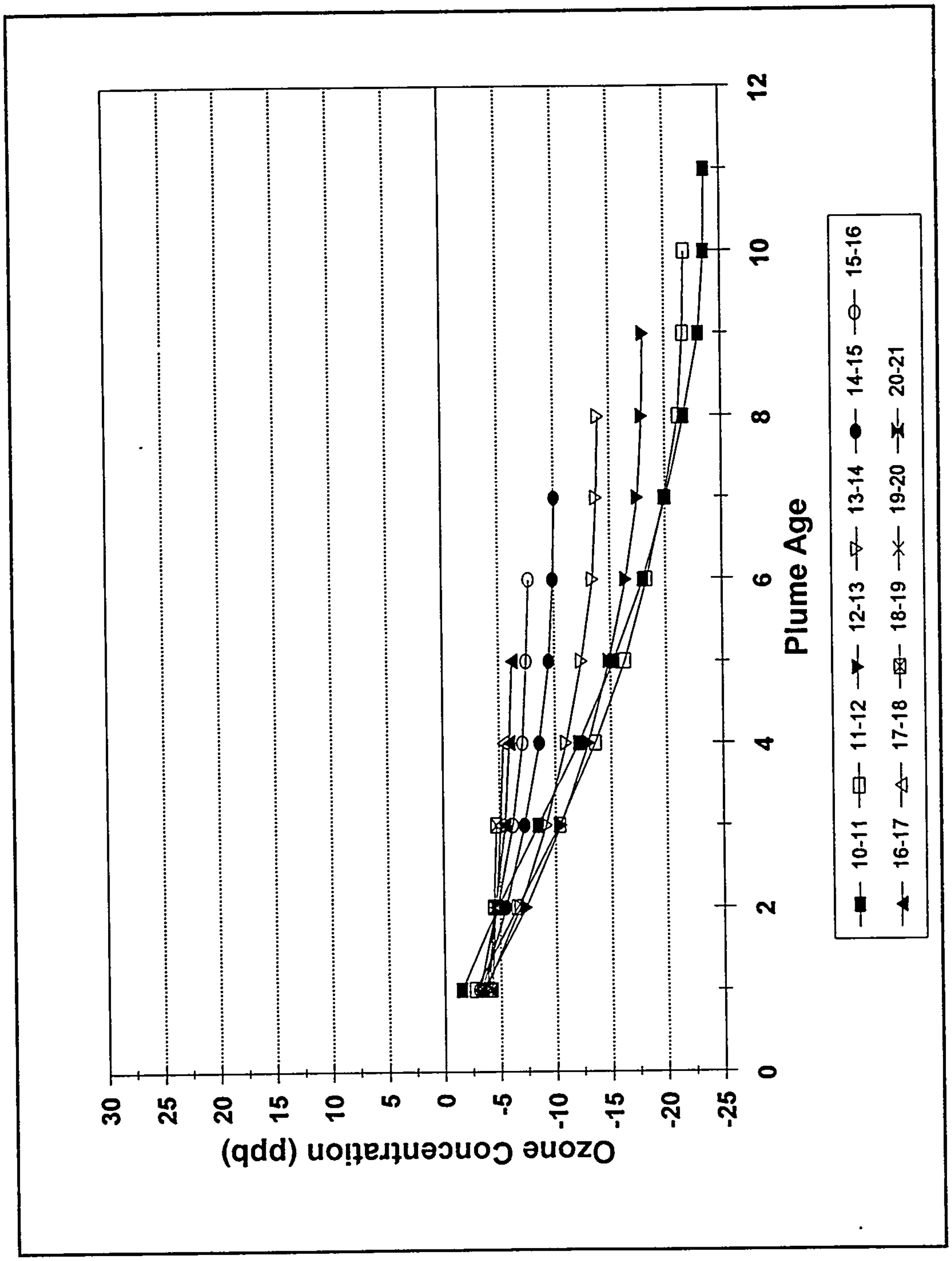

Fig. 18. Ozone concentration versus plume age for plume birth hours 10 to 20 under low ozone conditions

Refer to Table 13 for a comparison of Keifer's observations with the case study results from the OZIPM-4 model on low, median and high ozone days. Generally, Keifer reported that ozone formation plumes were not observed when surface temperatures were below 21° C. On low ozone days when the daily peak temperature was 19° C, results from the OZIPM-4 model simulations showed no ozone formation in any plume. Keifer noted that ozone bulges were observed only at great downwind distances (50 to 80 km or 8 to 10 hours travel time) when temperatures were between 21 and 26° C. On median ozone days when the daily peak temperature was 28° C, results from the OZIPM-4 model simulations showed ozone formation as high as 11 ppb at distances from 75 to 100 km from the plant (or 6 to 8 hours travel time). When temperatures were above 26° C, sizable ozone bulges on the periphery of the plume were observed within a few kilometers downwind (Keifer 1977). And on high ozone days when the daily peak temperature was 34° C, results from the OZIPM-4 model simulations showed a plume average ozone concentrations high as 5 ppb within 9 km (one hour) of the plant up to 26 ppb within 45 to 72 km (5 to 8 hours) of the plant.

Table 13. Comparison of the OZIPM-4 model case study results with Keifer's (1977) summary of plume measurement studies

Keifer (1977)[1]		**OZIPM-4 Model Case Study Results**	
Daily Peak Temperature	Observation	Daily Peak Temperature	Prediction
<21° C	No O_3 formation	19° C (low ozone day)	No O_3 formation
21 to 26° C	Formation at large distances	28° C (median ozone day)	11 ppb at 75 km to 100 km, (6 to 8 hours) (plume average concentration)
>26° C	Sizable bulges on periphery of plume within a few kilometers	34° C (high ozone day)	5 ppb within 9 km (1 hr); up to 26 ppb within 45 km - 72 km (5 to 8 hours) (plume average concentration)

(1) Based on measurement studies of seven power plants in Maryland, Virginia, and West Virginia

3.8 MAP-O_3 MODEL

Predicted ozone concentrations from the OZIPM-4 model correspond to travel time (in hours) from the start of a simulation or to a travel distance (assuming some wind speed) from the source. The MAP-O_3 model is used to spatially distribute the ozone concentrations predicted by the OZIPM-4 model, throughout a user-defined study area over some time period of interest. The objective is to generate predicted ozone concentrations in each sector or grid of the study domain, similar to the output from Eulerian-type models.

Surface observations of wind speed and direction are used to calculate the path of the plume (trajectory) during each hour from midnight to 9 p.m., for each day of the ozone season. Hourly ozone concentrations predicted with the OZIPM-4 model are assigned to each hour of each trajectory, according to the plume birth hour and age. As the plume passes over the gridded study area, hourly ozone increments due to the plant emissions are collected for each grid location.

Surface observations of wind speed and direction are used to calculate the path of the plume (trajectory) during each hour from midnight to 9 p.m., for each day of the ozone season.

These hourly ozone increments are then used to calculate the daily peak ozone increment for each day of the ozone season, in each grid location. The predicted daily peak ozone increment, as well as the measured Southeast Reference site background concentration, are reported for each grid, on each day that the combined total of the background concentration and the increment is greater than or equal to 80 ppb. These results are used for the health effects portion of the fuel cycle analysis of the coal-fired power plant. Background ozone concentrations are discussed in Section 3.8.2.

Alternatively, the hourly ozone increments, at each grid location, are used to calculate the 9 a.m. to 9 p.m. seasonal average ozone increment for the crop effects portion of the study.

Figure 19 shows a flowchart of the MAP-O_3 model. The first module of the model (trajectory module) reads the surface observations meteorological data file from the National Climatic Center (format 1440). Observations from the Knoxville, McGhee Tyson Airport for 1990 were used for the Southeast Reference site. The trajectory module calculates coordinates for plume trajectory endpoints for plume trajectories during the period from midnight to

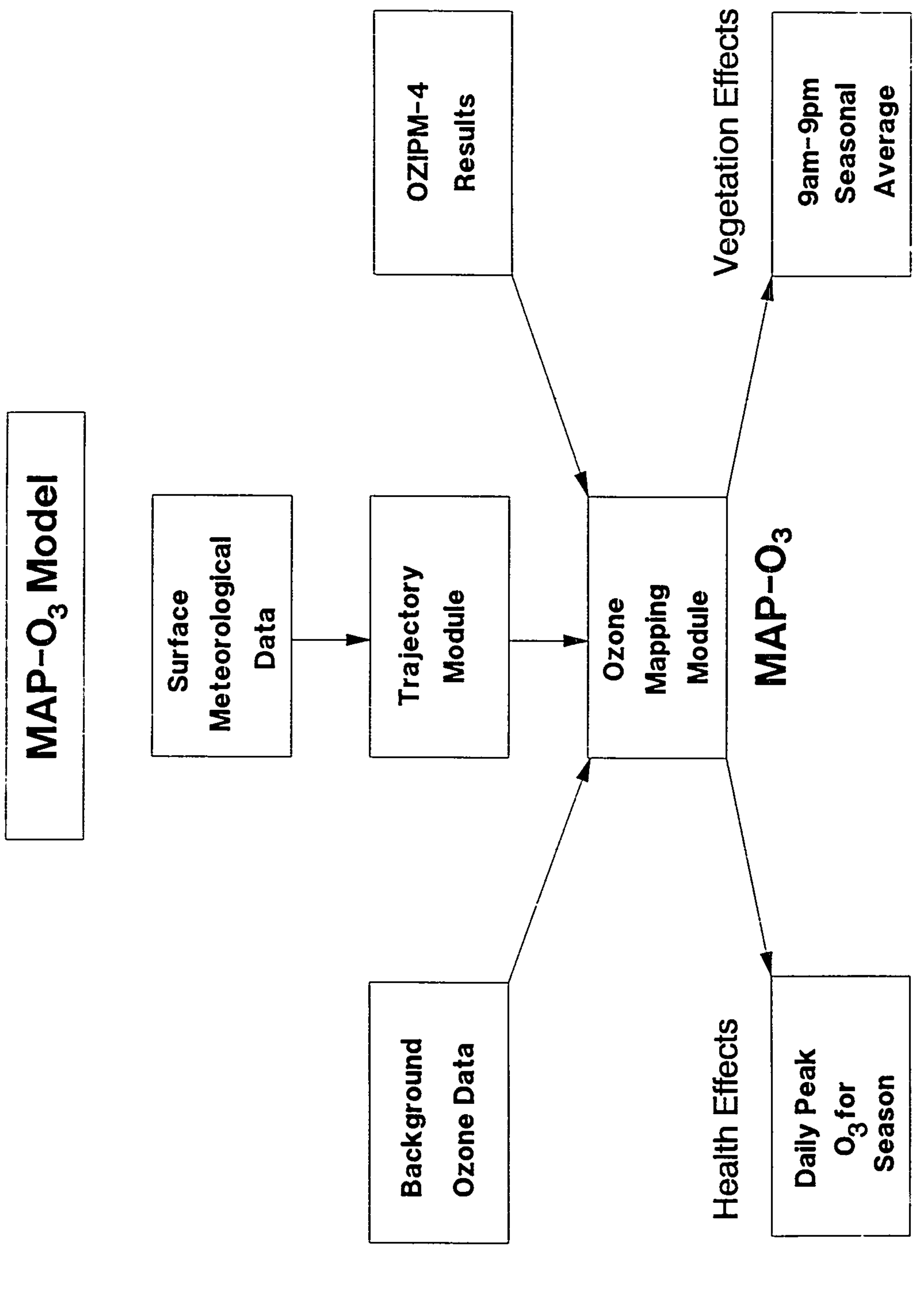

Fig. 19. Flowchart for the MAP-O_3 model

9 p.m. for the entire ozone season (May 1 to September 30). These plume trajectories vary in length from 21 hours to 1 hour (plume ages) and correspond to plumes which originate between midnight and 8 p.m. (birth hours 0 to 20).

The ozone mapping module of the MAP-O_3 model assigns ozone concentrations to each hour of each plume trajectory. Recall, that there are three sets of hourly ozone concentrations from the OZIPM-4 that correspond to low, median and high ozone conditions. The ozone mapping module reads a file containing the peak daily monitored ozone concentration for each day of the ozone season (153 days) and based on this ozone concentration assigns a designation of low, median or high to each day. Plume trajectories on high ozone days have ozone concentrations that correspond to high ozone conditions, trajectories on median ozone days have concentrations that correspond to median conditions and so on, until each hour of each plume trajectory has a predicted ozone concentration.

As discussed in Section 3.3, the cumulative frequency distribution of peak daily ozone concentrations from the Knoxville, Rutledge Pike monitoring station (Fig. 5) was used to assign designations of low, median, and high to each day of the ozone season, based on the assumption that the top third days of the cumulative frequency distribution of ozone concentrations are 'high' days, the middle third of the days are 'median' days and the lower third are 'low' days. Based on this distribution, high ozone days have peak daily ozone concentrations greater than 80 ppb, low ozone days have peak daily ozone concentrations less than 60 ppb and median ozone days have peak daily ozone concentrations between 60 and 80 ppb.

3.8.1 Polar Grid

In the MAP-O_3 model, the receptor grid over which plume trajectories are mapped is a polar grid. The grid has sixteen sectors and 34 downwind distances. A preliminary trajectory analysis using meteorological data for the month of June showed that the average hourly displacement of the plume from the source was approximately 10 kilometers. Therefore, this distance was chosen as the ring spacing for the polar grid. The first ring was spaced five kilometers from the source and every ring thereafter was spaced every ten kilometers, out to 325 kilometers. The first ring was spaced only half the expected plume displacement so that the plume trajectories will tend to terminate in the center of each cell. Table 14 shows the frequency with which plumes terminated at each of the downwind distances in the polar grid surrounding the power plant during the month of June. Ring distances were spaced out to 325 kilometers based on the decreasing frequency of plume impacts seen in Table 14. Plume trajectories during different times of the ozone

Table 14. Frequency of number of times that plume trajectories terminate at each of 34 radial distances of the polar grid surrounding the power plant during June 1990

Distance (km)	Count (hr)	Frequency (%)
0 to 5	204	2.9
5 to 15	1292	2.9
15 to 25	926	18.6
25 to 35	698	13.4
35 to 45	626	10.1
45 to 55	504	9.0
55 to 65	419	7.3
65 to 75	370	6.0
75 to 85	353	5.3
85 to 95	240	5.1
95 to 105	211	3.5
105 to 115	197	3.0
115 to 125	156	2.8
125 to 135	143	2.3
135 to 145	115	2.1
145 to 155	99	1.7
155 to 165	98	1.4
165 to 175	76	1.4
175 to 185	59	1.1
185 to 195	41	0.9
195 to 205	22	0.6
205 to 215	16	0.3
215 to 225	17	0.2
225 to 235	7	0.2
235 to 245	11	0.1
245 to 255	3	0.2
255 to 265	5	0.0
265 to 275	3	0.1
275 to 285	5	0.0
285 to 295	4	0.1
295 to 305	2	0.1
305 to 315	3	0.0
315 to 325	3	0.0
> 325	2	0.0
		0.0
Total:	6930	100

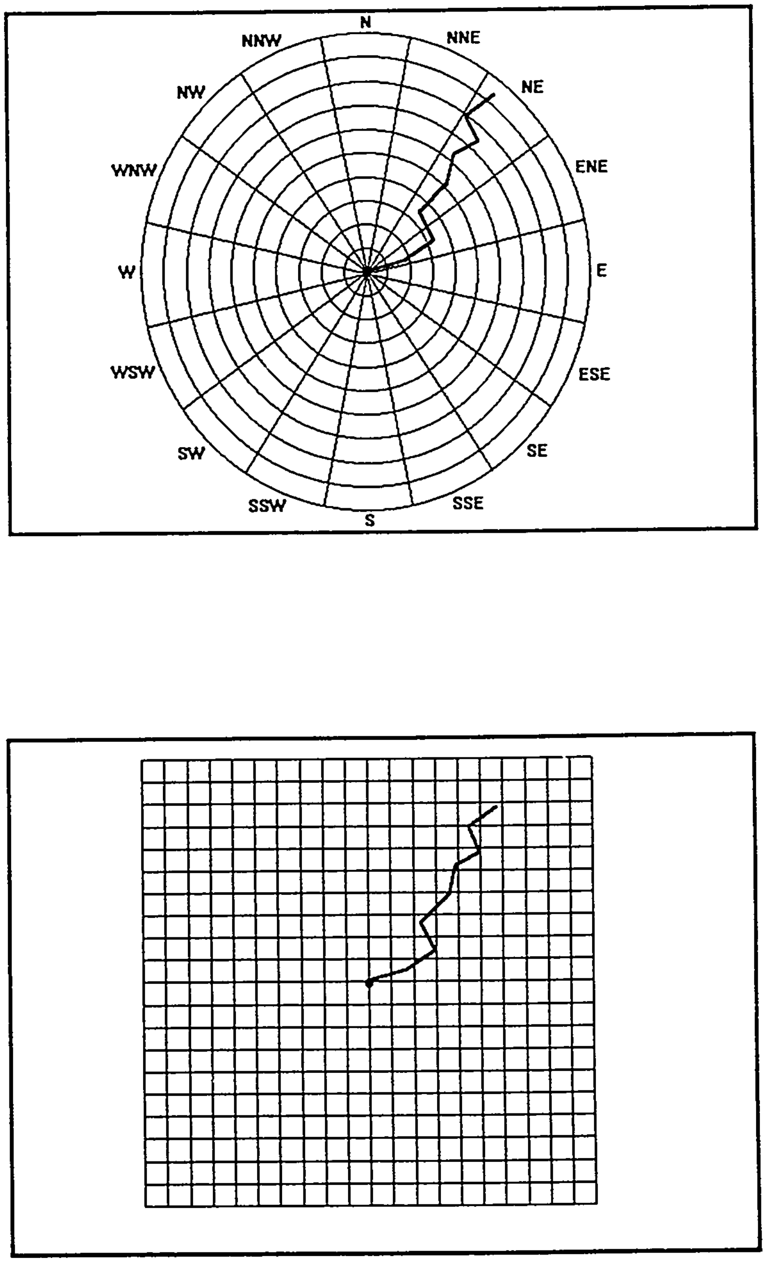

Fig. 20. Polar and rectangular receptor grids showing an eight hour plume trajectory

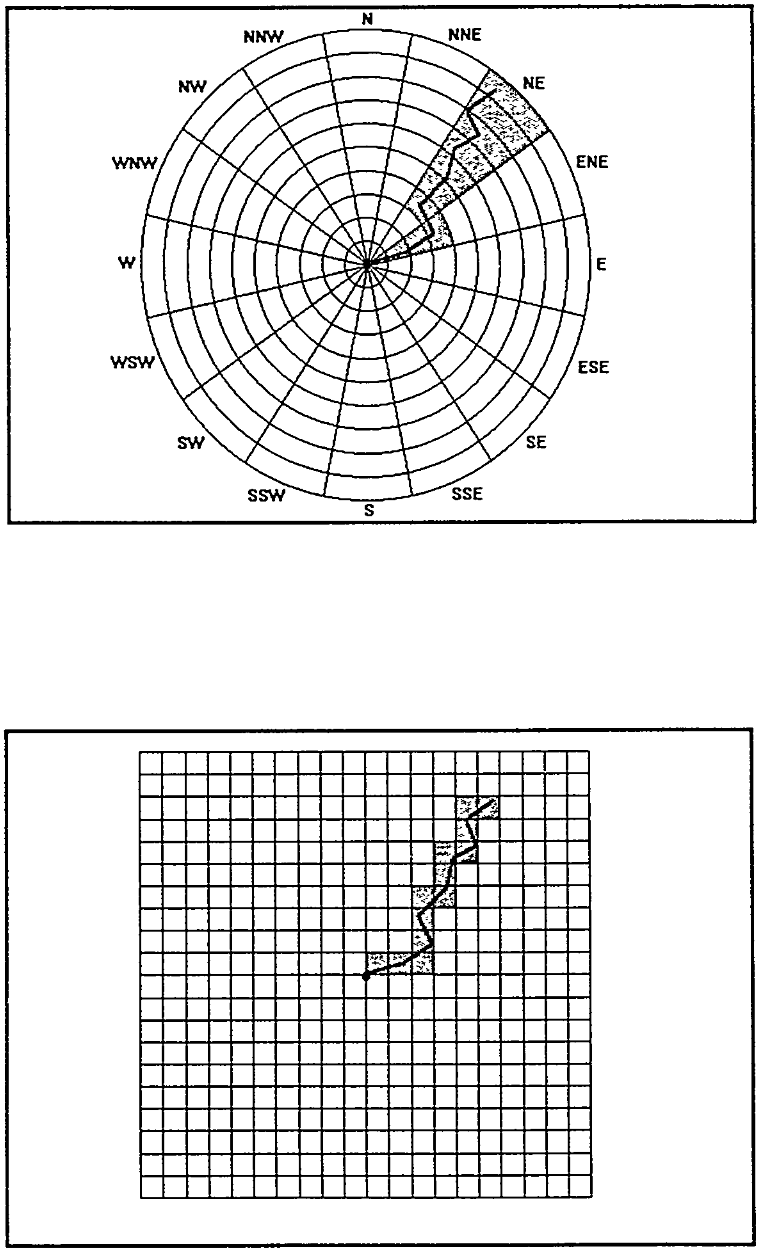

Fig. 21. Polar and rectangular receptor grids showing dispersion of an eight hour plume trajectory

season may travel longer distances, however 325 kilometers was thought to adequately account for the majority of plume trajectory distances.

There are several reasons for using a polar grid in the MAP-O_3 model, over the rectangular grid found in many air quality models. Essentially, a polar grid is thought to be more representative of the actual horizontal dispersion of a plume from a stack than a rectangular grid. Figure 20 shows an eight hour plume trajectory located at the center of both a rectangular and polar grid. The cell size in each grid (length and width of rectangular grids or ring spacing in polar grid) is intended to be similar in scale to the average hourly displacement of the plume. (This spacing would be approximately ten kilometers for the Southeast Reference site.) In Fig. 21 each cell that is impacted by the plume trajectory is shaded. The area that is shaded for the rectangular grid is smaller than for the polar grid, and does not account for the fact that the plume has spread more horizontally at greater downwind distances. The polar grid does account for the widening of the plume with downwind travel distance. In the MAP-O_3 model, ozone concentrations that occur during each hour of this example trajectory, occur in each shaded cell of the polar receptor grid.

The polar grid sectors were spaced every 22.5 degrees, since this angle is thought to adequately account for the horizontal spread of the plume over the ozone season. Typically, in Gaussian dispersion models the horizontal and vertical spread of the plume is defined in terms of σ_y and σ_z. (σ_y and σ_z are the standard deviations of the concentration distribution in the horizontal (y) and vertical directions (z). The most widely used method for determining σ_y and σ_z was developed by Pasquill (1961) and Gifford (1961) and modified by Turner (1967). The Pasquill-Gifford curve (σ_y versus downwind distance) was used to estimate horizontal plume widths. Table 15 shows σ_y values for each stability class, at 10 kilometers from the stack. The plume width, defined as $4.3\sigma_y$, is also shown for each stability class. (The plume width is defined here as $4.3\sigma_y$ so that the concentration at the plume edge is 10% of the plume centerline concentration.)

The horizontal plume spreading angles are also shown in Table 15. For stabilities D, E and F the spreading angles are 14, 10 and 7 degrees, respectively. For stabilities A, B and C they are 38, 29 and 21 degrees, respectively. As stated earlier the most frequently occurring stability during summer daytime hours is B stability (moderately unstable). Based on these results, it seems reasonable that ozone concentrations predicted at ground level with the OZIPM-4 model should be distributed, with the MAP-O_3 model,

Table 15. Horizontal plume spreading angle for each stability class

Stability Class	σ_y (meters)	Plume Width at 10 kilometers (4.3 σ_y) (meters)	Horizontal Plume Spreading Angle (degrees)
A	1600	6880	38
B	1200	5160	29
C	850	3655	21
D	550	2365	14
E	420	1806	10
F	280	1204	7

throughout the study area, based upon a polar grid with 22.5 degree sector spacing. This is also consistent with the way some other dispersion models handle horizontal plume spreading such as the Air Quality Dispersion Model (AQDM) and the Industrial Source Complex Long-Term model (ISCLT). These models do not utilize σ_y, but rather, spread the plume uniformly over 22.5 degree sectors (EPA 1990).

3.8.2 Background Ozone Measurements

In addition to the incremental ozone concentrations due to the coal-fired power plant, both the health effects and the crop effects portions of this study require background ozone concentrations. This section describes the background ozone concentrations that were used in the analysis. Background is used here to refer to ambient ozone concentrations in the vicinity of the Southeast Reference site that are not affected by the power plant emissions. Unfortunately, the nearest ozone monitoring station is located downwind of an urban area and may not be representative of ozone concentrations expected at the Southeast Reference site. This section discusses the implications of this situation and describes adjustments that were made to the measured background ozone concentrations.

The crop effects portion of the study requires a 9 a.m. to 9 p.m. seasonal average ozone concentration. Background ozone concentrations were obtained from monitoring data at the Knoxville, Rutledge Pike station. The twelve hour (9 a.m. to 9 p.m.) seasonal average ozone concentration was calculated from hourly observations during the period May 1 to September 30, 1990. This seasonal average ozone concentration is 53 ppb. Since the Rutledge Pike monitor is located 60 kilometers from the Southeast Reference site and is located within the influence of the urban area of Knoxville, TN, these ozone concentrations are probably higher than would be expected from a monitor located nearer the Southeast Reference site and outside the influence of any urban areas. However, no better monitor location is available. No adjustments will be made to the twelve hour seasonal average ozone concentration based on the Rutledge Pike monitoring data.

The health effects portion of this study requires a daily peak one-hour ozone concentration for each day. The highest ozone concentrations measured at the Knoxville, Rutledge Pike monitor most likely occurred on days when the site was within the influence of the urban plume from the city of Knoxville (population 165,000). For this reason, the highest background ozone concentrations measured at the Knoxville site were adjusted downward, based on modeling results, to better represent the maximum background ozone concentrations expected to occur at the rural Southeast Reference site. Figure 22 shows the top 50% of the cumulative frequency distribution of peak daily one hour ozone concentrations from the Knoxville, Rutledge Pike monitoring station (Fig. 5 shows the entire distribution). Also shown in the figure, are results from the OZIPM-4 simulations for the high and median base case simulations at the Southeast Reference site (triangle markers). The modeled daily peak one-hour ozone concentration under high ozone conditions for the Southeast Reference site was 88 ppb. The modeled daily peak one-hour concentration under median ozone conditions was 56 ppb. These values were expressed as ratios to the corresponding monitored concentrations. The peak daily monitored ozone at the same percentile as 88 ppb was 117 ppb and the peak daily monitored ozone at the same percentile as 56 ppb was 70 ppb. The average of the two ratios was 0.776. This number was used to adjust the monitored daily peak one-hour ozone concentrations from the Knoxville Rutledge Pike monitoring station to be more representative of background conditions within the vicinity of the Southeast Reference site. (The line in Fig. 22 through the Southeast Reference site modeled results (triangle markers) is 0.776 times the regression line through the monitored results.) Based on this adjusted background ozone frequency distribution, high ozone days at the Southeast Reference site have peak daily ozone concentrations greater than 62 ppb and median ozone days have peak daily ozone concentrations between 47 and 62 ppb. These adjusted background concentrations were used for the health effects portion of the fuel cycle analysis of the coal-fired power plant. (Due to the health effects threshold of 80 ppb, no results from the MAP-O_3 model were needed unless the combined total of the background and the increment due to the plant were greater than or equal to 80 ppb.)

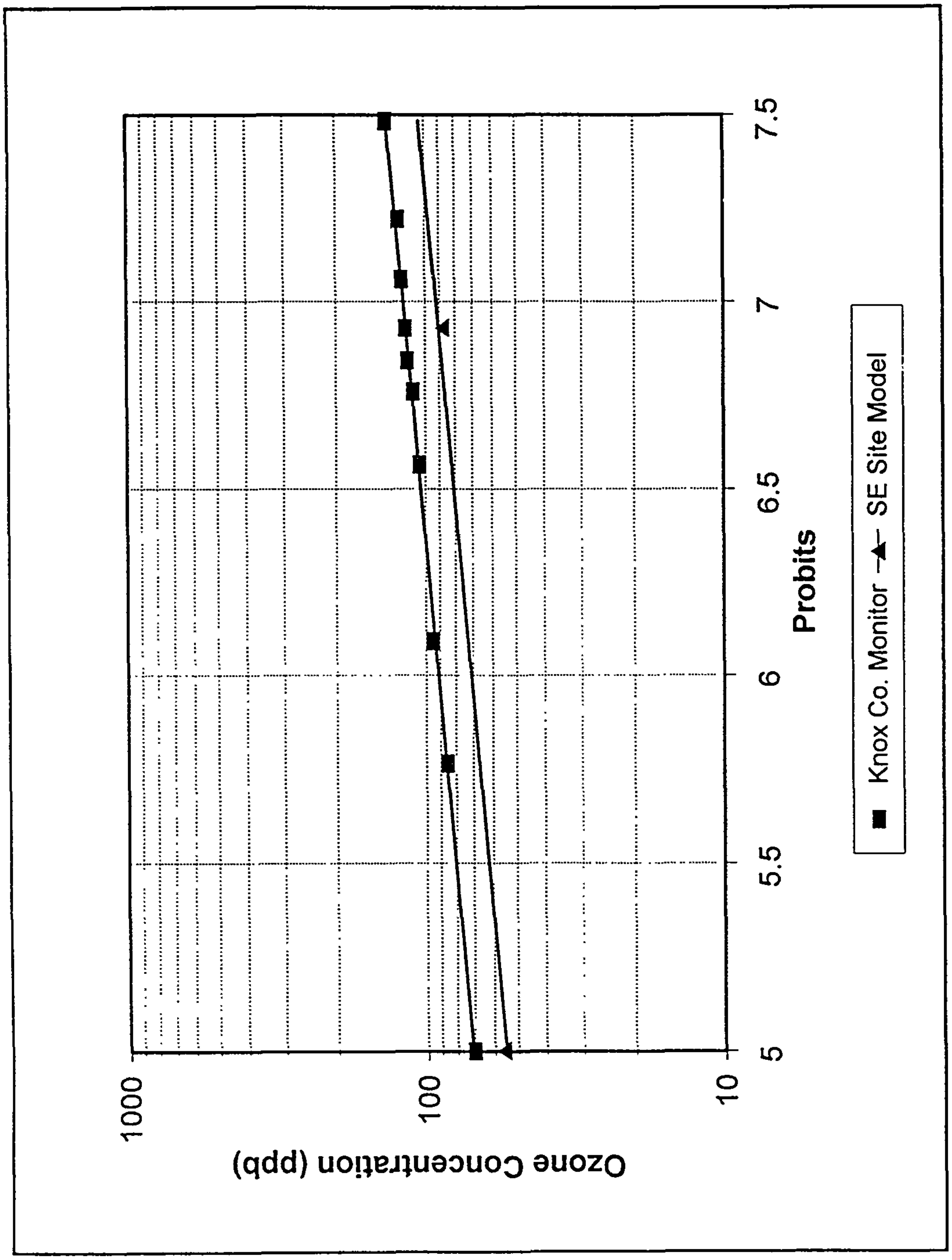

Fig. 22. Monitored and modeled daily peak ozone concentration versus probit for the 1990 ozone season

4. RESULTS

4.1 SEASONAL AVERAGE 9 A.M. TO 9 P.M. OZONE CONCENTRATIONS

Results from the MAP-O_3 model, for the crop effects portion of the coal fuel cycle analysis, are shown on isopleth maps in Figs. 23 and 24. (Results from the MAP-O_3 model were converted to Cartesian coordinates and written to files for import to the isopleth graphing routine SURFER.) The power plant is shown in the center of each isopleth map with a triangle marker. The scale of each figure is in kilometers from the plant. Incremental ozone concentrations are reported in ppb. Results are presented separately for two cases; one with and one without ozone depletion. (Ozone concentrations above base case will be referred to as ozone bulges and ozone concentrations below base case will be referred to as ozone depletions.)

Figure 23 shows the predicted impact of the coal-fired power plant emissions on the seasonal 12-hour average ozone concentrations due to ozone bulges only. These results represent an upper bound estimate of the impact of the power plant emissions on ozone concentrations, since ozone scavenging is not accounted for. As seen in Fig. 23, the highest 12-hour seasonal average ozone concentration (based on ozone bulges only) is 1.1 ppb (the smallest isopleth line) and occurred approximately 40 kilometers from the plant in the east northeast (ENE) direction. The lowest isopleth plotted in Fig. 23 is 0.01 ppb. This seasonal average ozone concentration occurred as far away as 300 kilometers from the plant in the northeast direction (NE) and 180 kilometers in the southwest (SW) direction.

Figure 24 shows the predicted impact of the coal-fired power plant emissions on the seasonal 12-hour average ozone concentrations due to both ozone bulges and depletions. These results represent the lower bound estimate of the impact of the power plant emissions on ozone concentrations. The highest increase in 12-hour seasonal average ozone concentration is 0.7 ppb (the smallest isopleth line) and occurred approximately 40 kilometers from the plant in the east northeast (ENE) direction. The lowest positive isopleth plotted in Fig. 24 is 0.01 ppb. This seasonal average ozone concentration occurred as far away as 200 kilometers from the plant in the northeast direction (NE) and 180 kilometers in the southwest (SW) direction. Ozone depletion is shown in Fig. 24 with hatched isopleth lines (depressions). The hatched isopleth lines in Fig. 24 are -0.2 and -0.1 ppb. Ozone depletion occurred mostly in the immediate vicinity of the plant and again approximately 80 kilometers from the plant in the southeast (SE) direction.

In addition to the results seen in Figs. 23 and 24, the seasonal 12-hour average measured background ozone concentration of 53 ppb was also used in the crop effects portion of the study.

4.2 DAILY PEAK OZONE CONCENTRATIONS

Results from the MAP-O_3 model for the health effects portion of the fuel cycle analysis are in tabular form. (The computer printout is too lengthy to include here.) The peak daily ozone increment due to the power plant, as well as the adjusted Southeast Reference site daily peak background ozone concentration, are reported at each location in the polar grid (each downwind distance and sector) for each day of the ozone season (provided the combined total of the background and the increment due to the plant were greater than or equal to 80 ppb). This criterion was met (and results were reported) for fifty-one days during the 1990 season. These days were the fifty-one 'high ozone' days in the ozone season. One of the fifty-one high days was in the month of May, eleven were in June, fifteen were in July, fifteen were in August and nine days were in September.

As mentioned earlier, the definition for a high ozone day, was the top one third (51/153 days) of days according to the peak daily ozone concentrations (See Fig. [5] for the frequency distribution of peak daily ozone concentrations). Due to the health effects threshold of 80 ppb, only results for high ozone days were included since the highest median base case concentration of 62 ppb plus the highest ozone increment due to the plant during median conditions was 11 ppb for a combined total of 73 ppb, which is well below the threshold of 80 ppb.

As stated above, results for the health effects study are in tabular form and correspond to fifty-one days of the ozone season. (If the actual results used in the health effects portions were presented here graphically it would require 51 figures, one for each day.) Figure 25 is provided here, to illustrate the spatial distribution of daily peak ozone concentrations during the 1990 ozone season at the Southeast Reference site. The ozone concentrations shown in Fig. 25 are the increases in the maximum daily peak ozone concentrations at each location in the receptor grid. As seen in Fig. 25 the largest increase in daily peak ozone concentration due to the power plant emissions during the ozone season, was 26 ppb, occurring within 20 km of the plant. An increase of 25 ppb in the peak daily ozone concentration occurred over a wider area, from 120 km in the northeast direction to 80 km in the southwest direction. An increase in daily peak ozone concentration of 1 ppb was seen as far away as 200 km in the northeast direction and 130 km in the southwest direction.

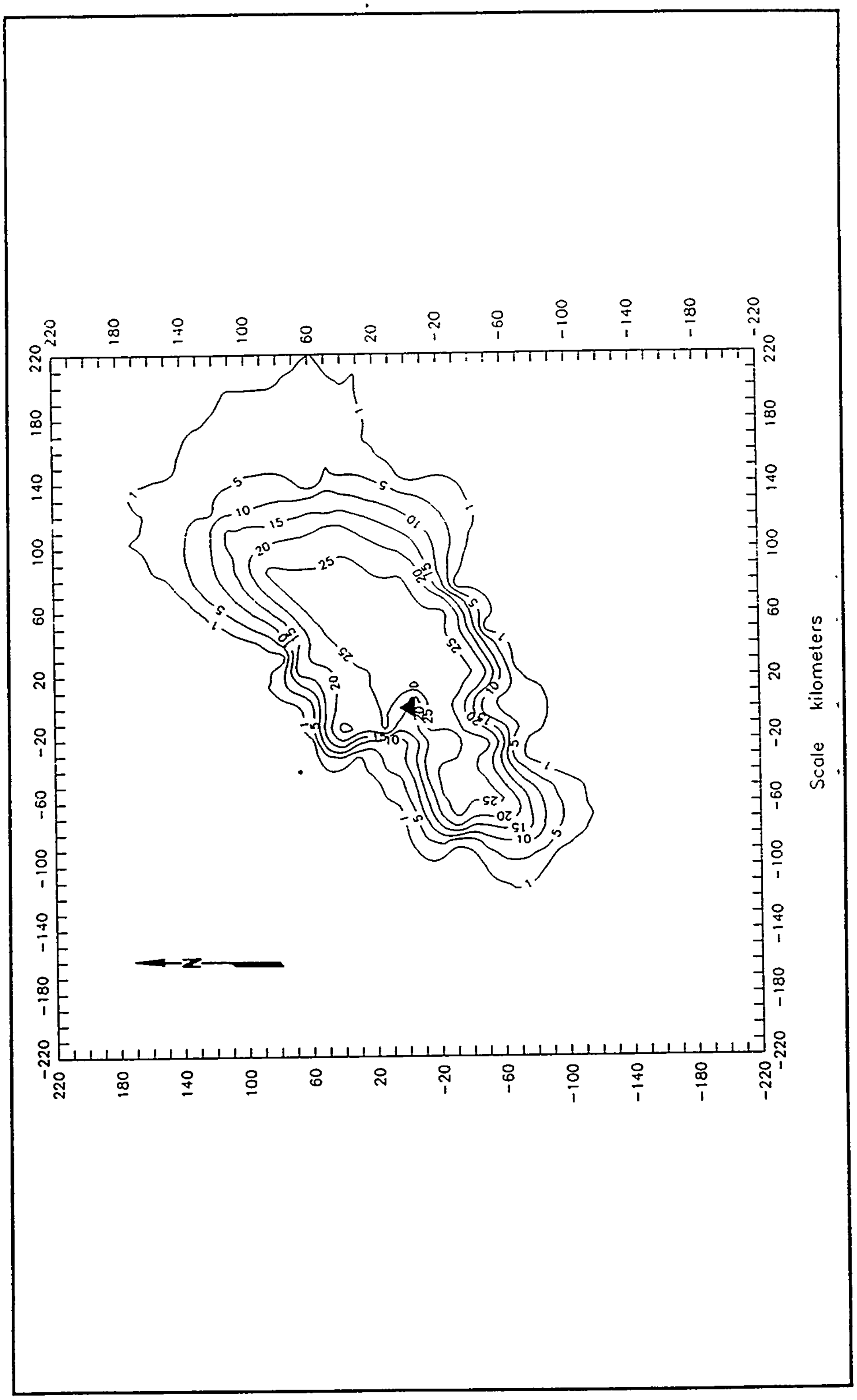

Fig. 23. Positive incremental 9 a.m. to 9 p.m. seasonal average ozone concentrations (ppb) for May to September 1990 due to emissions from the coal-fired power plant located at the Southeast Reference site (positive concentrations are above ambient)

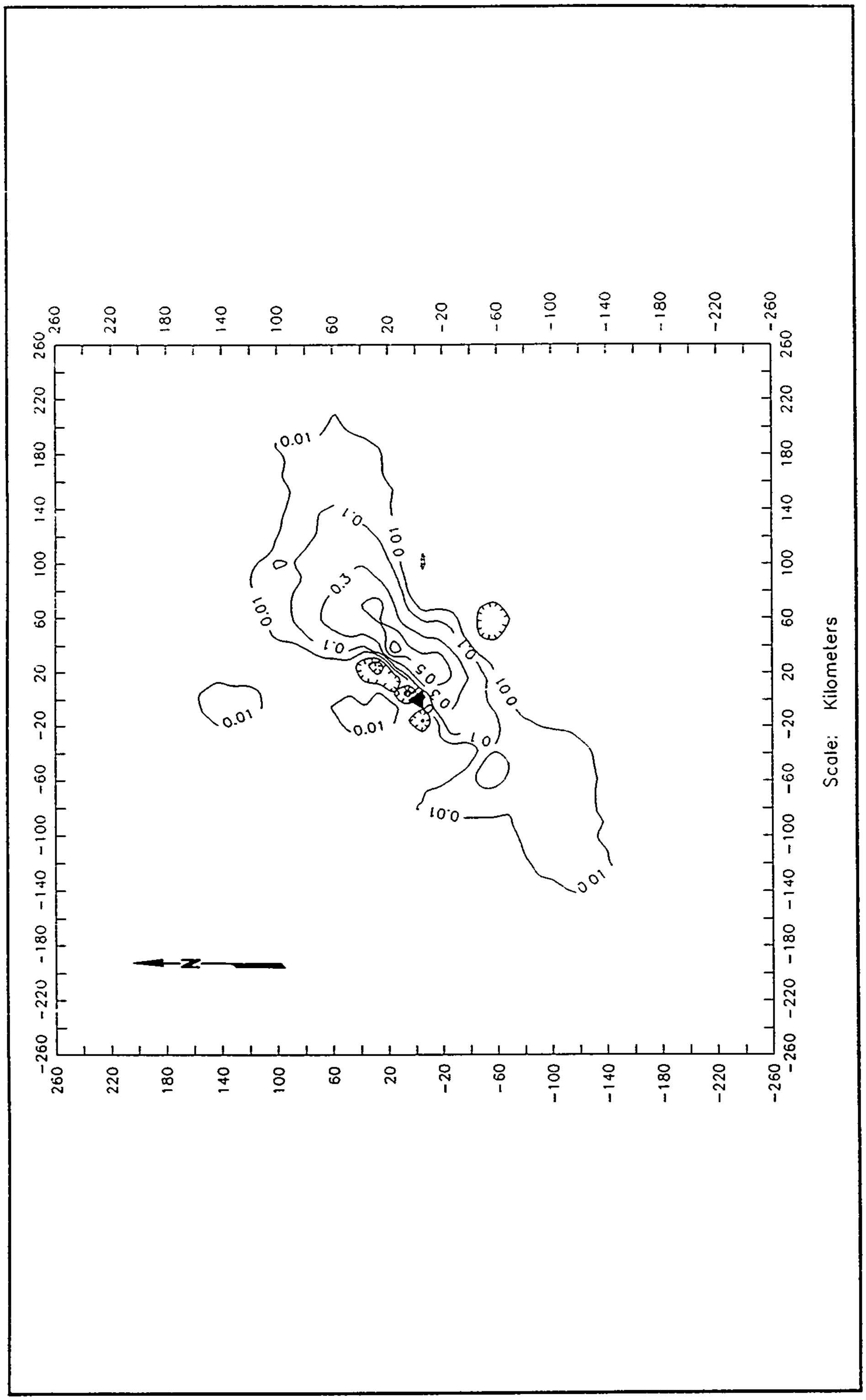

Fig. 24. Total incremental 9 a.m. to 9 p.m. seasonal average ozone concentrations (ppb) for May to September 1990 due to emissions from the coal-fired power plant located at the Southeast Reference site (total concentrations include both positive and negative incremental concentrations

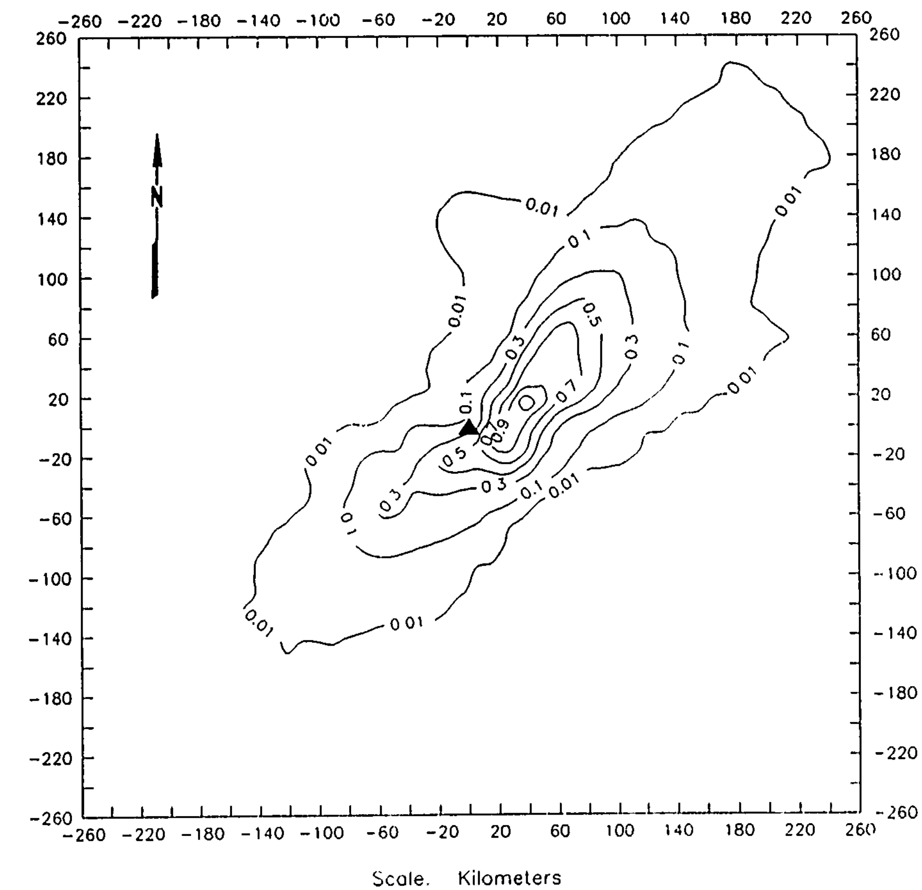

Fig. 25. Maximum daily peak incremental ozone concentrations (ppb) (one-hour average) for May to September 1990 due to emissions from the coal-fired power plant located at the Southeast Reference site

5. MODELING ASSUMPTIONS AND LIMITATIONS

In any modeling application, it is important to note the limitations and assumptions of the model. Model assumptions specific to the OZIPM-4 model are discussed in Section 3.2. This section refers to assumptions and limitations of the general modeling methodology and the MAP-O_3 model.

1. Three sets of base case conditions (base case scenarios) are used to model the range of conditions which occur over the entire ozone season.

2. The top one third of ozone season days are described as 'high' ozone days, the middle third are described as 'median' ozone days and the lowest one third are described as 'low' ozone days.

3. Biogenic NMOC emissions are assumed to be spatially constant throughout the study area and to vary with time of day.

4. Anthropogenic emissions of NO_x and NMOC emissions are assumed to be uniformly distributed throughout the study area and throughout the day.

5. Solar radiation is varied by selecting calendar days which represent a range of solar intensities.

6. Concentrations of ozone and ozone precursors transported aloft in rural areas on high ozone days could be expected to be as high as concentrations transported aloft in some urban areas.

7. The NO_x emission flux used to simulate emissions from a power plant in the OZIPM-4 model is not appropriate for time periods less than one hour.

8. The combined average of the observed 10-meter wind speed and the calculated stack top wind speed is representative of wind speeds transporting pollutants in the plume downwind of the power plant stack.

9. NO_x emissions from the power plant during the period 9 p.m. to midnight are not expected to have a significant impact on ozone concentrations the following day and are not handled by the model.

10. NO_x emissions from the power plant during the period midnight to 9 a. m. are assumed to be transported aloft (with additional horizontal dispersion) until 10 a.m. at which time they are mixed to the ground.

11. NO_x emissions from the power plant during the period 10 a. m. to 8 p.m. are assumed to be mixed to the ground within the first hour of travel time.

12. The same assumptions of atmospheric mixing apply to all three base case conditions.

13. The modeling methodology assumes no carryover of emissions or precursor concentrations from the previous day, i.e. each day is modeled independently.

The modeling methodology assumes no carryover of emissions or precursor concentrations from the previous day, i.e. each day is modeled independently.

14. Polar sectors of 22.5 degrees adequately account for horizontal dispersion of a plume (especially during daylight hours) from a power plant over the ozone season.

15. Daily peak background ozone concentrations from the Rutledge Pike monitor are adjusted by the ratio (0.776) of measured concentrations to modeled concentrations (for the 50th and 96th percentiles) at the Southeast Reference site.

16. In calculating one-hour ozone concentrations, the model does not account for horizontal dispersion beyond the first hour that the plume mixes to the ground.

6. AIR QUALITY IMPACT OF NO_x EMISSIONS ON OZONE AT THE SOUTHWEST REFERENCE SITE

The impact of NO_x and NMOC emissions from the coal-fired power plant located at the Southwest Reference site on ozone concentrations is expected to be significantly less than at the Southeast Reference site for several reasons. Differences between the sites are significant with respect to several important background conditions which lead to the formation of ozone. These conditions include meteorology, biogenic NMOC emissions, ratios of NMOC to NO_x emissions and ambient ozone concentrations. These differences are discussed briefly here.

Frequent and prolonged periods of stagnation can lead to the accumulation of ozone and its precursors in the atmosphere. Warm, moist slow-moving tropical air that pervades the southeastern U.S. during summer months contributes to frequent stagnation conditions (Reisinger and Valente 1985) which can lead to high concentrations of ozone. In contrast, meteorological data collected near the southwest site indicate that good dispersion conditions (Pasquill stability classes A through D) occur, on average, 56% of the time (U. S. DOI 1983). The poorest dispersion is found during nighttime when the near-surface air is stable, with low wind speeds.

In general, better dispersion occurs with greater mixing depth. Mixing heights which are a measure of the thickness of the layer of the lower atmosphere in which air pollutants are mixed are significantly different for the Southeast and Southwest sites. According to Holtzworth (1972) the mean summer afternoon mixing height in the vicinity of the Southwest site is approximately 4,000 meters. The corresponding value for the Southeast site is 1,800 meters.

Another notable difference between the two sites is the occurrence of biogenic NMOC emissions which contribute to the formation of ozone and should be included in any ozone assessment. Significant biogenic emissions occur from the forest that covers most of the 1,364 acres surrounding the Southeast site. Much smaller quantities of biogenic emission are expected from the semiarid grass and shrubland vegetation that surrounds the Southwest site.

It seems unlikely that the addition of more NO_x emissions from the power plant to an area characterized by a very low NMOC:NO_x emission ratio will result in increased ozone concentrations at the Southwest site.

The photochemical formation of ozone in the troposphere is a function of the NMOC:NO_x ratio of precursors. An optimum NMOC:NO_x precursor concentration will result in a maximum ozone concentration. According to data reported in the New Mexico Generating Station Environmental Impact Statement, 1980 anthropogenic emissions densities of NO_x and NMOC within the San Juan Air Basin were estimated to be 11.4 kilograms per square kilometer per day (kg/km^2-day) and 1.97 kg/km^2-day respectively. The average emission densities in 13 counties surrounding the Southeast Reference site, are 11.2 kg/km^2-day for NO_x and 15.7 kg/km^2-day HC (17.0 kg/km^2-day, including biogenic emissions). The Southwest Reference site emission inventory derived ratio of NMOC:NO_x emissions is 1.5. The corresponding value for the Southwest site is 0.17. It seems unlikely that the addition of more NO_x

emissions from the power plant to an area characterized by a very low NMOC:NO_x emission ratio will result in increased ozone concentrations at the Southwest site. There simply may not be NMOC emissions in sufficient quantities to react with NO_x emissions from the power plant. (The NMOC:NO_x ratios discussed here are for comparison purposes only and should not be confused with ambient ratios which are expressed on a volume basis.)

Additionally, ambient ozone concentrations at each site are very different. Fig. 26 is a plot of the monthly average peak daily one-hour ozone concentration for four ozone monitoring stations in the EPA AIRS database. The two southeast monitors are Rutledge Pike and Mildred Drive, located approximately 60 kilometers from the Southeast site. The two southwest monitors are Petrified Forest National Park and Bernalillo County, New Mexico, both located approximately 190 km from the Southwest site. As shown in the figure, significant differences (up to 28 ppb) between the two regions occur in the monthly average, daily peak 1-hour ozone concentration from June to September.

Finally, the arithmetic mean 1-hour ozone concentration for the third quarter of 1990 at the two southeast sites was 77.6 and 74.9 ppb. The corresponding values at the southwest sites were 54.1 and 53.3 ppb. Any small incremental increase to the ambient ozone concentration from the power plant is unlikely to bring the total ozone concentration above 80 ppb (selected as a practical threshold for health effects portion of this study).

Therefore, the impact of NO_x emissions from the power plant located at the Southwest Reference site on ozone concentrations is expected to be significantly less than at the Southeast Reference site and for this reason, a detailed modeling analysis was not undertaken.

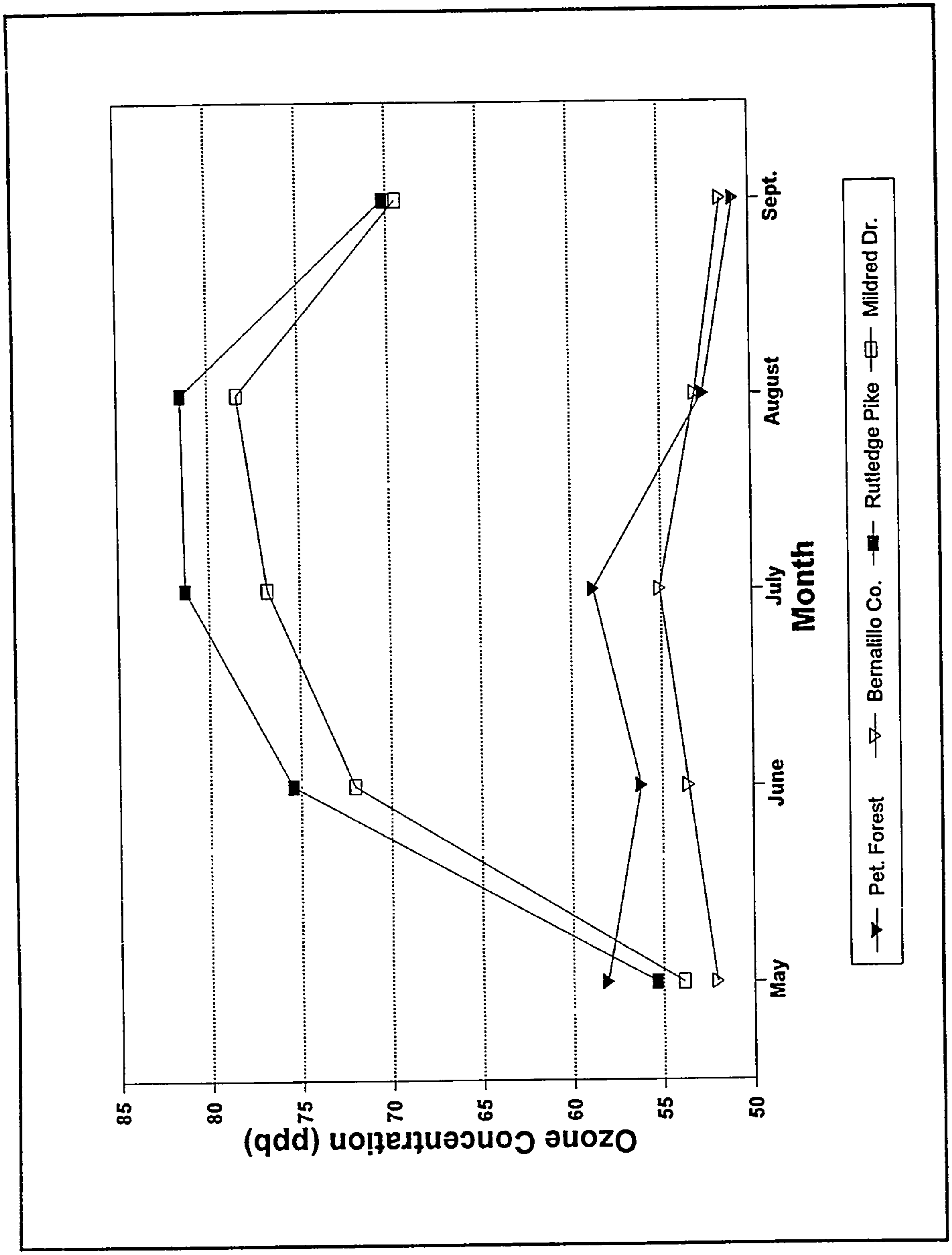

Fig. 26. Monthly average daily peak one-hour ozone concentrations for monitors located near the southeast and southwest sites

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PART II

ECOLOGICAL EFFECTS

PAPER 4 ECOLOGICAL EFFECTS OF FUEL CYCLE ACTIVITIES

PAPER 4[*]

ECOLOGICAL EFFECTS OF FUEL CYCLE ACTIVITIES

1. INTRODUCTION

The purpose of this paper is to summarize the approach used to characterize ecological impacts of the coal fuel cycle. The same approach is used for many of the impacts in other fuel cycles as well. The principal analytical approach being used in the study is an accounting framework — that is, a series of matrices that map each phase of the fuel cycle to a suite of possible emissions, each emission to a suite of impact categories, and each impact category to an external cost. This paper summarizes the ecological impacts of all phases of the coal fuel cycle, defines the ecological impact categories used in the study's "accounting framework," and discusses alternative approaches to quantification.

Externalities associated with CO_2-induced global climate change are beyond the scope of this paper and are not discussed. For some other potential impacts, much of the available literature reflects historical operating practices or noncompliance with standards. These impacts include (1) all aspects of coal mining, cleaning, and beneficiation, and (2) thermal discharges, aqueous effluents, and solid waste disposal from coal combustion. Externalities associated with these components of the fuel cycle are expected to be either small or localized, provided that all facilities are operated in compliance with Federal and State standards. For the remaining impact categories and classes of emissions, at least some evidence suggests that compliance with standards may not be sufficient to eliminate ecological externalities. These categories include (1) effects of gaseous air pollutants on vegetation and crops; (2) effects of acid deposition on crops, forests, and recreational fisheries; (3) effects of mine drainage on recreational resources; and (4) effects of long-range transport and deposition of

... some evidence suggests that compliance with standards may not be sufficient to eliminate ecological externalities.

[*] Based on a working paper by L. Barnthouse, G. Cada, R. Kroodsma, D. Shriner, V. Tolbert, and R. Turner.

mercury. For these categories we evaluate the available literature and, where feasible, recommend quantification methods.

2. OVERVIEW OF ENVIRONMENTAL IMPACTS OF THE COAL FUEL CYCLE

2.1 AQUATIC RESOURCES

Mining, transportation, and energy production from coal have a variety of potential impacts on aquatic resources. Principal concerns have historically included impacts of mining and coal cleaning on water quality, impacts of combustion-related aqueous discharges, and impacts of power plant cooling systems on fish and other biota. More recently, regional and global effects of acid deposition, CO_2 release, and heavy metals have become major concerns.

2.1.1 Coal Mining

Surface mining alters the topography and vegetative cover of land, which in turn affect the volume and rate of surface water runoff. Storm water runoff is often higher and carries a larger load of suspended materials where vegetation has been reduced. On the other hand, surface discharges may be reduced during nonstorm periods, especially if streams flow into mining-induced fractures or subsidence areas.

Cushman et al. (1980) reviewed the extensive literature on potential hydrologic and water quality effects of coal mining. They grouped surface water quality effects of coal mining into acid drainage, increases of other dissolved constituents (e.g., alkaline drainage and trace substances), and increased suspended solids loads. Mining-induced alterations in pH and dissolved contaminants can degrade groundwater as well, making the water unsuitable for consumption. Acid drainage from mines may continue long after their closure. Alkaline drainage is a problem in western coal fields, often compounded by the low dilution potential of many western streams.

The increased variability in streamflows can affect aquatic biota by reducing habitat, while floods will increase bottom scouring, and sedimentation will fill pools and degrade riffle habitat needed by bottom-dwelling organisms. Damage to aquatic life from acid mine drainage is attributed to a combination of low pH, high levels of metals and sulfates, and deposition of a blanket of iron hydroxide precipitates that smother bottom-dwelling organisms (Dvorak and Lewis 1978). Biotic impacts from alkaline mine drainage, on the other hand, result from turbidity, sedimentation, and osmotic stress from high dissolved solids concentrations. Often, these water quality and biotic impacts associated with mine drainage are most pronounced in small streams and are more likely to affect recreational fishing and biological diversity than commercial fishing.

Although underground mining generally has less effect on soils and vegetation than surface (strip) mining, subsidence of shallow, mined out spaces can also have major effects on topography and surface water runoff. Underground coal mining often causes greater quantities of acid or alkaline drainage than surface mining, but the relatively smaller amount of soil disturbance results in less soil erosion and sedimentation.

2.1.2 Coal Cleaning and Beneficiation

Coal cleaning and beneficiation produce both solid wastes and liquid effluents. Effects of coal processing are similar to those of mining, although of lesser magnitude (Cushman et al. 1980). Air and water can percolate through refuse piles, resulting in acidic or alkaline leachates. For example, processing and refuse leachates have been estimated to contribute 7.5% of the acid drainage in Appalachia (Dvorak et al. 1977). Uncontrolled discharges from refuse piles and holding lagoons can degrade the quality of both surface and ground waters. As with mining effects, waters affected by coal processing operations may exhibit altered pH and increased dissolved solids, including contamination by heavy metals and hydrocarbons.

Recreational fishing, biodiversity, and consumptive water uses could all be constrained by water quality degradation resulting from coal processing facilities.

2.1.3 Generation

Water quality in receiving streams could be affected by operation of the condenser cooling system, discharges from air pollution control equipment, and runoff from coal storage and residual waste disposal areas. Operation of the power plants' heat dissipation (condenser cooling) system requires large amounts of water that are usually withdrawn directly from surface waters and subsequently discharged with an added load of heat and chemical contaminants. These surface waters contain aquatic organisms that may be injured or killed through their interactions with the power plant. Because such losses could adversely impact recreational and commercial fisheries, power plant condenser cooling system operations are strictly regulated.

Aquatic organisms that are too large to pass through the intake debris screens and cannot swim away from the intake may be impinged against the screens. Impingement can affect large numbers of fish and shellfish, and can result in mortality if the organisms are suffocated by being held against the screens for long periods of time (Langford 1983).

Aquatic organisms that are small enough to pass through the debris screens will travel through the entire condenser cooling system and be exposed to heat, mechanical and pressure stresses, and possibly biocidal chemicals before being discharged back to the water body. This process, called entrainment, may kill a wide variety of small plants, invertebrates, and fish eggs and larvae (Schubel and Marcy 1978).

Discharges from the condenser cooling system may affect the receiving water body through heat loading and chemical contaminants, most notably chlorine or other biocides. Heated effluents can cause direct mortality among aquatic organisms by either heat shock or cold shock. In addition, there are a number of indirect or sublethal stresses associated with heated discharges that have the potential to alter aquatic communities, such as increased incidence of disease, predation, or parasitism, as well as changes in dissolved gas concentrations (Langford 1983).

Coal pile runoff is a potential problem at coal-fired power plants. If runoff is not carefully monitored and controlled, coal fines as well as leachates containing organic compounds, heavy metals, and other contaminants may degrade surface waters or groundwater. Water quality degradation from coal pile runoff can have effects similar to those of acid mine drainage (for high-sulfur coals) or alkaline mine drainage (for low-sulfur coals)(Dvorak and Lewis 1978).

Coal combustion produces ash and slag that require disposal. The ash is often slurried to a holding pond, and the slurry water is recycled, evaporated, or discharged. Potential effects on water quality from solid waste collection and disposal stem from increases in trace contaminants, pH, chemical oxygen demand, and dissolved and suspended solids (Cushman et al. 1980). Ash pond overflow, dike failure, ash-slurry pipeline rupture, and site erosion may introduce high levels of dissolved and suspended solids to surface waters. Leachates from stored or land-filled ash or scrubber sludge may also have chronic effects on both surface and groundwater quality, the extent of which is related to such factors as soil type, depth of the water table, effectiveness of pond sealing, dilution, and element mobility (Cushman et al. 1980).

Aquatic resources can be indirectly affected by atmospheric emissions from coal-fired power plants. Sulfur and nitrogen oxides contribute to acidic deposition; surface waters may become acidified from either direct deposition (both wet and dry fallout) or runoff of acidic water. The resulting lowered pH and elevated heavy metals concentrations have effects on aquatic communities similar to those of acid mine drainage. Acidic deposition is believed to have contributed to the severe reduction of biotic diversity and reproductive failure of fish species in some lakes in the northeastern United States. Recent studies have documented significant increases in the mercury content of fish in certain areas (Travis and Hester 1991); long-range transport of atmospheric mercury from combustion of coal is believed to be a major cause of this accumulation (Mitra 1986; Nriagu and Pacnya 1988).

2.2 TERRESTRIAL RESOURCES

Activities associated with the various stages of the coal fuel cycle can have impacts on terrestrial biota and other land resources (e.g., crops, pasture, forests, orchards, threatened and endangered species, game animals, and public and private recreation lands). Principal concerns are the impacts of power-plant gaseous emissions on plants and the impacts resulting from occupation of land by surface mines, generating facilities, and solid waste disposal areas.

2.2.1 Coal Mining

Underground mines have relatively little effect on terrestrial resources because only small areas are typically disturbed. Impacts of surface mining can, however, be substantial.

The principal types of surface mining of coal are contour mining and mountain top removal in mountainous terrain and area strip mines in non-mountainous areas. Contour mining and mountaintop removal in mountains (e.g., the Cumberland Mountains of Tennessee) usually result in a loss of forested area on the mine itself and reduced productivity of adjacent downslope areas that receive discarded stripped material (spoil). When spoil is deposited downslope from a contour mine, an area usually more than twice the size of the mine itself is directly impacted (Dvorak et al. 1977). Extensive downslope erosion and gullying may also result, particularly where timber cuts occur near the mines. Many currently active contour and mountaintop mines and some old ones are being reclaimed; that is, the mine may be revegetated, or the land may be restored to original contour before being revegetated (Vogel 1981). However, reclaimed areas usually have significantly lower productivity than undisturbed areas.

Contour mines usually occupy only a small fraction of mountainous land. Therefore, they may have minimal impact on forest wildlife of the mountains and, when reclaimed, may provide herbaceous and brushy habitats of value to game species such as ruffed grouse and white-tailed deer (Samuel et al. 1978; Vogel 1981). However, impacts may be relatively adverse if mining occurs in certain areas where particularly important wildlife resources are located (Honig et al. 1981). Contour mining often results in increased recreational activities (e.g., hunting, motorcycling) in the mountains, because roads to such mines allow increased access. Although contour mining usually does not occur within parks, mines within the viewshed of mountain parks can have aesthetic impacts on park visitors.

Large area strip mining is conducted in relatively level or gently sloping terrain, both in the eastern and western United States. The topography is altered, and former land uses (forest, crops, pasture, rangeland) are displaced until the mines are reclaimed (Haynes et al. 1979). Land at the mines is also replaced by buildings, railroad spurs, and access roads. Soils on reclaimed mines are not usually very productive, and 10 to 30 years may pass before row-crop agriculture can resume in the eastern United States (Dvorak et al. 1977).

In the Southwest, lands that are surface mined are less productive than lands in the eastern United States and in some areas rangelands are less productive because of overgrazing. Because of low rainfall, it can be very difficult to establish vegetative cover and reclaim a mine without irrigation (Dvorak et al. 1977).

2.2.2 Coal Cleaning and Beneficiation

Coal cleaning, with its attendant generation of refuse, is done primarily in the central interior and eastern areas of the United States. The primary impact on terrestrial resources is the displacement of land by refuse disposal areas (gob piles and slurry lagoons). Terrestrial environments near such disposal areas can be impacted by windblown coal fines and acid drainage from the refuse. Proper reclamation of the disposal areas can minimize impacts, producing land suitable for wildlife cover, pasture, or recreation.

2.2.3 Coal Transportation

Sources of transportation impacts on terrestrial resources include the following:

- land requirements for new railroad spurs and haul roads to the mine sites,
- land requirements for new loading and unloading facilities for barges,
- land requirements for slurry pipelines to transport coal from western and midwestern states to eastern markets, and
- accidental spillage and windblown dusts of coal associated with moving coal cars and coal trucks.

Terrestrial resource impacts of coal transportation, aside from impacts of transportation emissions, will be minimal if existing transportation facilities are adequate and little construction of new transportation facilities is required. Most disturbance will occur in the vicinity of new mines requiring the construction of roads and railroads from the mine site to existing roads and railroads.

2.2.4 Generation

The generation stage will impact terrestrial resources through the replacement of land by generating facilities and the release of gaseous emissions (Dvorak et al. 1977, 1978; U.S. DOE 1989). At or near the generation site, land will be occupied by the plant facility itself and by various support facilities, including electrical substations, coal storage yards, ash and slag disposal areas, and disposal areas for limestone scrubber sludge. Cropland is permanently lost when replaced by such facilities, which are often located on highly productive cropland near rivers.

Air pollution can damage plant tissue and cause decreases in production of crops and native vegetation. The extensive literature on the effects of air pollutants on plants has been summarized by Shriner et al. (1990). Gaseous and particulate emissions can also decrease visibility over vast areas. Aesthetic quality at parks in the Southeast and the Southwest has been adversely affected by pollution-caused decreases in visibility.

2.2.5 Power Transmission

New power lines may be required for new generating facilities. Impacts of power lines are relatively minimal (Kroodsma and Van Dyke 1985; Lee et al. 1989). The greatest impact occurs when a new line is routed through forest, which must be cleared to accommodate the new line. Because power-line corridors through forests are narrow and usually support dense, brushy vegetation, impacts on wildlife and forest land uses are minimal. In open areas such as pasture and cropland, existing land uses can continue beneath the new lines except in spots occupied by support towers. The towers occupy relatively little land and have minimal economic impact on agriculture. Extensive studies of electromagnetic fields produced by power lines have not shown significant impact on human health, wildlife, or livestock. Where power lines cross residential areas, property values may be reduced.

3. DEFINITIONS OF IMPACT CATEGORIES

This section defines the ecological impact categories to be used in the accounting framework (i.e., the column headings in the matrices mapping emissions into impacts). The categories are defined in terms of the biological resources valued by society, rather than by medium or path. Using this scheme, soil and water quality are intermediate impacts rather than resource categories, because they affect the suitability of the environment for valued organisms or for other human uses. A particular resource such as agriculture can be affected by multiple emissions and by multiple environmental pathways (e.g., both through direct effects of air pollutants on plants and on indirect effects of degraded soil quality). The impact categories are defined below and summarized in Table 1.

3.1 CROPS AND SUBURBAN LANDSCAPE

Crops include both conventional row crops such as corn, soybeans, and orchards, and other plants harvested for direct economic value. Crops can be affected by direct phytotoxicity (toxic effects of contaminants on plant tissue), by soil degradation due to contaminant deposition or disturbance, or by irrigation with contaminated water. Common crops in the southeastern United States include corn, soybeans, and cotton. Although the acreage is smaller, similar crops are grown in the southwest on irrigated land.

The extensive literature on the effects of air pollutants on plants has been summarized by Shriner et al. (1990). The impact of specific pollutants decreases in the following order: ozone (O_3) > sulfur dioxide (SO_2) > acidic deposition > nitrogen dioxide (NO_2). Reductions in production of various crops is decreased up to 56% by O_3, depending on the crop species, location, and exposure. No effects on regional crop yield have been shown to result from the other pollutants. Reductions of crop production may rarely or occasionally be caused by SO_2 in the vicinity of SO_2 point sources.

Available data are generally adequate to quantify the impact of a power plant's emissions on crop production, provided that one can quantify the ozone concentrations for which the power plant is responsible. Shriner et al. (1990) conclude that empirical models are currently adequate for assessment of air pollution impacts on crop yield and provide a case-study assessment. The acreage and production of each type of crop in each appropriate region can be determined from data available in U.S. Department of Agriculture (USDA) and State reports on agricultural statistics. The quantitative relationships between crop production and ozone concentration can be obtained from existing literature for a large number of crop species and cultivars (Heck et al. 1986; Shriner et al. 1990). Then, the acreage and dose-response data can yield estimates of each crop's production losses resulting from the power plant. Impacts of other coal emission sources and stages of the fuel cycle other than power generation would only be minor but could be quantified provided that their respective air pollutant concentrations could be quantified.

Table 1. Summary of impact categories

Impact category	Impact—pathways	Definition
Crops and suburban landscape	contaminant deposition on plant surfaces or uptake by plants (phytotoxicity)	loss of crop yield (orchards, row crops, nurseries)
	soil contamination (phytotoxicity or nutrient leaching)	
	disturbed or lost acreage	
	contaminated irrigation water	
Livestock	contaminant deposition on plant surfaces	loss of livestock productivity due to reduced production or availability of pasture
	soil contamination	
	lost acreage	
Timber	contaminant deposition on plant surfaces or uptake by plants (phytotoxicity)	loss of timber yield due to reduced tree growth or acreage loss
	soil contamination (phytotoxicity or nutrient leaching)	
	lost acreage	
Commercial fishing	water quality degradation	loss of commercially harvestable fish due to reduced production or contamination above regulatory standards

Table 1. (continued)

Impact category	Impact-pathways	Definition
Recreational fishing	water quality degradation flow reduction habitat loss	loss of opportunity or quality of recreational fishing due to reduced production or contamination above regulatory standards
Hunting	habitat/landscape destruction or disturbance	loss of opportunity to hunt
Recreational use of forests or parks	habitat/landscape destruction or disturbance reduced visibility impaired air/water quality	loss of opportunity for touring, hiking, birdwatching, swimming, and other nonconsumptive uses
Biodiversity	impaired air quality water quality soil quality habitat destruction or disturbance	impacts on threatened and endangered species; impacts on other aesthetically valued plants or wildlife; altered community structure/function
Buildings and materials	deposition of particles, aerosols, and contaminated rainwater	enhanced weathering of exposed metal and stone
Nonecological land use	land disturbance or contamination	depression of land value
Nonecological water use	impaired water quality or quantity	cost of treatment for other uses
Visibility	dust and haze	visual range; plume blight; visual insult reduced visibility

3.2 LIVESTOCK

Livestock includes animals and poultry raised for meat or dairy products. Animals and poultry could theoretically be directly impacted whenever they inhale air pollutants or consume vegetation contaminated by air pollutants. Ambient air pollution levels in rural areas are, however, usually far below levels that could cause significant decreases in livestock productivity. No data demonstrating such direct impacts are available. Cattle and poultry are the principal livestock raised in the southeast. Cattle and sheep grazed on open range are predominant in the southwest.

3.3 TIMBER

Timber refers to wood products harvested from managed natural forests or tree plantations. Irrigation of forest land is rare in the United States. Otherwise, timber production is subject to the same sources of environmental impacts as is crop production. Extensive pine plantations grown for pulp production are present in the southeast. National Forests in the southern Appalachians provide some hardwood production. Commercial timber harvest is negligible in the southwest.

In contrast to the availability of data for crop production, data are generally unavailable to show the responses of whole trees or stands of trees (e.g., mixed hardwood forests or pine plantations) to air pollutant stress. Consequently, empirical models and conclusive quantitative estimates of such responses do not exist. Existing process models relate to responses of tree seedlings and branches to air pollutants. These models are being modified to provide preliminary estimates of whole tree response, which could then be used to extrapolate to stand response. This work has not been completed. The review by Shriner et al. (1990) provides no estimate of stand response.

3.4 COMMERCIAL FISHING

Commercial fishing refers to harvest of fish and shellfish for sale. Fishery value can be affected by (1) changes in water quality that affect the survival, growth, or reproduction of the organisms, or (2) accumulation of contaminants in the harvested product. The term "water quality" encompasses a variety of characteristics that influence the suitability of a water body for use by man or by the resident biota. Among these are temperature, pH, dissolved materials such as plant nutrients (phosphates and nitrates), degradable organic pollutants, metals and other toxic chemicals, oil, grease, and suspended sediment. Commercial fishing is not a major industry in coal-bearing regions of the United States. A small mussel industry (primarily for pearl production) still exists in the southeast, and aquaculture for trout and catfish production is common in this region. Commercial harvest of finfish from southeastern rivers and reservoirs is negligible. Commercial fishing is negligible in the southwest. Major commercial fisheries in the Great Lakes or in coastal waters could be affected by CO_2-related climate change and long-range transport and deposition of atmospheric pollutants.

Effluents from coal mining, beneficiation, and combustion all have the potential to adversely impact water quality and, in turn, aquatic communities. Water quality impacts from coal mining primarily result from soil erosion (which increases turbidity and sedimentation in surface waters) and contaminated mine drainage (sulfur, heavy metals, and organic compounds). These potential impacts could also occur at beneficiation and generation sites but are more localized and thus more easily controlled. The amount of sediment and contaminated drainage produced per hectare of mined land can be estimated for each coal mining region (DOE 1981), but the consequent biological effects are more difficult to quantify, especially on a regional basis.

Solid wastes (coal fines, noncoal mineral refuse, ash, and slag) can enter surface waters as suspended solids and impact both water quality and aquatic biota. In addition to the physical effects of turbidity and sedimentation, leaching of dissolved contaminants (toxic organic compounds and heavy metals) from these solid wastes could degrade surface waters. Volumes and chemical characteristics of leachates from solid wastes piles have been estimated (e.g., DOE 1981).

Fish and wildlife resources having significant problems in specific geographic areas have been identified and mapped (e.g., Honig et al. 1981) so that the coincidence of the distribution of coal fuel cycle impacts with the distribution of threatened or endangered species can be studied. Alternatively, the measured biological effects of estimated levels of pollutants (e.g., parts per million (ppm) of suspended solids, concentrations of metals, or particular pH values) can be coupled with information about biological resources (e.g., abundance or yield to the fishery) to estimate a reduction in the resource. Because biological populations are highly variable from site to site, this approach would best be applied to either a specific site (for which the species and numbers are known) or a hypothetical case study (in which reasonable values that reflect regional resources are assumed).

Acidification of surface waters as a result of acid precipitation could also limit diversity and production of aquatic communities. Although lake acidification experiments indicate that biotic effects of acid rain can be severe, causal linkages between acidic deposition, acidification of surface waters, and the loss of biological resources are poorly quantified on a national or continental scale. As a result, projections of potential impacts to aquatic resources are difficult (DOE 1989).

Atmospheric deposition has recently been identified as the principal source of polychlorinated bipheny, dioxin, and heavy metal pollution in many water bodies. No quantitative estimates of biological impacts are available at this time.

3.5 RECREATIONAL FISHING

Recreational fishing refers to harvest of fish for sport or for consumption by the fisherman. Recreational fishing is affected by the same sources of impact as commercial fishing. Recreational fishing is common in ponds, rivers, and reservoirs of the southeast. Dozens of species are fished. The most important recreational fisheries in warmwater reservoirs, rivers, and ponds involve the families Centrarchidae (largemouth and smallmouth bass, bluegills, crappie), Ictaluridae (catfishes), and Percidae (perches, walleye, sauger). Striped bass fisheries exist in many large reservoirs. Coldwater streams in the southern Appalachians and on the Cumberland Plateau support fisheries for rainbow trout, brown trout, and brook trout. Recreational fishing is less common in the southwest; however, tailwaters below the major dams on the Colorado River support significant trout fisheries.

Recreational fisheries are subject to the same sources of damage as commercial fisheries. Compared to commercial fisheries, recreational fisheries are more likely to be located in close proximity to coal fuel cycle activities. They are therefore more likely to be affected by runoff and aqueous effluents than are commercial fisheries. Exposures to coal-derived contaminants are likely to be more readily estimated. However, because recreational fish populations are generally less intensively studied than are commercial stocks, biological damage may be more difficult to quantify.

3.6 HUNTING

Hunting refers to noncommercial harvesting of game birds and mammals. These animals can be affected by air and water pollution and by physical disturbances (habitat destruction, noise) related to energy production. Hunting is common on private and public land throughout the southeast and southwest.

Reduced wildlife populations clearly result in reduced hunter success but may not cause significantly reduced hunting activity. Wildlife populations in hunting areas may be reduced by surface mining and may remain at reduced levels until reclamation is fully effective in restoring original conditions. Little information is available for quantifying the reduction of wildlife populations or hunting activity. Air pollution resulting from the coal fuel cycle is not likely to cause direct toxic effects on wildlife.

3.7 RECREATION

Recreational use of forests and parks refers to nonconsumptive activities such as touring, boating, swimming, hiking, camping, and trail-biking. Recreation can be affected by adverse changes in forest composition or wildlife abundance, by reduced visibility, or by noise and visual impacts of mining or electricity generation. Changes in water quality related to energy production or other industrial facilities can affect the recreational value of rivers and lakes. All rivers and reservoirs in the southeast support intensive recreational use. National forests and the Great Smoky Mountains National Park are also important recreational resources. The southwest contains several important National Parks and monuments. These are especially vulnerable to reduced visibility due to haze formation.

3.8 BIODIVERSITY

The term "biodiversity" has been defined many ways by many different people. A variety of aspects of biodiversity and threats to biodiversity were recently discussed in the proceedings of the National Forum on Biodiversity (Wilson 1988). We define impacts on biodiversity to include all ecological effects not directly related to exploitation or recreational use. Specific impacts addressed in this project include (1) impacts on threatened or endangered species or on legally protected systems (e.g., Wild and Scenic Rivers), and (2) impacts on other aesthetically valued natural ecosystems (e.g., wetlands, pine barrens, bogs) of the types that are protected by organizations such as the Nature Conservancy. Although heavily modified by man's activities, the southeastern United States supports a number of endangered and threatened species as well as relict examples of a number of previously common ecosystem types. The southwest has also been heavily modified by man. In this region, moist riparian habitats are especially important reservoirs of biodiversity.

To some degree, all emissions from all components of the coal fuel cycle can affect biodiversity. Because biodiversity (as used here) reflects nonuse aspects of the environment, this is one category for which physical habitat destruction can lead to external costs - although estimating nonuse value is an extremely controversial issue.

4. QUANTIFICATION METHODS

In general, there are three types of approaches to deriving quantitative relationships between levels of environmental stress and ecological responses:

(1) empirical modeling, (2) mechanistic (or process) modeling, and (3) expert judgement. The appropriate choice for a given circumstance depends on the quantity of data available, the degree of understanding of underlying processes, the similarity of the new situation to previous situations, and the amount of extrapolation required beyond existing conditions.

Empirical relationships are derived through statistical analysis of measured data. In ecology, statistical models (including regression analysis, discriminant analysis, and other multivariate techniques) are often used to summarize and interpret field observations (Green 1980). In environmental impact assessment, statistical models are fundamental components of the design of monitoring programs to detect differences (1) between control and affected sites (Thomas 1977, Loehle and Smith 1990) and (2) between pre-impact and post-impact observations (Vaughan and Van Winkle 1982, Loehle and Smith 1990). In assessments of impacts of acidification on surface waters, empirical models have been used extensively to extrapolate from observations on subsamples of lakes and streams to impacts on regional surface water resources (Baker et al. 1990). Empirical models have also been used to estimate reductions in regional crop yield due to O_3, SO_2, and NO_2 (Shriner et al. 1990).

Empirical models do not explain observations in terms of causal relationships between the independent and dependent variables. They simply summarize observed relationships between variables. Such models can be used for predictive purposes (e.g., predicting responses of crop production to air pollution), provided that (1) independent evidence of causal relationships is established, and (2) the range of expected stress and response levels lies within (or at least close to) the range over which the observations used to develop the model were made. These conditions were met in both of the above examples. In many important assessment problems, however, empirical modeling is inadequate or unfeasible. In assessments of the ecological effects of global climate change, for example, the major interest is in predicting long-term responses to climatic regimes well outside of the historically observed range. Long-term management of fisheries (Walters 1986) and prediction of environmental responses to PCB remediation (Limburg 1986) are similarly inappropriate for empirical modeling.

As an alternative, dose-response relationships may be estimated using mechanistic models derived from first principles. The purpose of a mechanistic model is to describe in quantitative terms the relationship between some phenomenon and its underlying causes. Whereas in a statistical model the fitted coefficients such as the slope and intercept of a regression line have no intrinsic meaning, the parameters in a mechanistic model have real operational definitions and are (at least in principle) amenable to independent measurement. The classical laws of physics (such as Newton's Laws and Maxwell's equations) are mechanistic models. Contaminant transport models used to predict deposition patterns or concentrations of contaminants in terms of the physical and chemical processes are also mechanistic.

Mechanistic models can be generally categorized as (1) steady-state models that predict a final endpoint (e.g., surface water chemistry) given certain inputs (e.g., atmospheric sulfur deposition) and known geochemical relationships, without regard to the time it takes for the changes to occur, and (2) dynamic models that take into account factors such as rates of change that indicate how long it will take to reach an endpoint or what pathway it follows to get there. Dynamic models provide more information concerning responses to stress, but predictions about steady states are usually more reliable. Examples of mechanistic models used in acidic deposition assessment of aquatic effects include the MAGIC and ILWAS models and others described by Thornton et al. (1990). Kiester et al. (1990) discuss mechanistic models used for predicting effects of air pollutants on trees and crops. Walters (1986) has discussed the use of mechanistic models of fish population dynamics in fishery management.

If the mechanisms are correctly represented, and the parameters have been adequately measured, then predictions should be possible outside as well as within the range of observations. However, the degree of validity of mechanistic models is often unknown. Empirical approximations and simplifying assumptions are always required, and experimental or observational testing of assumptions and predictions is often impossible. For this reason, there is no clear-cut rule for selecting between empirical and mechanistic approaches to prediction. If data are limited or large extrapolations are required, but causal mechanisms are reasonably understood, mechanistic modeling may be preferable. If a large amount of relevant data exist, and extrapolation beyond the range of observations is not required, then empirical approaches may be superior. If both large data sets and mechanistic understanding exist, then mechanistic models may again be preferable.

In many cases, data and understanding are insufficient to support either empirical or mechanistic prediction. For these situations, subjective stress-response relationships derived from expert judgement provide a third means of characterizing stressor-response relationships. For example, existing data are insufficient for developing rigorous statistical estimates of the decline in forest productivity per unit ozone or sulfur released. However, experts in plant physiological ecology could probably use the large data base for crops, together with general understanding of the differences between crop plant and tree physiology, to develop simple coefficients for use in impact assessments.

All three approaches are required to assess the ecological consequences of alternative fuel cycle technologies. Reasonably well understood effects (such as effects of air pollutants on crops and effects of acid deposition on water quality and fish community structure) can be partially quantified using a mix of empirical models, mechanistic models, and expert judgement. Highly site-specific effects such as sedimentation and acid drainage from mines can be quantified from site-specific data, but generic, predictive models do not exist. Poorly documented effects (such as changes in biodiversity or recreational opportunity) and poorly understood sources (such as atmospheric mercury) cannot be quantified using any of these approaches.

5. EVALUATIONS OF KEY IMPACT CATEGORIES

5.1 EFFECTS OF ACIDIC DEPOSITION ON RECREATIONAL FISHERIES

5.1.1 Background

The principal source of quantitative information on effects of acidic deposition on recreational fishing is the National Acid Precipitation Assessment Program Integrated Assessment (NAPAP 1990) and its associated State of Science/Technology reports (e.g., Baker et al. 1990, Turner et al. 1990, J. Baker et al. 1990, and Thornton et al. 1990). These reports summarize the surveys, models, data sets, and conclusions about relationships between acidic deposition and effects on aquatic biota from the 10-year NAPAP study. The primary emphasis of the aquatic effects assessment is on numeric models applied to regions. The regions for which projections of long-term or chronic aquatic effects were made by NAPAP were a subset of the regions surveyed as part of the NAPAP National Surface Water Survey (NSWS) (Fig. 1). Two steps were employed in the modeling process: (1) modeling of watershed chemistry to relate deposition scenarios to projected long-term chemical characteristics of the surface water, and (2) modeling of fish responses to changes in pH and other water quality parameters.

Several different approaches to both modeling steps were developed as part of NAPAP (Thornton et al. 1990, J. Baker et al. 1990). Long-term regional water chemistry projections ultimately were based principally on the MAGIC model. The principal biotic response model employed was an empirical model derived from observed associations between fish population status and acid-base chemistry in field studies. The output of the combined models consists of region-specific estimates of the fraction of streams or lakes with long-term acid-base chemistry suitable for fish survival under different scenarios of future sulfur deposition.

The percent of affected systems provides an objective reference value for comparing relative differences between deposition scenarios. Chemical changes

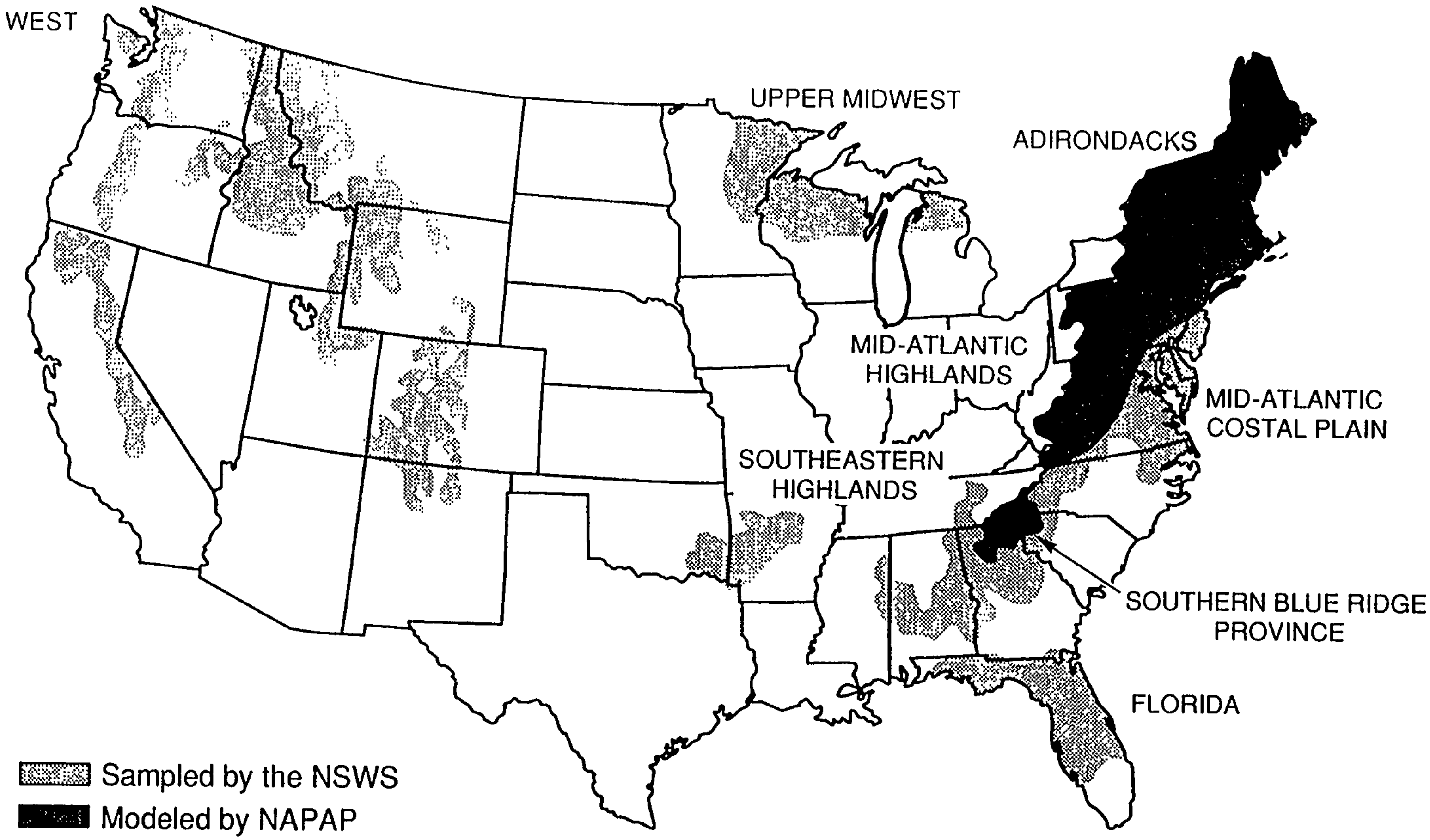

Fig. 1. Locations of regions and subregions considered in the National Surface Water Survey (NSWS). National Acid Precipitation Assessment Program (NAPAP) model analyses were conducted for the Adirondacks, New England, Mid-Atlantic Highlands, and Southern Blue Ridge Province. *Source*: NAPAP (1990).

in surface waters are summarized as percent of systems with chemistry above or below certain thresholds over a 50-year period. Biological effects are projected as presence or absence of fish because intermediate responses (i.e., reductions in abundance short of extinction) cannot be modeled and unequivocally attributed to acidic deposition. Responses differby region because of differences in both watershed chemistry and fish sensitivity. Brook trout, for example, are relatively tolerant to low pH compared to many other fish, and watersheds at high altitudes are typically more sensitive to acidic deposition than are those at low altitudes.

NAPAP developed dose-response relationships for two measures of fish response: "sensitive fish species" and "brook trout habitat." The sensitive fish species index, for fish such as rainbow trout and the common shiner, is based on laboratory test data that relate rainbow trout survival to pH, aluminum, and calcium concentrations. The brook trout habitat measure is based on field studies in which empirical models are calibrated from observed associations between fish population status (i.e., presence or absence) and observed acid-base chemistry. The sensitive-species model is applied to all streams in a region; the brook trout habitat model is applicable only to streams known to be otherwise suitable for brook trout.

5.1.2 Southeastern Reference Site

Most of the streams and reservoirs within a 50-km radius of the reference plant site (Fig. 2) lie in the Ridge and Valley physiographic province at relatively low elevations, are well-buffered by carbonate bedrock, and are insensitive to the effects of acidic deposition. Many small streams draining ridges in this region originate in highly weathered soils with little buffering capacity and, during storms, show pulses of acidic runoff (Elwood and Turner 1989, Mulholland et al. 1990). These acidic pulses may have adverse effects on benthic organisms inhabiting these streams, but the streams are neutralized before they are large enough to support fish suitable for a recreational fishery.

A small section of the Cumberland Plateau lies within the 50-km radius northwest of the reference plant site (Fig. 2). Streams in other portions of the Cumberland Plateau have been shown to have low acid neutralizing capacity (ANC) and high potential sensitivity to effects of acidic deposition (Kaufmann et al. 1988). Watersheds in the Cumberland Plateau were not modeled by NAPAP, but they are similar in geology, soils, and sulfur deposition to those in the more-northern Mid-Atlantic Highlands region that were modeled. Significant changes in chemistry and suitability for fish were projected for that region (see discussion below). The Cumberland watersheds could be expected to respond similarly over time. Streams in this area of the Cumberland Plateau are used for recreational fishing. Information on wet and dry atmospheric deposition, runoff,

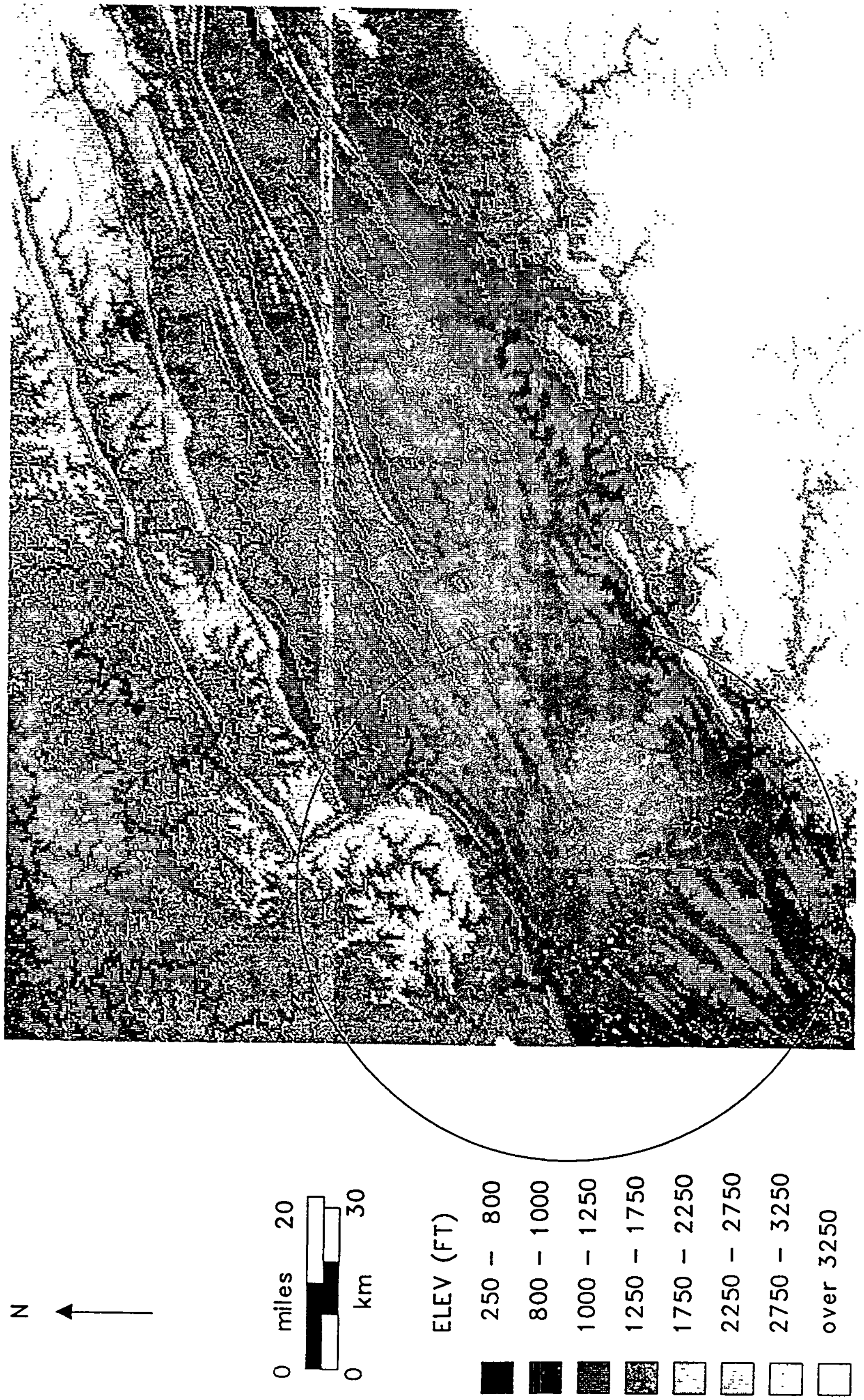

Fig. 2. Digital elevation map of the area surrounding the Southeast Reference plant site. The circle encloses area within a 50-km radius.

current water chemistry, soils, and fish population status for a sample of these watersheds would be needed to project effects of the reference plant on future conditions for these streams.

Long-range transport of acid sulfates and nitrates could increase acidic deposition on sensitive, high-elevation streams beyond the 50-km distance from the plant. The NAPAP regions of interest for the fuel cycle study are the NAPAP Southern Blue Ridge Province, which contains an estimated 2,021 stream reaches and approximately 2,300 km of streams, and the Mid-Atlantic Highlands, which contains about 28,000 stream reaches and more than 78,000 km of stream length (Fig. 1). No streams in the Southern Blue Ridge Province were found to be acidic in the NSWS, and no streams are projected to become acidic within the 50-year time frame used by NAPAP, though the projected trend in ANC is downward. Estimates of fish response were not made in the Southern Blue Ridge because (1) field information for developing the models is lacking, and (2) models developed elsewhere were judged not suitable for extrapolation (J. Baker et al. 1990).

Figure 3 presents dose-response information for the two fish-effects indices for the Mid-Atlantic Highlands region. If acidic deposition (kg sulfur/ha/year) in the Mid-Atlantic Highlands (caused by the reference plant) can be estimated, then Fig. 3 can be used to quantify the change in suitability of Mid-Atlantic Highland streams for fish. The projections in Fig. 3 are based on 50-year simulations to account for these streams not being currently at steady-state with the existing deposition level. Hence, even at no change in acidic deposition, the percent of streams unsuitable for brook trout because of acidification is projected to increase from 8% at present to about 20% after 50 years. Because the responses are nonlinear, assessing the incremental effect of the reference plant requires making an assumption about the expected future sulfur deposition patterns. Figure 3 summarizes results of six sulfur deposition scenarios reflecting, in increasing order, a 50% reduction, a 30% reduction, a 20% reduction, no change, a 20% increase, and a 30% increase. The percentage of systems projected at the end of 50 years to have certain water quality conditions (such as pH<6 or ANC<0) can also be shown in this manner (Fig. 4). Similar data for calcium and aluminum concentrations are shown in NAPAP (1990). The procedure for estimating fuel-cycle impacts is to select a baseline future deposition scenario, add the increment from the plant, and then interpolate between the adjacent points on the graph.

These graphs show regional response of lake and stream populations. The statistical framework of the NSWS was not designed to evaluate the geographic distribution of affected watersheds within regions. Generally, in the Mid-Atlantic Highlands, the watersheds projected to be affected by acidic deposition are small, forested watersheds in the higher elevations (NAPAP 1990). These watersheds occur geographically throughout the region, following the physiography of the region.

Figures 3 and 4 show aquatic conditions at the end of 50-year projections. Simulated changes can also be shown over time, so that ecological effects or economic costs can be calculated at different points in time, or cumulatively. Figure 5 shows projections for the percentage of systems with ANC<0 and the percentage of systems with chemistry unsuitable for sensitive fish species for the Mid-Atlantic Highlands region under two NAPAP sulfur deposition scenarios. S1 is a scenario of no new legislated controls (developed prior to the 1990 Clean Air Act Amendments). S4 is a scenario of emissions/deposition that may be similar to what will occur under the Clean Air Act Amendments as passed (about a 50% reduction in sulfur emissions by 2000). To estimate fuel cycle impacts of the reference plant, one would add the increment of sulfur deposition attributable to the reference plant to each of the modeled watersheds, rerun the models under the new deposition loads, and then recalculate the projected regional effects. This might result in lines very slightly higher than those shown in Fig. 5.

The increment of sulfur deposition at each watershed attributable to the reference plant could be calculated as NAPAP did using the Regional Acid Deposition Model (RADM) (Dennis et al. 1990, Clark et al. 1989). There is a great deal of uncertainty in estimating wet and dry deposition to watersheds, both for current deposition (Turner et al. 1990) and for future deposition (Dennis et al. 1990). Figure 6 shows a potential range of SO_2 emissions for the United States from 1990 to 2030, depicting the uncertainty around the NAPAP S1 scenario. Similar uncertainty would exist around any other emissions scenario. Because the aquatic effects of sulfur deposition are not linear, the incremental effect of the reference plant could be quite different under higher versus under lower sulfur emissions/deposition rates. The incremental effects could also change over time. For example, they could be higher when the regional sulfur deposition loading (i.e., from other power plants) was high, and lower or nonexistent below a certain threshold or critical load of sulfur deposition.

To conclude, numbers for regional long-term or chronic effects of sulfur deposition attributable to the reference power plant on aquatic chemistry and suitability for fish could be calculated with available models. Uncertainty in the estimates of sulfur deposition used to drive the models is high (Dennis et al. 1990, NAPAP 1990), as is uncertainty in the watershed chemistry and biological models themselves (Thornton et al. 1990, J. Baker et al. 1990, NAPAP 1990). The models are considered useful for making general regional comparisons between sulfur deposition scenarios over time. They are not considered useful for quantifying what conditions will actually be in future years because they do not include effects of many watershed factors other than long-term sulfur deposition (Turner et al. 1990, NAPAP 1990). To more accurately quantify the incremental

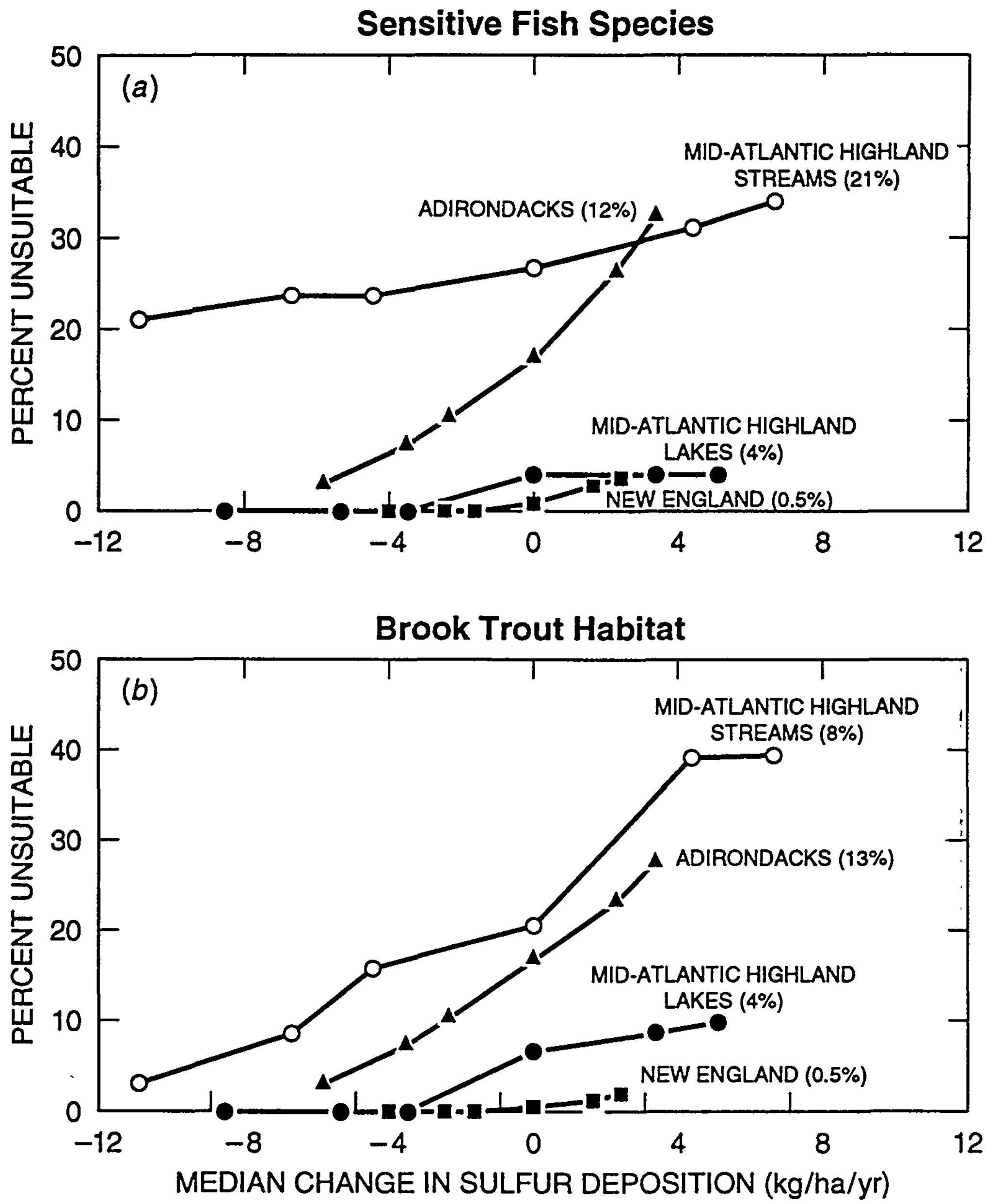

Fig. 3. Effects on fish habitat in each subregion, based on projections from MAGIC after 50 years versus median change in sulfur deposition for each deposition scenario (-50%, -30%, -20%, 0%, +20%, +30%). (a) Percentage of the modeled target population unsuitable for sensitive fish species due to acid-base chemistry. (b) Percentage of waters potentially supporting brook trout that are projected to become unsuitable for brook trout because of changes in acid-base chemistry. Values in parentheses after each region indicate the simulated condition at the start of the 50-year projection. *Source*: NAPAP (1990).

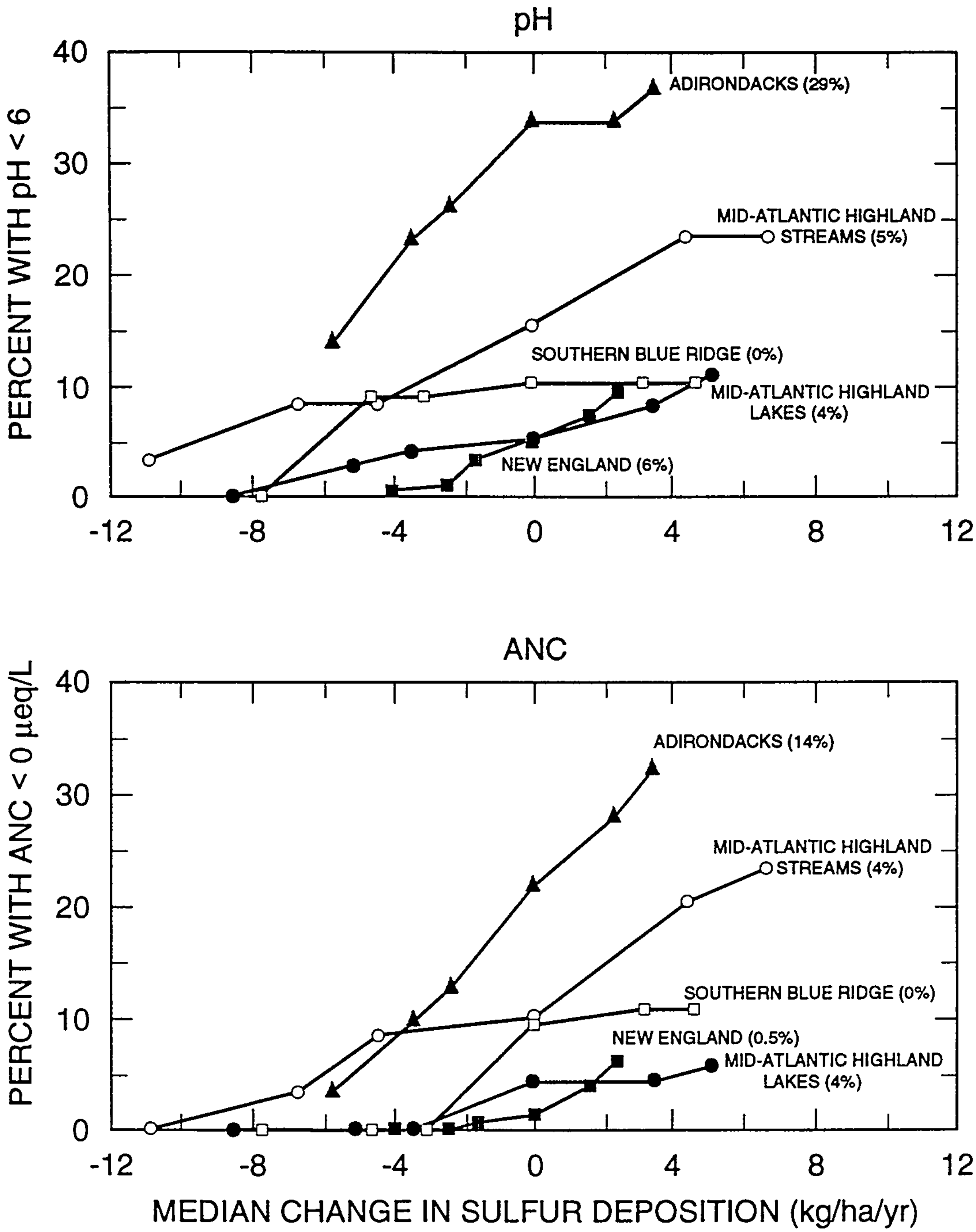

Fig. 4. Percentage of the modeled target population in each subregion with (a) pH<6 and (b) acid neutralizing capacity (ANC)<0 μeq/L after 50 years, based on MAGIC projections, versus median change in sulfur deposition for each deposition scenario (-50%, -30%, -20%, 0%, +20%, +30%) and subregion. Values in parentheses after the subregion name indicate the simulated condition at the start of the 50-year projection. *Source*: NAPAP (1990).

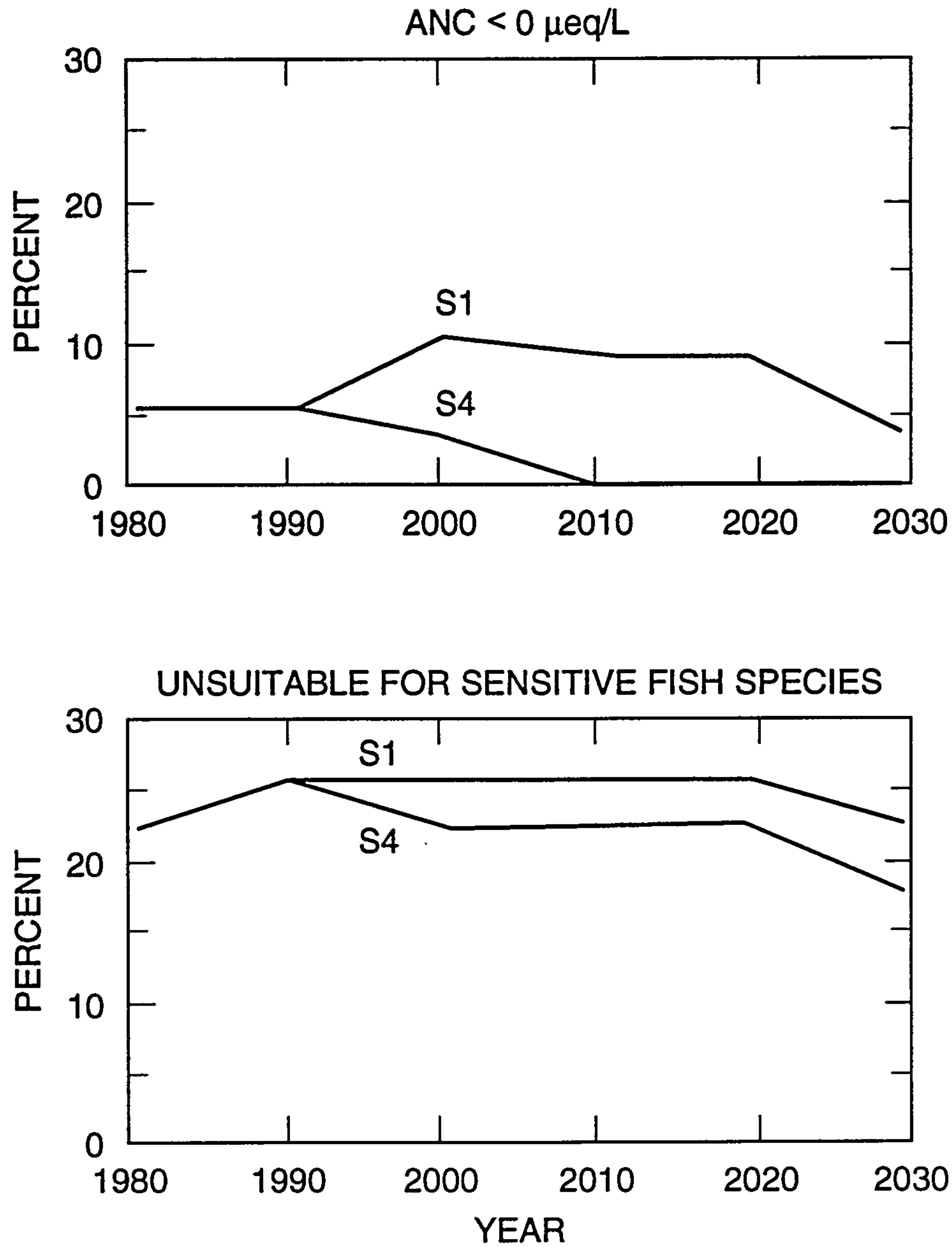

Fig. 5. Percentage of Mid-Atlantic Highland streams having (a) acid neutralizing capacity (ANC)<0 μeq/L and (b) chemistry unsuitable for sensitive fish species versus time for two NAPAP sulfur deposition scenarios (see text for discussion). *Source*: adapted from NAPAP (1990).

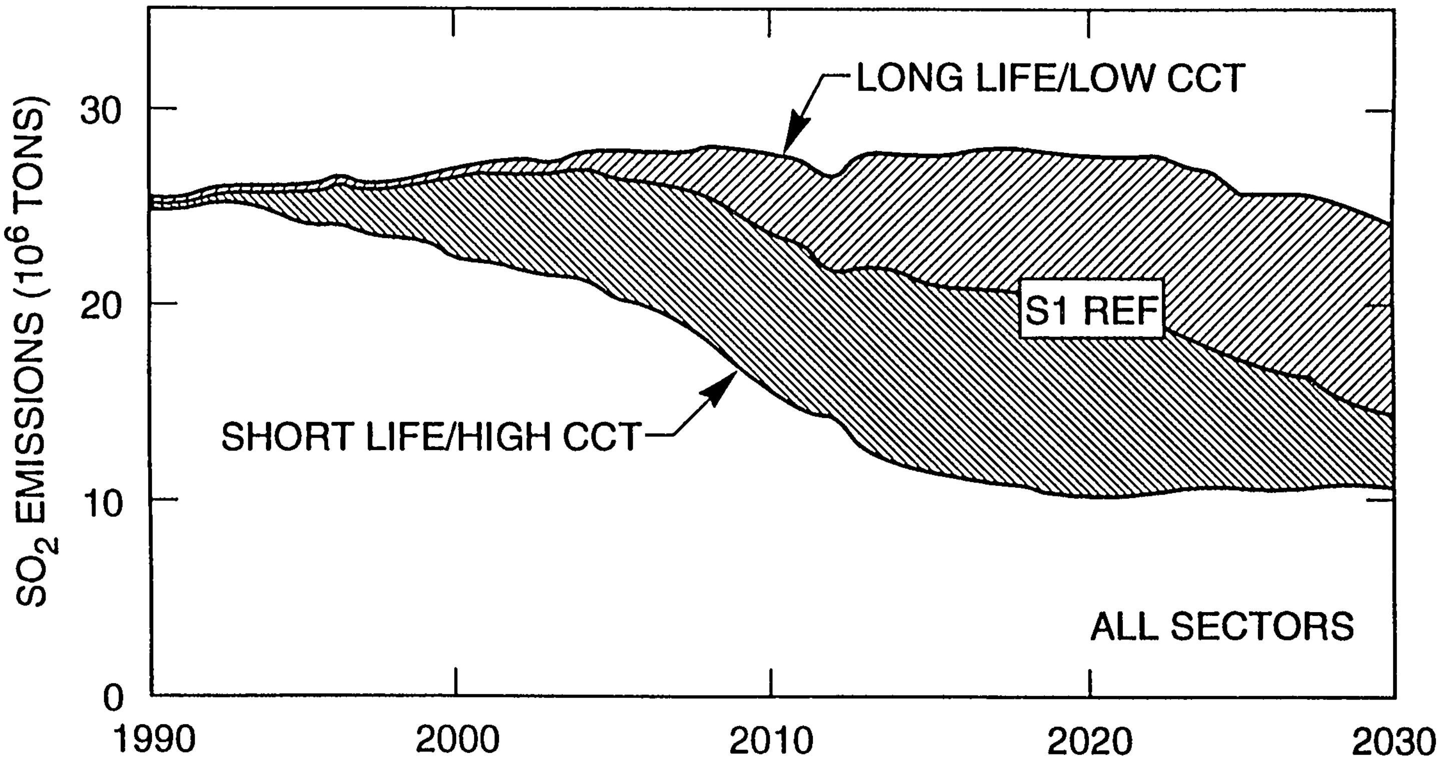

Fig. 6. A potential range of United States sulfur dioxide (SO_2) emissions in the absence of new acidic deposition control legislation (i.e., pre-1990 Clean Air Act Amendments). Upper and lower bounds result from combining alternative assumptions on power plant life times and rates of market penetration of Clean Coal Technology (CCT) at existing plants. *Source*: NAPAP (1990).

effects of a single power plant, additional research is needed to (1) reduce uncertainty in projections of future regional atmospheric deposition (or to hypothesize specific scenarios for evaluation), (2) reduce uncertainty in estimation of wet and dry atmospheric deposition (of acidifying and neutralizing substances) to individual watersheds, (3) improve our ability to model all important watershed processes that affect water chemistry and fish response on both long-term or chronic and short-term or episodic time scales (and to survey all the input data for the watersheds needed to drive the models), and (4) improve our models of fish response to short- and long-term changes in water chemistry.

5.1.3 Sample Calculation

Figure 3 shows that if there is no change in current sulfur deposition, 20% of Mid-Atlantic Highland streams are projected to be unsuitable for brook trout in 50 years; if deposition increases by 4 kg/ha, 40% of these streams are projected to be unsuitable. If we assume that there will be no changes in future deposition except for an average deposition of 0.01 kg/ha due to the reference plant, then the incremental impact of that plant would be [(0.01/4) x 20 =] 0.05% increase in the number of streams unsuitable for brook trout. This translates into the loss of 14 of the 28,000 stream reaches in the region.

...the incremental impact of that plant would be [(0.01/4) x 20 =] 0.05% increase in the number of streams unsuitable for brook trout.

5.1.4 Southwestern Reference Site

Rivers draining this region are well buffered by geologic processes and are not likely to be acidified by an additional power plant in the region. Landers et al. (1987) found in the Western Lake Survey (part of the NSWS) that there were numerous lakes in the high-elevation mountain regions of the West that have low ANC and are potentially highly sensitive to effects of acidic deposition, though currently no lakes are acidic. NAPAP did not model future effects in the West, because no regional effects have been documented to date and because uncertainty in current and projected wet and dry deposition to these lakes is very high. Watershed models have been calibrated and run for several western lakes (Thornton et al. 1990, Turner et al. 1990). They could be used to estimate effects of the Southwest Reference plant on some high elevation systems, but problems with uncertainty would be greater than those for the Southeast site.

5.2 EFFECTS OF INCREMENTALLY INCREASED SURFACE MINING ON AQUATIC RESOURCES

5.2.1 Background

Surface mining of coal is known to have adverse impacts on surface waters and aquatic resources. The environmental impacts of a single mine site are greatest immediately downstream of the site and generally decrease with distance from the site. Because coal mining operations and equipment choices are extremely varied and, in general, are determined by the local geology and other natural conditions, no single mining procedure is representative of the entire industry. The exact environmental effects will, therefore, depend upon the chosen mining technique, the prevailing geological and geochemical characteristics, and the success of the reclamation efforts.

5.2.1.1 Objective

The objective of this assessment is to determine the feasibility of estimating the contribution of a single reference unit to regional ecological impacts, even though only the cumulative impacts of many sites may be directly measurable at the regional scale. The sources of external costs addressed here are effluents from the mine site that may adversely affect the aquatic environment. A background perspective of potential modes of surface mining impacts on the aquatic environment is provided. This analysis uses a comparable adjacent watershed approach to estimate impacts associated with mining and applies this approach to a hypothetical mine site in the eastern coal region. Finally, it discusses the limitations to the application of this approach based on existing information, to quantification of the impacts of mining on aquatic systems in general, and to estimation of impacts to recreational resources.

5.2.1.2 Literature Background

Surface mining operations often clear large land areas and can affect the volume and rate of surface water runoff, which in turn affect aquatic systems. Depending on the geographic location, topography, nature of the overburden, and the underlying geology, the flow regime of receiving streams also may be altered by the water-retaining potential of the mine site. Dry weather flow in small streams may be either increased or decreased as the result of changes in storage of groundwater in unconsolidated material at the mine site (U.S. Army Corps of Engineers 1974; Minear and Tschantz 1976). This water retention capacity may alter the response of storm water runoff, which is typically higher and carries a larger load of suspended materials in the absence of vegetative cover.

Surface Water Effects

Surface water quality effects of coal mining can be grouped into those resulting from acid drainage, increased dissolved constituents (e.g., alkaline drainage and trace substances), and increased suspended solids loads (Cushman, Gough, and Moran 1980). The hydrologic changes associated with surface mining can reduce aquatic habitat as the result of the increased storm water runoff (which increases bottom scouring) and increased sedimentation (which fills pools and degrades riffle habitat required by bottom-dwelling organisms) (Hynes 1970; Tolbert 1980; Stair, Tolbert, and Vaughan 1984).

Acidic drainage impacts are greatest to aquatic resources of tributaries, while mainstream rivers generally are not visibly affected because of dilution and neutralization by calcareous materials in the geologic strata (Hyde 1970, U.S. DOE 1981; Herlihy et al. 1990). Where the neutralizing capacity of the area is limited, acidic runoff can exceed the dilution capacity of receiving rivers (Haynes et al. 1979, Office of the Inspector General 1991). Acid mine drainage impacts aquatic biota as a result of low pH, high levels of metals and sulfates, and/or deposition of iron hydroxide precipitates that smother bottom-dwelling organisms (Dvorak et al. 1977). Mining-induced alterations in pH and dissolved contaminants also potentially degrade groundwater, thereby making the water unsuitable for consumption and potentially for irrigation.

Historically, acid mine drainage (AMD) has been shown to continue long after mine closure (U.S. Department of Interior 1978; Short, Black, and Birge 1990), such that impacts to aquatic biota from individual or collective mine sites may continue for undetermined periods of time. Biotic impacts from alkaline mine drainage, which is more typical of mine drainage in the western United States, are primarily the result of turbidity, sedimentation, and osmotic stress from high dissolved solids concentrations. The impacts of alkaline drainage, associated with western coal fields, are often compounded by the low dilution potential of many western streams.

Streams classified as acidic due to AMD are unlikely in most cases to support trout (and probably any other fish populations). For nonacidic classes of mine drainage, conditions do not appear to be toxic, although biota may still suffer impacts as a result of physical habitat and substrate alteration, sedimentation, and altered chemical composition (Herlihy et al. 1990). Adverse effects of coal mining on aquatic resources are attributable, in part, to untreated or inadequately treated effluents or discharges and to discharge of effluents into small waterbodies where flow may be inadequate to dilute effluent concentrations (Haynes et al. 1979, Office of Inspector General 1991). Branson and Batch (1972), Tolbert and Vaughan (1980), and Short et al. (1990) found that the diversity and the composition of the aquatic community in stream reaches not directly impacted by acid mine drainage can be adversely affected by non-acidic mine drainage.

Historically, surface mining for coal has been shown to have significant adverse effects on aquatic resources. In 1969, the Appalachian Regional Commission determined that there were about 10,500 miles of streams in eight States of the Appalachian Region that were affected by surface mining drainage. The pollutants considered in their determination were acid, sediment, sulfates, iron, and hardness; acid was considered the most significant, impacting approximately 5,700 miles of streams (ARC 1969). The Commission determined that acid mine drainage in Appalachia added $3.5 million in annual costs to water users, in addition to the unquantified costs associated with general environmental and aesthetic degradation of affected areas, the destruction of aquatic life, and the deterrents to water-based recreation that could exceed the identified costs (ARC 1969). In 1978, the U.S. Fish and Wildlife Service (U.S. Department of Interior 1978) estimated that orphaned mines in the east and over 10,000 miles of streams affected by mine drainage occur in Appalachia and over 15,000 miles in the eastern United States (U.S. Department of Interior 1978).

Requirements of the Surface Mining Control and Reclamation Act (SMCRA) of 1977 were intended to mitigate many of the environmental impacts historically associated with the mining of coal. However, in spite of these requirements, impacts from acid mine drainage, stream sedimentation, mining of prime farmlands, and impacts to wild and scenic rivers, endangered and threatened plants and animals, and hydrology continue to be of concern (Haynes et al. 1979; Office of the Inspector General 1991). In 1986, Herlihy, et al. (1990) identified areas potentially impacted by mine drainage and, after sampling for verification, estimated that there were over 8,300 miles of mine-drainage impacted streams in the U.S. EPA National Stream Survey (NSS) area, which included the Mid-Atlantic and Southeastern United States. Large areas of Ohio, Kentucky, and West Virginia (which contain major coal deposits) were not included in the NSS. Small intermittent streams, large streams and rivers, and watersheds larger than 60 mi^2 were also not included in their impacted stream reach estimates (Herlihy et al. 1990). A study by the Office of the Inspector General (1991) found that the Office of Surface Mining regulations and State implementation and enforcement in Pennsylvania, Ohio, and West Virginia (the only three State programs studied in detail) were inadequate to prevent impacts to surface and groundwater quality for protection of aquatic life. The study found that at the end of the June 30, 1989, fiscal year there were greater than 4,700 acres of abandoned mine sites that remained unreclaimed in these three States.

In order to accurately address impacts to aquatic resources in a stream receiving surface mining drainage, a site-specific analysis of water quality, underlying geologic characterization, a toxic-handling plan, and characterization of the biological resources are required. For those areas where a significant potential exists for impacts to protected, sensitive, or economically important resources, provisions of SMCRA (U.S. Department of Interior 1979) that provide for designation of areas as unsuitable for all or certain types of coal mining may restrict coal mining and hence impacts. Haynes et al. (1979) concluded that control of effluent sediment loads to within standards set by SMCRA should result in generally acceptable sediment releases from coal mining, but that such control would be difficult to achieve. Thus, point-source and nonpoint-source effluent regulation for coal mining is crucial to minimizing the magnitude of impacts on aquatic resources. However, based on conclusions of the Office of the Inspector General (1991) for Ohio, Pennsylvania, and West Virginia, and the OSM (1991) review of the West Virginia SMCRA implementation program, measures to minimize impacts to water resources from both sedimentation and acid mine drainage are not being met. The Inspector General's report (1991) concluded, based on a recent Pennsylvania report, that the annual cost to treat the State's discharges (those occurring as a result of mining since passage of SMCRA in 1977) ranged from $19 million to $25 million. Examples of impacts to aquatic systems cited in the report showed virtual elimination of aquatic biota in two streams, one in West Virginia and one in Kansas, as the result of acid mine drainage.

Requirements of SMCRA (runoff from mine sites be collected in sedimentation basins, sedimentation basins be designed to contain runoff from a 10-year, 24-hour storm event, and pH be adjusted to between 6.0 and 9.0 prior to release from the mine site) should minimize the potential for acidic drainage leaving the site and for mobilization of metals from the overburden and coal strata (U.S. Department of Interior 1979, Sect. 816.42; Kenny Vaughan, Kentucky Department of Surface Mining, Madisonville, personal communication to V. R. Tolbert, ORNL, August 16, 1991). However, the lack of (1) accurate water-monitoring data, (2) geologic testing, and (3) toxic-handling plans based on identification of acidic strata by geologic testing has contributed to continued discharge of acidic waters from mined sites since 1977 (Office of the Inspector General 1991). Adequate plans and testing are required under the Act, and their preparation in areas with the potential for acidic drainage is particularly important.

Based on the existing SMCR requirements, acid-mine drainage and sediment transport from surface mining sites should be minimized with use of appropriate mining techniques, control techniques during mining, and reclamation. However, review of the literature shows that in many instances significant impacts to water quality can be expected, particularly in the absence of reclamation, and that releases from mine sites could well be expected to affect aquatic biota in the vicinity of a mine site supporting a hypothetical 500 MW plant. Comparison of data from the U.S. Department of Interior's

...in the absence of reclamation and acid mine drainage controls, impacts to water quality could occur.

cumulative hydrologic impact assessment using the MUSLE (modified universal soil loss equation) model supports the conclusion that, in the absence of reclamation and acid mine drainage controls, impacts to water quality could occur. The data show that for a small watershed (5 mi^2), sediment released during mining could increase 10-fold, while the increase for a larger watershed (25 mi^2) as the result of mining the same sized area (e.g., 380 acres) would be expected to increase by less than one-half a percent (data provided by James Hughes, Office of Surface Mining, Division of Permitting, Knoxville, Tennessee, to V. R. Tolbert, ORNL, 28 October 1991).

Recreational Fisheries Resources

Impacts to water quality and subsequently to aquatic biota as a result of mine drainage are most pronounced in small streams that generally intercept runoff from mining areas. Drainage from surface mines is more likely to affect recreational fishing, species of special concern (threatened and endangered species), and biological diversity than commercial fishing; commercial fishing is not a major industry in coal-bearing regions of the United States or in the small streams that are most likely to be severely impacted by mining. Recreational fishes, which commonly occur in ponds, rivers, and reservoirs of the southeast, are harvested for sport or consumption by the fisherman. The most important warmwater recreational fish species include members of the families Centrarchidae (largemouth and smallmouth bass, bluegills, crappie), Ictaluridae (catfish), and Percidae (perches, walleye, sauger). Striped bass fisheries exist in many large reservoirs. Because sport fish populations are generally less intensively studied than are commercial stocks, biological damage is more difficult to quantify. Recreational fishing occurs primarily in medium to large rivers and reservoirs rather than in the small streams that are generally associated with mining areas. The major impacts associated with acid-mine drainage and heavy sediment release from mine sites would be expected to occur in the smaller streams rather than in waterbodies with greater recreational fisheries resources.

In the coal-bearing regions of the western United States, particularly in the more mountainous, less arid portions of Utah and southern Colorado, trout would be the recreational fish most commonly impacted. In coal-bearing regions of Arizona and New Mexico, and drier portions of Utah, Colorado, Wyoming, and Montana, most streams are ephemeral or maintain flow for only portions of the year. Potential impacts in these areas would be the result of reduced groundwater flow or contamination and impacts on water availability for irrigation rather than on recreational species. Studies in the Powder River Basin of Wyoming and Montana determined that aquifers receiving recharge from coal spoil may contain dissolved solids concentrations two to three times greater than those in adjacent undisturbed aquifers (BLM 1981). Concentrations of heavy metals in spoil leachate were also determined to be greater than in most of the natural surface water and exceeded recommended maximum concentrations for irrigation on a continuous basis, livestock use, public supply, and for protection of aquatic biota (BLM 1979). As in the east, if reclamation is carried out as prescribed by SMCRA, sedimentation from mining would be expected to increase in the short-term in much of the western coal-bearing area but to decrease after reclamation and stabilization by vegetation. Measures to minimize contact of surface with spoil material both during and following mining would minimize water quality degradation of underlying aquifers.

5.2.2 Case Study

Despite passage of SMCRA, which resulted in regulations requiring significant changes in mining methods, overburden handling, sediment control, acid-mine drainage treatment, and reclamation, quantitative information is lacking that relates mining impacts to fisheries resources. Potential impacts to aquatic resources from surface mining extraction of coal to support a hypothetical 500-MW conventional generation facility in eastern Tennessee were calculated in an attempt to quantify impacts to recreational fisheries. These calculations were based on assumptions about the coal needs (approximately 1.9 million tons of coal per year from a hypothetical mine site in northwestern Kentucky) and the characteristics of the watersheds analyzed. Based on production of approximately 5,000 tons of cleaned coal/acre for a mine in northwestern Kentucky (Jim Grise, Green Coal Company, personal communication to V. R. Tolbert, ORNL, 27 August 1991), approximately 380 acres/year (1.537 km^2/year) would be disturbed to supply the necessary coal for this 500-MW plant over a projected life span of forty years.

There are no data on the numbers of sport fish caught in small streams in northwestern Kentucky. Further, it is difficult, if not impossible, to directly quantify surface mining impacts on recreational fisheries. Therefore, two adjacent watersheds (one with mining, one without) were compared using survey data derived by sampling similar areas of streams using either electroshocking equipment or chemical sampling (Commonwealth of Kentucky 1981, 1982, 1983; David Bell, Fisheries Biologist, Kentucky Division of Water, Madisonville, personal communication to V. R. Tolbert, ORNL, August 16, 1991). As is typical of most smaller streams, water quality data, which are generally restricted to measurement of dissolved oxygen, pH, alkalinity, temperature, and transparency, were unavailable for comparison of most streams in the area.

Available information on two adjacent watersheds in northwestern Kentucky, one with approximately 10.5 km^2 of mining disturbance and one with no mining, and fisheries resources in the streams draining them were used in an attempt to make a comparison of the resources and to identify potential impacts that might be associated with mining. The information from this comparison can be used to estimate the potential impacts that might be associated with mining of 380 acres/year (1.54 km^2/year) of a watershed. Qualitative factors that affect the assumptions and application of the conclusions to the determination of actual impacts to fisheries resources were discussed in the "Background" section.

Two 5th-order streams primarily within Daviess County (North Fork Panther Creek and South Fork Panther Creek) were compared in an attempt to assess potential effects of mining on fish populations (Table 2) and to qualitatively determine impacts that could occur to recreational fisheries. According to David Bell Kentucky Division of Water, personal communication to V. R. Tolbert, ORNL, August 15, (1991), coal mining has occurred only in the South Fork watershed. Examination of map overlays for the watershed provided by the Kentucky Department of Environmental Resources (1991) show that a total area of approximately 10.5 km^2 has been mined upstream of the fisheries sampling site since 1981; this would average 1.05 km^2 mined/year. Because the two watersheds are adjacent, approximately the same size, and otherwise have approximately the same type of agricultural land use [based on the SCS (1974) report], mining was considered to be their major difference. The Panther Creek watersheds were chosen for comparison because of their proximity to the hypothetical mine area, their similarity in major characteristics, and mining occurring in the South Fork watershed only since implementation of SMCRA requirements. The mine sites in the South Fork watershed have all been reclaimed according to SMCRA regulations (Kenny Vaughan, Kentucky Division of Surface Mining, personal communication to V. R. Tolbert, August 16, 1991).

...the number of recreational fish species has been reduced by at least 3%...sedimentation was determined to be the cause of these reductions.

Comparison of the qualitative fisheries information for the two Panther Creek watersheds shows that the number of recreational fish species has been reduced by at least 3% in South Fork compared to North Fork (Table 2). For these two streams, this would be the equivalent of a reduction from 392 recreational fish per acre of waterbody to 379 fish per acre (Table 2). Considering all species, the density of fish in the mining-impacted stream was approximately half that of the non-mining-impacted stream. The greatest reductions in recreational species in South Fork were those species that visually locate prey within the water column (e.g., largemouth bass, sunfish). Based on the diversity of fish species in the South Fork (Table 2) (Commonwealth of Kentucky 1981, 1982, 1983), it is probable that this stream is not impacted by acid mine drainage (AMD). Herlihy et al. (1990) and Short et al. (1990) showed that, in streams impacted by AMD, fish populations would be severely reduced to absent.

Table 2. Comparison of fisheries resources in South Fork Panther Creek[a] and North Fork Panther Creek

Species	South Fork[b]		North Fork[c]	
	No. Collected	No./Acre	No. Collected	No./Acre
Spotted bass (s)[d]	5	32	0	0
Largemouth bass (s)	5	32	13	87
Black crappie (s)	4	25	2	13
White crappie (s)	2	13	0	
Channel catfish (s)	9	57	4	26
Bluegill (s)	11	69	3	20
Green sunfish (s)	3	19	5	33
Longear sunfish (s)	13	82	52	343
Warmouth (s)	1	6	5	33
Carp	14	88	142	937
Freshwater drum	9	57	2	13
Yellow bullhead (s)	7	44	27	180
Gizzard shad	18	114	157	1046
Emerald shiner	4	25	0	
Mimic shiner	1	6	0	
Bluntnose minnow	72	455	0	
Mosquitofish	70	442	2	13
Grass pickerel	0		1	6
White sucker	0		20	133
Blackstraped top minnow	0		6	39
Silverjaw minnow	0		5	33
Spotfin shiner	0		15	100
Number species	17		17	
Number individuals	250	1568	461	3049
No. sport fish spp.	10		8	
No. sport fish		379		392
% reduction sport fish spp.[e]	3%			
% reduction overall[f]	49%			

[a]Mining disturbed.
[b]6900 ft^2 sampled.
[c]6540 ft^2 sampled.
[d]Sport fish.

[e]Calculated as $100 \times \left(1 - \frac{\text{sport fish per acre South Forl}}{\text{sport fish per acre North Forl}}\right)$

[f]Calculated as $100 \times \left(1 - \frac{\text{total fish per acre South Fork}}{\text{total fish per acre North Fork}}\right)$

Approximately 45% of the mining recorded to date in the South Fork watershed occurred during the 1981 to 1983 time frame (Jeno Blassa, Kentucky Department of Natural Resources, map overlays, 1991), which is within the time period that the fisheries data were collected. Information from the Kentucky Department of Natural Resources showed that the mine sites in the South Fork watershed were reclaimed as appropriate under SMCRA regulations. (It is assumed that if AMD is occurring into the small headwater streams that feed South Fork, the effects are neutralized before reaching the sampling location based on the fish species and densities present. There were no water quality data to determine presence or absence of AMD for comparison of the two streams.) Stair et al. (1984) and Branson and Batch (1972, 1974) found that, even in the absence of acidic mine drainage, fish species and populations were reduced in streams receiving drainage from mine sites; sedimentation was determined to be the cause of these reductions.

5.2.3 Application of the Case Study to Other Sites

The information from comparison of fish populations in the two forks of Panther Creek (Table 2) was used to see if it appeared reasonable to estimate reductions in fish populations in Richland Slough. Richland Slough, one of the last remaining unimpacted ox-bows in western Kentucky, is a unique resource near the hypothetical mine site and is connected to the Green River at both ends. The Slough, which drains a wetland area, is a warmwater body with sluggish waters, low flow, and heavy siltation. The fish fauna is comprised primarily of lowland fishes which are tolerant of heavy siltation and low flow (David Bell, Fisheries Biologist, Kentucky Division of Water, personal communication to V. R. Tolbert, 16 August 1991). Based on a comparison of the qualitative fish survey information for North and South Forks of Panther Creek (Table 2), approximately a 3% reduction in the numbers of individuals of recreational species within the fish community of Richland Slough could occur as the result of sediment from mining disturbance (Table 3). Impacts from AMD would be expected to result in elimination of all but the most tolerant fish species. Using the reduction in recreational fisheries populations in Panther Creek to predict potential reductions in Richland Slough does not address the potential impacts to fisheries resources of the ox-bow from use as a spawning or nursery area. This use could be of primary importance to the fisheries resource of both the Slough and the Green River in the hypothetical mine vicinity. The qualitative comparison of impacts to Richland Slough also does not consider the ecological importance of this being probably the last undisturbed ox-bow in the area (David Bell, Kentucky Division of Water, personal communication to V. R. Tolbert, ORNL, August 16, 1991).

The application of this approach to other mining areas and to determination of the potential effects of mining on fisheries resources in other streams is subject to constraints and limitations. The above analysis assumed that mining was the only impact parameter in the watershed and that mining and reclamation occurred

Table 3. Fisheries resources in Richland Slough

Species	Number collected	Number/Acre
Largemouth bass	1	6
Bluegill	18	101
Longear sunfish	20	112
Redear sunfish	1	6
Warmouth	6	34
Smallmouth buffalo	2	11
Freshwater drums	2	11
Gizzard shad	3	17
Number of species	8	
Number of indiv.	53	305
Number rec. spp.	5	
Number rec. indiv.	46	259

as prescribed by the Surface Mining Control and Reclamation Act. Haynes et al. (1979) provided an estimate of 17,000 metric tons sediment/km^2/yr as a most likely scenario for mining; this amount was supported by the Kentucky Division of Water (Mike Mills) as a reasonable estimate for that State. Given the differences in mining techniques, differing geologic strata, geochemistry, water quality, and reclamation success as discussed in the Office of Inspector General's 1991 report and OSM evaluation of the West Virginia mining and inspection and enforcement of SMCRA (OSM 1991), quantitative extrapolation from one site to another to predict either the effects on recreational fisheries or to determine an economic endpoint is highly speculative. Only general, qualitative comparisons of effects should be made. Comparison of lowland slack-water streams in low elevation watersheds typical of northwestern Kentucky and southern Illinois and Indiana with mountainous, high-gradient streams in eastern Kentucky, eastern Tennessee, West Virginia, and western Pennsylvania would give erroneous conclusions as to the effects of surface mining on aquatic communities because of the differences in the physical factors listed above as well as differences in the fish species and population sizes. Sediment would probably accumulate in greater amounts in the larger, low-gradient streams and rivers because of the less frequent flushing of these systems and the deposition from upstream sources. The impacts from sediment deposition in the higher gradient streams of more mountainous areas would probably be greater because the fish populations in these streams are dependent upon clean riffle and pool areas for spawning and feeding. Fish in mountainous streams depend upon benthic invertebrates as a primary food source;

these species have been shown to be reduced in streams subjected to sediment deposition and subsequent habitat destruction.

... quantitative extrapolation from one site to another to predict either the effects on recreational fisheries or to determine an economic endpoint is highly speculative.

Estimation of impacts to aquatic biota is based on an average erosion rate from active surface mining of 17,000 metric tons of sediment/km^2/year or about 2,000 times the amount from an undisturbed forest (Haynes et al. 1979). Because of the size of the Green River and its watershed relative to the 46 km^2 area that would be disturbed over a thirty-year life of the power plant, an increase of 784,000 metric tons (26,130 metric tons/1.537 km^2/year) of sediment into this waterbody would be expected to have a minimal overall impact. This value may underestimate the amount of sediment produced from a mine site; however, the calculation assumes that the entire mining area is not exposed at any one time. Reclamation is assumed to occur as mining progresses according to requirements of SMCRA which would help minimize soil erosion. For Panther Creek, the mine sites were assumed not to be adjacent to receiving headwater streams; there was no comparison of fisheries resources in the headwater streams. The riverine nature of the Green River compared to Richland Slough should minimize sediment buildup in the river. By comparison, the low flow in the ox-bow could allow sediment accumulation and potential habitat destruction. Because of the size of the Green River watershed, the considerable extent of agriculture in the watershed, and subsequent nonpoint source runoff, sedimentation impacts to the river from the increased mining activity would be expected to have a minimal incremental effect.

With the lack of reclamation of more than 4,700 acres identified in Pennsylvania, West Virginia, and Ohio alone (Office of the Inspector General 1991), mining could contribute significant amounts of sediment and either AMD in the east or alkaline drainage in the west to water resources in States with coal-bearing strata in proximity to streams. Depending upon the erodibility of soils in the mine area and the stability of the slope (Esling and Drake 1988), this value could be more or less for a particular mine site. Kenny Vaughan, Department of Surface Mining, Enforcement, Regional Office, Madisonville, Kentucky (personal communication to V. R. Tolbert, ORNL, August 16, 1991) and Mike Mills, Fisheries Biologist with the Kentucky Division of Water (personal communication to V. R. Tolbert, ORNL, August 26, 1991), concluded that the estimate of 14,000 metric tons of sediment/km^2 of active surface mining/year was appropriate for northwestern Kentucky. The erosion potentially associated with mining can be used along with information on the number of acres mined in a watershed to calculate the amount of sediment that can be mobilized from a mine site before, during, and after mining (James Hughes, U.S. Department of the Interior, Office of Surface Mining, Knoxville, Tennessee, personal communication to V. R. Tolbert, ORNL, September 24, 1991). As discussed earlier, for a small watershed (5 $miles^2$), sediment released during mining can increase 10-fold while the increase for a larger watershed (25 $miles^2$) as the result of mining the same sized area (e.g., 380 acres) would be expected to increase by less than one-half a percent. This comparison shows that the major impacts associated with surface mining would occur in small watersheds such as used in the case study comparison.

There has been no assessment of impacts associated with mining on fisheries in Kentucky nor any attempt to quantify sedimentation or AMD associated with mining on a site-specific or regional scale. There have been no attempts made to compare pre- and post-mining disturbance and impacts on fish populations in Kentucky (David Bell, Kentucky Division of Water, Madisonville, personal communication to V. R. Tolbert, ORNL, August 16, 1991, Mike Mills, Kentucky Division of Water, Frankfort, personal communication to V. R. Tolbert, ORNL, August 26, 1991). In addition information on fisheries is generally restricted to qualitative data because creel survey information is generally restricted to larger rivers and reservoirs (e.g., Commonwealth of Kentucky 1981, 1982, 1983, 1988). This resulted in the comparison of North and South Forks of Panther Creek to estimate the reductions in fish populations that could potentially occur as the result of mining impacts. According to Mike Mills (Fisheries Biologist, Kentucky Division of Water, personal communication to V. R. Tolbert, ORNL, August 26, 1991), the major impacts to fisheries resources in Kentucky would be expected to be from sedimentation. Acid mine drainage is not regarded as a problem in Kentucky since implementation of SMCRA (Kenny Vaughan, Kentucky Division of Surface Mining, personal communication to V. R. Tolbert, ORNL, August 16, 1991); however, for those sites not mined and reclaimed according to SMCRA regulations, AMD could potentially be a problem.

5.2.4 Use of Models to Estimate Effects on Recreational Fisheries

Several models have been used in an attempt to determine the economic value of damages to recreational fishery resources. Ribaudo and Piper (1991) used water quality variables and information from the 1980 National Survey of Hunting, Fishing, and Wildlife-Associated Recreation (which included in-person interviews with a subsample of fishermen) to provide more detailed information on fishing habits, expenses, and success. They proposed to use this information for a national model to estimate changes in recreational fishing participation from national water quality policies without having to use site data. They concluded that changes in recreational fishing activity due to improvement in regional water quality can be measured, in an aggregate sense, with a sequential decision model. This calculation was determined to be useful where direction and general magnitude of change were all that was required (Ribaudo and Piper 1991).

However, these authors also determined that, for a national conclusion based on regional data and for more accurate results, regions need to be broken down into smaller units.

The models used to determine the value of damages to recreational trout fishing in the upper Northeast due to acidic deposition were based on specific, carefully measured chemical parameters that are specific to acidic deposition effects (Brown et al. 1991), and detailed laboratory studies with selected fish species were used to determine effects (Baker et al. 1991; Mount et al. 1989). These models were linked to an economic evaluation model using an intricate set of assumptions and actual interviews with fishermen (Englin, Cameron, Mendelsohn, Parsons, and Shankle 1991). Adapting this methodology to evaluating the economic effects of coal mining would require substantial additional research and gathering of information on the above parameters. In particular, the methodology would need to be reviewed, investigated, and examined for applicability to this different situation. Better methods of evaluating the physical effects of surface mining on water quality parameters would be needed. Finally, information about fishermen and fishing activity, particularly in smaller rivers and streams, would need to be obtained for the affected regions.

A method for estimating the impact of surface coal mining and different land reclamation practices on trout populations in streams draining proposed strip mining areas in the Yampa River Basin of northwestern Colorado was presented by Smith (1980). A regression was developed for suspended sediment concentration as a function of stream flow under baseline (pre-mining) conditions. Estimates of the proportional increments of sediment associated with mining, and for four reclamation scenarios were based on other studies. The information on suspended sediment, together with flow simulations based on a 58-year flow record, enabled prediction of sediment concentrations in streams. Laboratory studies of rainbow trout egg and adult mortality were used to predict effects of sediment levels on salmonids. A Leslie matrix model, parameterized using brook trout life history data from Michigan and a fitted function to describe density-dependent mortality, was used to project mining effects over a 100-year period, given different reclamation scenarios.

Smith (1980) recognized stringent limitations to the conclusions that can be drawn from his results. He mentioned in particular the need to quantify the level of density-dependent mortality in the population. Investigations of this topic in another context by Christensen and Goodyear (1988) indicate that such quantification may not be possible with Smith's approach. Smith acknowledged the need to calibrate and verify the model with field data before applying it to any of the Yampa subbasins, and suggested that his results be used to illuminate data needs rather than to make predictions.

5.2.5 Conclusions and Constraints to Application of the Case Study

The applicability of the comparative case study presented previously to other situations is limited by assumptions and data gaps as have been discussed. These same constraints, particularly the absence of water quality information, site specific data, and information on response of resident fish populations, limit the ability to apply ecologic models to quantify the effects of surface mining on fisheries resources. Although the effects of surface mining may ultimately extend to the larger streams and rivers in the form of sediment effects, these effects are difficult to attribute solely to mining because of the overall contribution of agriculture and other land use practices to sediment loads in receiving water bodies. The lack of information on fisheries resources in smaller streams is particularly evident because these water bodies generally do not provide recreational fishing opportunities. This does not decrease their overall importance to the ecological system nor their overall economic value, only their recreational fisheries value. These systems provide the appropriate habitat for fish species of special concern, for example, Cumberland darter, Tennessee red-bellied dace, even though these species are of no recreational value.

... the absence of water quality information, site specific data, and information on response of resident fish populations, limit the ability to apply ecologic models to quantify the effects of surface mining on fisheries resources.

From the background discussion and calculation of case study values, it is evident that surface mining has definite and potentially significant effects on aquatic resources whether recreational or non-recreational fish species or benthic food organisms. If surface mining is conducted according to the requirements of SMCRA, the impacts can be minimized. However, as the levels of sediment, acidity, and metals increase in the runoff leaving mined sites, the impacts to fisheries resources and aquatic biota in general will increase. By assuming best reclamation practices, minimal impacts to fisheries resources can be projected and calculations of externalities associated with mining can be assumed to be minimal. If on the other hand, bond is forfeited and no reclamation occurs, or if stabilization of the site does not occur following reclamation, fisheries resources can be estimated to be virtually eliminated. The difference probably lies somewhere in between and would need to be determined on a site-by-site basis. When SMCRA regulations can be adequately enforced and impacts can be accurately predicted based on water quality data, geologic characterization, toxic-handling plans, and applicable reclamation plans, links between mining activity, effects on fisheries resources, and economic endpoints may be more feasible. Without these data, impacts to fisheries resources cannot be quantified.

5.3 EFFECTS OF ATMOSPHERIC MERCURY INPUTS ON AQUATIC RESOURCES

5.3.1 Background

From the perspective of health and environmental risks, mercury is one of the most important of the heavy metals associated with the coal fuel cycle. Relative to most other metals, mercury is highly mobile in the environment and is highly toxic both to man and to nonhuman biota. Mercury can exist in the environment in several chemical forms, the most important of which is an organic form, methylmercury (MeHg). Methyl mercury is especially significant because it is readily absorbed by organisms and is efficiently transferred from prey to predators. Piscivorous (i.e., fish-eating) fish can accumulate extremely high levels of methylmercury even when concentrations in the water-column and sediment are nearly undetectable. Fish-eating birds and mammals can accumulate toxic levels of methylmercury from their diets. Human populations where subsistence fishing is common are also susceptible to high levels of methylmercury exposure (Travis and Hester 1990, Clarkson 1990).

5.3.2 Relevance to the Coal Fuel Cycle

In the late 1960s it was shown that inorganic mercury (HG^{++}) in lake and river sediments can be transformed by microbes into methylmercury (Jensen and Jernelov 1969). Trace quantities of mercury are found in coal and can be released to the environment in atmospheric, aqueous, and solid waste streams. Studies of mercury as related to the coal fuel cycles were common in the 1970s (e.g., Braunstein et al. 1977, Braunstein 1978). Until recently, the principal pathways considered in assessments of coal-derived mercury were aqueous emissions due to leachate from coal and solid wastes. Known instances of contamination and human health effects related solely to point sources, primarily associated with mining and smelting where release rates are far higher than those associated with coal combustion (Braunstein et al. 1978). Atmospheric emissions were not thought to be a problem except in the vicinity of uncontrolled point sources (Goldwater 1971). As recently as 1989, the Final Programmatic Environmental Impact Statement for the Clean Coal Technology Demonstration Program (DOE 1989) included no discussion of atmospheric mercury emissions.

Recent studies suggest that atmospheric emissions of mercury are much more significant than previously thought, and that combustion of coal may be a major contributor to these emissions (Mitra 1986). Instances of mercury concentrations in fish exceeding human health standards have become increasingly frequent since 1980 (Sorenson et al. 1990, Glass et al. 1990, Travis and Hester 1990). Many of the lakes affected are in remote areas far from point sources. Mass balance studies have shown that atmospheric loading is the principal source of mercury in these lakes (Sorenson et al. 1990). In some cases (e.g., Glass et al. 1990), specific sources such as incinerators can be identified. For most contaminated lakes, however, no plausible local source can be identified. Rada et al. (1989) surveyed mercury in sediment cores of 11 Wisconsin lakes and concluded that atmospheric mercury deposition has been increasing.

5.3.3 Potential Ecological Effects

There is no evidence that the fish themselves are being adversely affected by mercury. However, piscivorous birds and mammals are vulnerable to adverse effects from high levels of mercury in their diets (Eisler 1987). Mercury has been identified as a contaminant of concern in Watts Bar Reservoir, Tennessee because of potential effects on piscivorous wildlife (Suter 1991).

Recreational fishing is already being affected by mercury because of health advisories that apply to consumption of fish. The states of Minnesota, Wisconsin, and Michigan have issued public health warnings for several hundred lakes, and Michigan has issued a warning covering all 10,000 of its inland lakes (Glass et al. 1990). Species with the highest reported levels include walleye and northern pike (Sorensen et al. 1990).

5.3.4 Principal Scientific Uncertainties

Key areas of uncertainty concerning the significance of airborne mercury deposition include: (1) the relationship between sources and deposition, and (2) relationship between water body characteristics and bioaccumulation in fish.

Key areas of uncertainty concerning the significance of airborne mercury deposition include: (1) the relationship between sources and deposition, and (2) relationship between water body characteristics and bioaccumulation in fish.

The global mercury cycle involves a number of chemical transformations that are not well understood (Porcella et al. in press). Atmospheric mercury exists principally as elemental mercury vapor (Hg^0). This is the form released during coal combustion and incinerator operation. The oceans and, potentially, terrestrial ecosystems (Lindberg et al. 1979) are also sources of Hg^0. Natural production of elemental mercury involves microbial reduction of ionic mercury (Hg^{++}) to volatile

Hg^0. Anthropogenic sources such as coal-fired power plants and incinerators release Hg^0 directly. The inference that a large fraction (up to 1/3 globally) of atmospheric mercury is derived from observations that atmospheric concentrations and deposition rates are nearly twice as high over the heavily industrialized northern hemisphere as over the southern hemisphere (Fitzgerald 1989). Recent estimates that include emissions from eastern European countries suggest that up to half of the anthropogenic inputs of mercury (1/6 of the global total) may come from coal combustion (Nriagu and Pacyna 1988).

Studies of mercury deposition in rainfall (wet deposition) show that mercury is deposited principally as Hg^{++}. The chemical reactions responsible for this transformation are not well understood. Qualitative aspects of the fate of mercury in aquatic systems are known, but quantitative understanding is limited. Measurements of total dissolved mercury in freshwater are highly variable. Measurements made prior to the mid-1980s are considered unreliable due to the insensitivity of available analytical techniques (Fitzgerald and Watras 1989). Factors controlling the transformation of Hg^{++} to meHg are being investigated but are not well understood. Empirical studies of lakes in the Great Lakes show that high bioaccumulation of mercury in fish is associated with the same factors that make lakes vulnerable to acidification: low pH, alkalinity, calcium and specific conductance (Grieb et al. 1990, Sorensen et al. 1990, McMurtry et al. 1989). Continental-scale bioaccumulation has not been observed. In Watts Bar Reservoir, Tennessee, for example, concentrations measured in fish do not exceed regulatory action levels for human consumption, in spite of historically large releases of mercury from nearby Department of Energy facilities (Hoffman et al. 1991).

5.3.5 Feasibility of Quantifying Externalities

Quantification of ecological impacts of atmospheric mercury derived from coal combustion is not now possible, because (1) the relationships between release rates and deposition rates, and (2) the distribution of water bodies vulnerable to mercury bioaccumulation are unknown. These uncertainties are currently being addressed in a research program sponsored by the Electric Power Research Institute (EPRI 1990).

5.4 EFFECTS OF AIR POLLUTANTS AND ACIDIC DEPOSITION ON CROP AND FOREST VEGETATION

5.4.1 Introduction

5.4.1.1 Purpose

The purpose of this section is to estimate the reduction in economic yield of forests, food crops, and forage crops if they are exposed to emissions from a hypothetical coal-fired power plant. One hypothetical power-plant site in the Southeast and one hypothetical site in the Southwest will be examined. The immediate objective of this study is to provide, for each crop of interest, an exposure-response function that can be used to estimate yield reductions when the ambient level of an air pollutant is raised to a new, higher level. A later objective, to be met after air pollutant levels are determined for the regions around the hypothetical power-plant sites, is to estimate the crop losses that would occur in these regions. The immediate objective is accomplished by obtaining the necessary information from existing literature.

5.4.1.2 Literature Background

The impacts of air pollutants on vegetation and crops were recently addressed by two major research programs—the National Acid Precipitation Assessment Program (NAPAP) and the National Crop Loss Assessment Network (NCLAN). The results of these programs are reported by Shriner et al. (1991) and Heck et al. (1988), respectively. Also, Shriner et al. (p. 2-202 ff.) summarize the NCLAN program and its results.

The NAPAP began with Congressional authorization under the Acid Precipitation Act of 1980 (P.L. 96-294, Title VII). The program is managed by a Joint Chairs Council including the U.S. Environmental Protection Agency (EPA), National Oceanic and Atmospheric Administration, Department of Agriculture, Department of Energy, Department of Interior, and the Council on Environmental Quality. One of the many NAPAP objectives includes the development of exposure-response functions with respect to soils, soil organisms, aquatic and amphibious organisms, crop plants, and forest plants. Other topics include sources of acid rain, monitoring of the levels of acid precipitation, atmospheric chemistry and transport, and impacts of acid precipitation on metals, wood, paint, masonry, and public welfare. More than 1000 scientific publications have been supported by NAPAP. A comprehensive State-of-the-Science/Technology (SOS/T) report has been prepared or is nearing completion for each of the 26 or so subject areas covered by NAPAP. One of these is the Shriner et al. report (1990), which addresses the effects of major air pollutants and acid rain on crops and natural vegetation. An Integrated Assessment, which is being prepared based on all SOS/T reports, focuses on providing information necessary for policy makers (NAPAP 1990).

The NCLAN, which began about 1980 and ended in 1988, stemmed from a growing concern in the 1970s for the impacts of air pollutants on agriculture. To address this concern, the EPA interacted with other government agencies and several private institutions to initiate a program that came to be known as the NCLAN. The primary purpose of NCLAN was to determine the economic impacts on agriculture of O_3, SO_2, and NO_2, alone and in combination, emphasizing O_3 (Wilhour 1988). Research efforts focused on major agricultural crops and major production regions.

For O_3, an empirical research approach involving field experiments was adopted that required "(1) O_3 exposure-response functions be developed for major crop species and cultivars, (2) crops be grown under normal agricultural practices and exposed to O_3 in regions where they are economically important, and (3) common experimental methodologies and quality assurance procedures be employed at all research sites in all regions" (Wilhour 1988). Field studies of plant response began in 1980, and a variety of special studies were also conducted to support the interpretation of empirically generated exposure-response functions.

Adequate O_3 data for a national evaluation of crop losses expected from ambient O_3 were not available for many agricultural regions. Therefore, the NCLAN adapted the use of kriging (a spatial statistical method) to estimate seasonal mean, ambient O_3 levels over large geographic regions (Fig. 7).

The exposure-response functions developed from the field studies and the O_3 levels obtained by the use of the kriging method provided the basis for a national economic assessment of crop losses. The NCLAN also estimated the economic benefits of hypothetical reductions in ambient O_3 levels.

5.4.1.3 General Background on Air Pollutant Effects

Air pollution can damage plant tissue and cause decreases in production of crops and native vegetation. In recent years extensive studies of the effects of O_3, SO_2, NO_2, and acid deposition on crops have been performed. These studies have recently been summarized by Shriner et al.(1990). The toxicities (% reduction in plant growth per unit pollutant exposure) of the major air pollutants have been found to decrease in the order: $O_3 > SO_2 >$ acidic deposition $> NO_2$. Responses to all of these pollutants vary among crop species and cultivars and are influenced by local meteorology, soil conditions, and background pollutant levels.

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In the United States, ambient SO_2 levels are generally far below toxic concentrations except in the immediate vicinity of point sources. Ambient concentrations of NO_2 and current acid deposition rates in the United States are also well below concentrations found to be toxic to cultivated plants. Depending on soil nutrient status, small amounts of sulfur

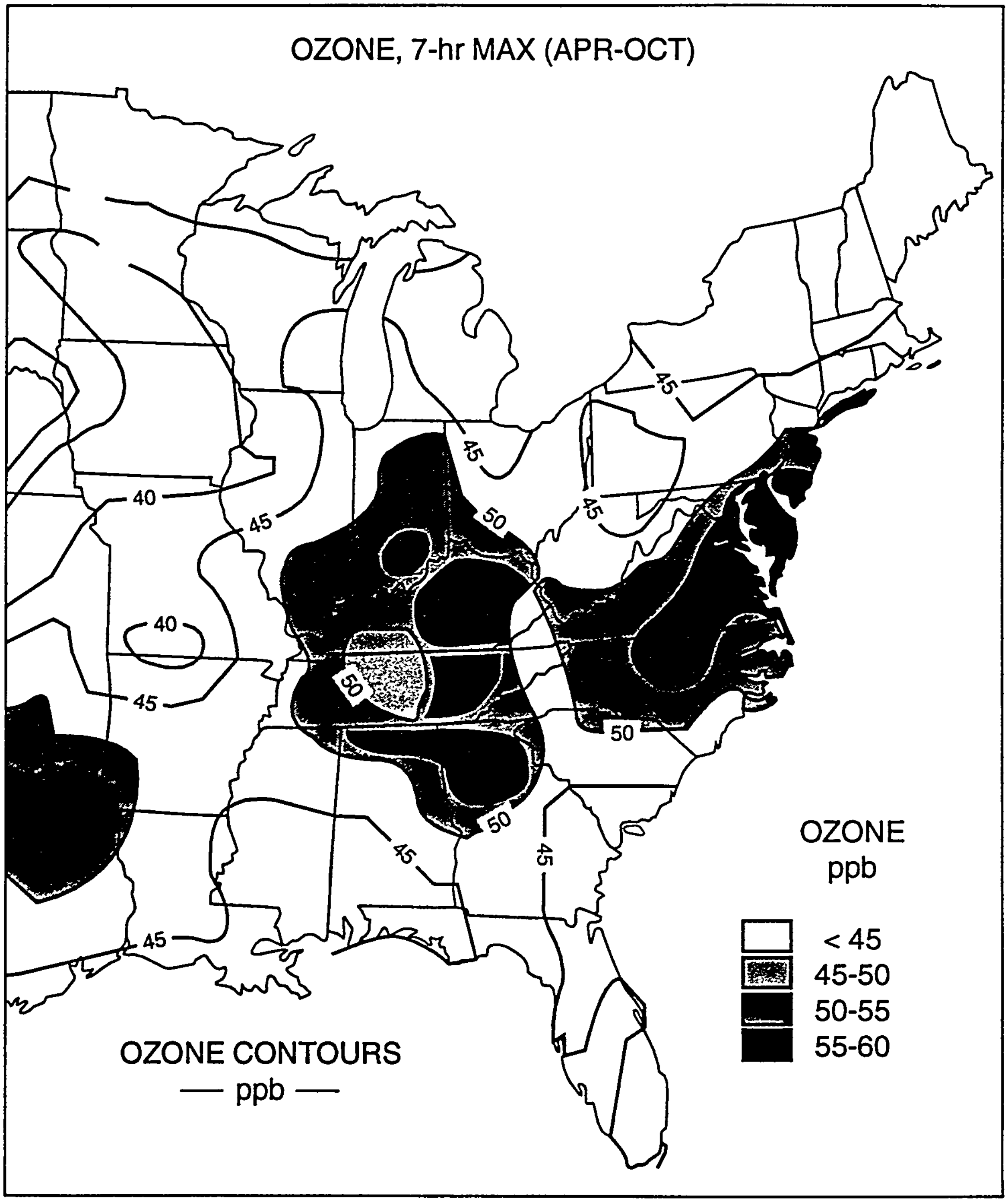

Fig. 7. Growing season averaged maximum 7-h daily ozone concentrations in parts per billion, for the eastern United States (1980-1985). Heck et al. (1988).

and nitrogen deposition from anthropogenic sources may actually improve crop production.

In contrast to SO_2 and NO_2, O_3 has been demonstrated to be causing damage to forests in southern California and at high elevations in the eastern United States. Although visible damage to crop plants has not been documented, ambient levels of O_3 throughout the United States exceed concentrations found to reduce crop plant growth in controlled experiments (Shriner et al. 1990). Assessments of regional and national losses due to O_3 exposure have been performed (Adams et al. 1988).

5.4.2 Impacts of Ozone on Agricultural Crops

The NCLAN report (Heck et al. 1988) provided exposure-response functions for ozone effects on major crops. NCLAN research, limited by time and financial constraints, was designed to address the major factors that could influence crop response to ozone. However, the magnitude of influence of many factors remains relatively poorly quantified. These factors include the effects of cultivar genetics (new commercial cultivars are continually being introduced), developmental stage, environmental variables, pollutant interactions, exposure dynamics, and actual O_3 levels in agricultural regions. Thus, the NCLAN was limited in its ability to determine the representativeness of the crop exposure-response functions for all of these factors. Preston and Tingey (1988) discuss specific uncertainties and the research needs for addressing them.

Exposures of plants to O_3 are usually reported in terms of 7-h or 12-h seasonal mean ozone concentrations. The means represent the daily periods, during the growing season, from 9 am-4 pm and 9 am-9 pm standard time, which are thought to correspond to the periods of high plant sensitivity and high O_3 levels. However, the literature indicates that these seasonal means may not indicate plant response as well as means based on higher, shorter-term exposures, for example, a seasonal mean of daily 1-h maximums. Hogsett et al. (1988) provide detailed discussions of these means, other indices of exposure relevant to estimating plant response (see also Lee et al. 1988), and factors that should be considered in selecting an appropriate mean exposure statistic for assessment studies.

Exposure-response functions for O_3 were produced by 38 NCLAN field experiments based on the use of open-top chambers. Heagle et al. (1988) discuss in detail the possible influences of the design of these chambers and NCLAN methods on the resulting exposure-response functions. NCLAN methods could have influenced the functions due to differences in plant culture, cultivar selection, exposure techniques, and experimental design. Chamber design affected microclimate, plant gas exchange, plant growth, and O_3 fluctuation and gradients.

The Weibull model, which fit most observed crop responses shown by the field studies, possessed a number of desirable features (Shriner et al. 1990, p. 2-195) and was used by Heagle et al. (1988) to represent the NCLAN exposure-response functions. Based on these functions, yield losses for major crop cultivars were presented for five seasonal mean O_3 concentrations representative of the range of ambient O_3 levels in the United States (Table 4). The yield losses presented in this table can be used for the purpose of the current study, that is, to estimate yield losses resulting from an increase in O_3 levels caused by a coal-fired power plant. For a given predicted increase in O_3, the yield loss may be estimated by interpolation of the table's data.

Expected changes in seasonal 7 or 12 h average ozone concentrations as a result of the proposed ozone increment associated the hypothetical power plant are small ($\leq$5 ppb). This suggests that, for the range of seasonal ambient ozone concentrations typical of much of the eastern United States (40-50 ppb), the yield reduction expected from a hypothetical power plant is <1% (Fig. 8) for a relatively ozone-tolerant species such as sorghum, and ~4% for a relatively ozone-sensitive species such as peanut (Fig. 9). Figures 8 and 9 illustrate that, for the purposes of this analysis, linear extrapolation from the values in Table 4 is appropriate in the 40-50 ppb range.

The crop cultivars listed in Table 4 were selected for illustration from those presented by Heagle et al. (1988) and do not represent all cultivars studied or all regions where field studies were conducted. To assess crop losses caused by hypothetical coal-fired power plants in specific locations in the United States, data for appropriate crop cultivars used in these locations would be obtained from the more extensive Heagle et al. data.

Factors that could significantly influence the response of crop vegetation to ozone in a southeastern U.S. location as compared to a southwestern U.S. location for a hypothetical power plant include both temperature and relative humidity. The temperature during plant growth and exposure has a significant influence on plant sensitivity to ozone. For most plants, foliar sensitivity increases with growth temperature up to approximately 30-32°C; at higher temperatures, the response is reversed (Shriner et al. 1990). Similarly, pollutant uptake and plant injury increase with relative humidity. It is difficult to differentiate between the influences of temperature, light (which can also influence pollutant uptake), and relative humidity under field conditions where they are interacting with soil moisture as well. Although each of these individual facts may influence plant response predictably, their combined effect on plant response under field conditions may be substantially less predictable (Guderian 1985).

Table 4. Crop-yield losses (percent) estimated to result from various ozone concentrations[a]

Crop (cultivar)	Mean ozone concentration (ppb) during growing season				
	40	50	60	70	80
Dry beans	4	9	15	23	31
Turnips	10	19	24	40	50
Peanuts	7	13	20	28	37
Lettuce	0	0	0	1	2
Tomato					
1981	1	3	5	8	12
1982	8	16	30	44	58
Sorghum	1	2	3	4	5
Tobacco					
7-h mean	3	6	9	13	18
12-h mean	7	12	17	23	28
Field corn	3	7	11	17	25
Winter wheat					
(Vona)	26	39	50	59	67
(Abe and Arthur)	3	6	9	14	20
(Roland)	9	16	23	31	38
Cotton					
(Acala SJ-2)	6	15	26	40	55
(Stoneville 213)	4	9	16	24	33
(McNair 235)	7	13	21	30	40
Tall fescue-Ladino clover	6	11	17	24	32
Alfalfa	6	9	13	17	20
Red clover-timothy	9	19	31	44	59
Soybean[b]	8	13	17	21	25

[a]Data selected from Heagle et al. (1988). Ozone concentrations are for 7-h or 12-h seasonal means.
[b]Representative values were selected from numerous reported experiments.

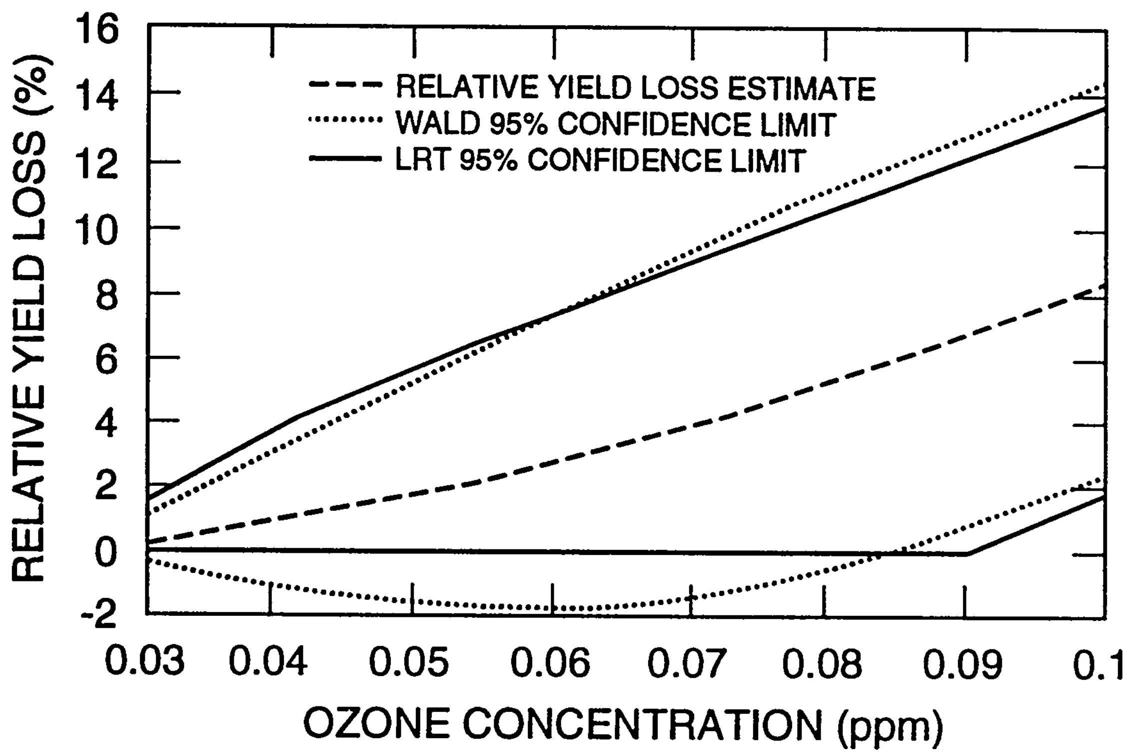

Fig. 8. Wald and likelihood ratio (LRT) 95% confidence interval estimates of relative yield losses (RYL) for sorghum using the Category I dose-response equation. Point estimates of RYL shown with lighter solid line. *Source:* Heagle et al.

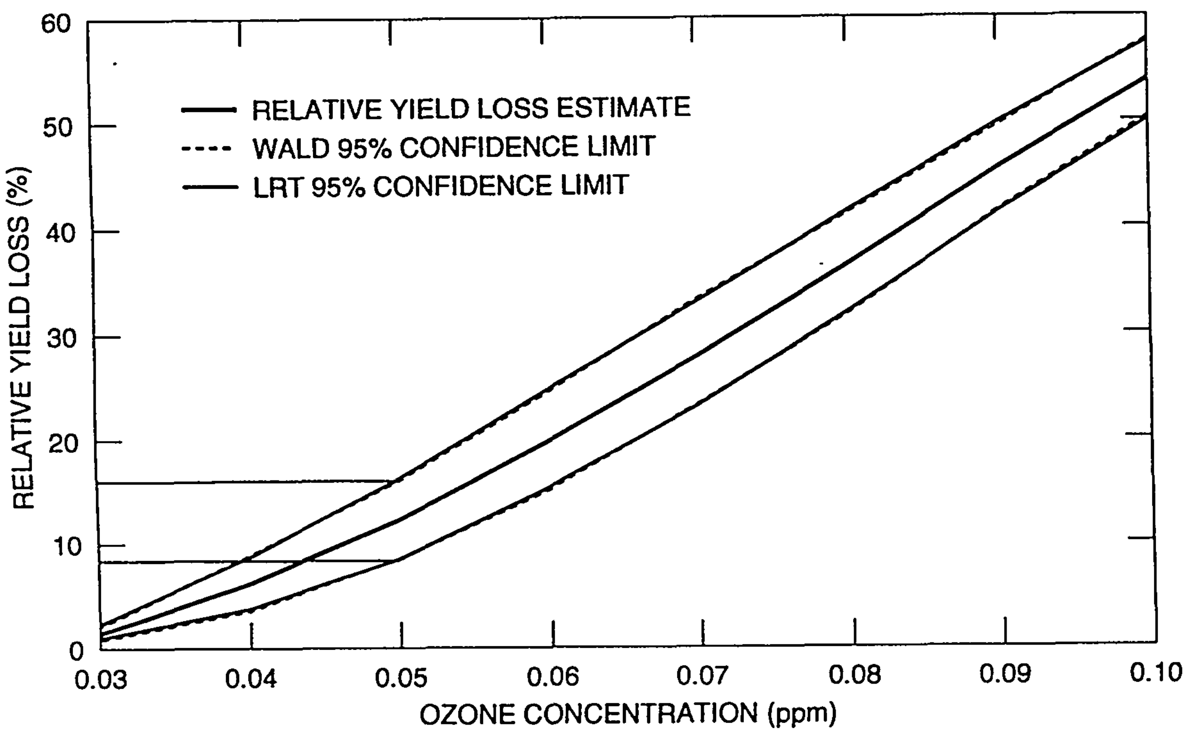

Fig. 9. Wald and likelihood ratio (LRT) 95% confidence interval estimates of relative yield losses (RYL) for peanut using the Category I dose-response equation. Point estimates of RYL shown with lighter solid line. Source: Heagle et al.

In the field, however, the generally more humid conditions of the southeastern United States permit plant injury by substantially lower ozone concentrations than those required to produce injury under the arid conditions of the southwestern United States (Heck et al. 1977).

Heagle et al. (1988) also addressed soil moisture and ambient SO_2 concentrations as factors that could interact with O_3 to produce cumulative or synergistic effects. Significant interactions were generally not found, although available data are not extensive. Also, a literature review by Mansfield and McCune (1988) was unable to make a useful conclusion about interactions between O_3 and SO_2. Table 5 summarizes the available information regarding the response of crop species to pollutant mixtures.

The dose-response data summarized by Heagle et al. (1988) can be combined with crop production data to produce estimates of the reduction in local crop production associated with an incremental increase in ozone exposure. Table 6 presents crop production data for Tennessee's District 6, which lies primarily east of a line from Chattanooga in Hamilton County to Jacksboro in Campbell County, and includes 9,640 square miles (Tennessee Department of Agriculture 1990). For all crops except hay, column 1 of the table presents the total planted acreage in region 6 and column 2 presents the fraction of that acreage planted within a 50 km of an assumed point source at the Clinch River Breeder Reactor site. Column 3 presents the average harvested yield per acre.

Hay production statistics in Tennessee are not broken down by district and county. For hay, column 1 presents the total planted acreage for the state of Tennessee in 1988 and column 2 presents the fraction of the total area of the state planted in hay. The estimated reduction in yield within a 50 km radius around a point source of ozone can be estimated from

$$\mathbf{R = S \times I \times A \times Y}$$

where **R** is the reduction in yield, **S** is the slope of the dose-response function, **I** is the increment in ozone concentration due to the source, **A** is the acreage calculated from Table 6, and **Y** is the yield per acre from Table 6. For illustration assume that emissions from a coal-fired generating plant result in an increase of 1 ppb in the average ground-level concentration of ozone within a 50 km radius

R = S x I x A x Y

where R is the reduction in yield, S is the slope of the dose-response function, I is the increment in ozone concentration due to the source, A is the acreage...and Y is the yield...

Table 5. Plant response to pollutant mixtures

Mixture	Test plant	Exposure information	Plant response to the mixture
$O_3 + SO_2$	Bean, snap	0.20 ppm (392 $\mu g/m^3$ O_3; 524 $\mu g/m^3$ SO_2) each gas; 7 hours/day, 4 days; varied salinity	Stomatal conductance: synergistic (variable); foliar injury: antagonistic; growth: additive; effect changed with salinity (Bytnerowicz and Taylor 1983)
	Bean, snap	0.15 ppm (294 $\mu g/m^3$ O_3; 393 $\mu g/m^3$ SO_2) each gas; 6 hours/day, 5 days, consecutive	Foliar injury: antagonistic; stomatal closure: synergistic, variable responses (Beckerson and Hofstra 1979)
	Bean, field	0.05-0.30 ppm (98-588 $\mu g/m^3$) O_3; 0.04 ppm (105 $\mu g/m^3$) SO_2; 4 hours	Net photosynthesis: additive or antagonistic, depending on O_3 concentration (Black et al. 1982)
	Tomato	0.2 ppm (392 $\mu g/m^3$) O_3, 0.2 or 0.8 ppm (2096 $\mu g/m^3$) SO_2; 3 or 4 hours, 15 times; infection with nematodes	Growth and yield additive at 0.2 ppm SO_2, antagonistic at 0.8 ppm SO_2 (Shew et al. 1982)
	Tomato	0.005 to 0.468 ppm (13 to 1226 $\mu g/m^3$) SO_2; 0.015 ppm or 0.056 ppm (29 to 110 $\mu g/m^3$) O_3; 5 hours/day, 5 days/week, 57 days; field study with open top chambers (Heggestad et al. 1986)	Ripe fruit decreased 16% by O_3 (low SO_2 treatment), 18% by SO_2 (low O_3 treatment); 32% in high O_3-high SO_2 treatment; additive response (Heggestad et al. 1986)

Table 5. (continued)

Mixture	Test plant	Exposure information	Plant response to the mixture
	Lettuce, radish	0.4 ppm (78 $\mu g/m^3$) O_3, 0.8 ppm (2096 $\mu g/m^3$) SO_2; 6 hours	Use of covariates increased precision for lettuce and radish; lettuce growth and injury effects antagonistic; radish was additive (Ormrod et al. 1983)
	Potato	Four O_3 concentrations, filtering of ambient O_3; 0.1 ppm (262 $\mu g/m^3$) SO_2; for 6 hours/day, 255 hours	Reductions in various growth and yield parameters were additive (Foster et al. 1983)
	Soybean, 2 cultivars	0.25-1.0 ppm (490-1960 $\mu g/m^3$) O_3, 0.50-1.5 ppm (1370-3930 $\mu g/m^3$) SO_2; 0.75, 1.5 or 3 hours	Foliar injury and reduced shoot fresh weight: additive, antagonistic, or synergistic, depending upon concentration and time (Heagle and Johnston 1979)
	Soybean	0.06 or 0.08 ppm (118-157 $\mu g/m^3$) O_3, 0.06 or 0.11 ppm (157-288 $\mu g/m^3$) SO_2; 5 hours/day, 16 days; in open field facility	Both O_3 and SO_2 caused decreases in a number of yield measures; mixture responses were additive (Reich and Amundson 1984)
	Soybean	0.20 ppm (392 $\mu g/m^3$) O_3, 0.70 ppm (1834 $\mu g/m^3$) SO_2; various combinations of individual and both gases over 2 hours of exposure (controlled chambers)	No effect on carbon exchange rate (CER) by individual gases; O_3 or SO_2 for 1 hour followed by mixture for 1 hour gave a 38 and 59% reduction

Table 5. (continued)

Mixture	Test plant	Exposure information	Plant response to the mixture
			in CER; mixture gave 67% reduction; suggests not a stomatal response; synergistic (Chevone and Yang 1985)
	Soybean	0.04 to 0.08 mean ppm (78-156 $\mu g/m^3$) O_3, 0.00 to 0.11 mean ppm (0-288 $\mu g/m^3$) SO_2; 5 hours/day, 16 days, during pod fill; linear gradient system in field	Ozone caused 26% seed yield reduction; SO_2 a 6% reduction with no significant interactions (Reich and Amundson 1984)
	Grape, 2 cultivars	0.20 and 0.40 ppm (39-78 $\mu g/m^3$) O_3, 0.15 and 0.30 ppm (393-786 $\mu g/m^3$) SO_2; 4 hours	Foliar injury and reduced shoot length were antagonistic to synergistic,depending on concentration; leaf abscission was synergistic (Shertz et al. 1980)
$O_3 + NO_2$	Four species - bean, mint, radish, wheat	0.08 to 0.10 ppm (150-188 $\mu g/m^3$) NO_2 for 3 hours prior to 0.08 to 0.10 ppm (156-196 $\mu g/m^3$) O_3 for 6 hours; greenhouse exposures	NO_2 increased growth in radish and bean; O_3 reduced growth in wheat and bean; apparent synergism from mixture in radish and wheat, additive to antagonistic in bean and no effect in mint (Runeckles and Palmer 1987)
$SO_2 + NO_2$	Bean, snap	0.1 ppm (262 $\mu g/m^3$ SO^2; 188 $\mu g/m^3$ NO_2) of each gas; 5 days	Transpiration: increased by individual gases; decreased by combination (Ashenden 1979)

Table 5. (continued)

Mixture	Test plant	Exposure information	Plant response to the mixture
	Tomato	0.11 and 0.05 ppm each gas; continuous, 14 and 28 days NO_2: (94-207 $\mu g/m^3$) SO_2: (131-288 $\mu g/m^3$)	No effects of individual gases; mixture caused decrease in leaf fresh weight and area (14 days), root fresh weight, and dry weight (28 days) (Marie and Ormrod 1984)
	Grass, Poa pratensis	0.10 ppm of each gas; 104 hours/ week, long-term NO_2: (188 $\mu g/m^3$) SO_2: (262 $\mu g/m^3$)	Reduced growth of roots and shoots: synergistic in late winter, but not later; shoot recovery during summer; effects effects on flowering were additive (Whitmore and Mansfield 1983)
	Alfalfa, 12 strains	0.08 ppm (210 $\mu g/m^3$) SO_2, 0.120 ppm (226 $\mu g/m^3$) mean NO_2 (singly and as mixture); 15 days; greenhouse type chambers	Strains showed from +8 to -35% effect from SO_2; from +13 to -27% effect from NO_2; from -8 to -50% effect from mixture; mixture gave from additive to synergistic responses on different strains (Lorenzini et al. 1985)
	Soybean	0.2, 0.4, 0.6 ppm of each gas; 2 hours NO_2: (376, 752, 1128 $\mu g/m^3$ SO_2: (524, 1048, 1572 $\mu g/m^3$)	Photosynthesis: synergistic; stomatal conductance: synergistic; respiration: additive (Carlson 1983)

Table 5. (continued)

Mixture	Test plant	Exposure information	Plant response to the mixture
	Soybean	0.13-0.42 ppm (341-1100 $\mu g/m^3$) SO_2; 0.06-0.40 ppm (157-1048 $\mu g/m^3$)NO_2; 3 hours/exposure, 10 days during pod fill (2 year study, ambient air with O_3)	Chlorophyll reduction, synergistic; yield reduced from 9 to 25% in 2 years, synergistic (Irving and Miller 1984)
	Soybean	0.13 to 0.42 ppm (341-1100 $\mu g/m^3$) mean SO_2; 0.06 to 0.40 ppm (157-1048 $\mu g/m^3$) mean NO_2; 3 hours/exposure, 10 times during pod fill; two year study in field open-release plots	Yield not affected by NO_2 in either year; 6% reduction from SO_2 in second year; mixtures reduced yield from 9 to 25%; response was synergistic (Irving and Miller 1984)
	Soybean	0, 0.2, 0.3 ppm (0, 524, 786 $\mu g/m^3$) SO_2; 0, 0.1, 0.2 ppm (0, 188, 376 $\mu g/m^3$) NO_2; 3 hours/day, every other day, 15 exposures; greenhouse chambers	Leaf weight was increased by 2% and 9% SO_2 and NO_2, respectively; low combination reduced leaf weight by 7%, high combination by 16%; root weights were -10% and +2% for SO_2 and NO_2, and -31% and -38% for low and high combinations. Probably synergistic (Klarer et al. 1984)
O_3 + PAN	Petunia; bean, kidney	0.10-0.40 ppm (196-784 $\mu g/m^3$) O_3; 0.01-0.10 ppm (49-495 $\mu g/m^3$) PAN; 4 hours	Foliar injury, antagonism to synergism depending on concentration of pollutants (Nouchi et al. 1984)

Table 5. (continued)

Mixture	Test plant	Exposure information	Plant response to the mixture
SO_2 + HF	Corn, sweet	0.09 ppm (236 $\mu g/m^3$) SO_2; 0.55 ppb (0.45 $\mu g/m^3$) HF; 32 days (continuously)	No effect from HF or SO_2; mixture reduced fresh and dry weight of stalk and reduced yield, reduced accumulation of foliar F; interaction not tested (Mandl et al. 1980)
NO_2 + HF	Corn, sweet	0.6 and 1.2 ppm (1128 and 2256 $\mu g/m^3$) NO^2, 6 hours/day, 4- to 5-days/week; 0.6 and 1.9 ppb (0.49-1.55 $\mu g/m^3$) HF (continuous)	Foliar injury, additive or antagonistic; interaction shown for increased stomatal resistance (Amundson et al. 1982)
O_3 + SO_2 + NO_2	Radish	0.1, 0.2, 0.4 ppm of each gas; 3 hours NO_2: (188, 376, 752 $\mu g/m^3$) SO_2: (262, 524, 1048 $\mu g/m^3$) O_3: (196, 392, 784 $\mu g/m^3$)	Reduced yield: no three-way interaction but an NO_2 x O_3 and an NO_2 x SO_2 interaction; interactions appear synergistic (Reinert et al. 1982b)
	Radish, Marigold	0.3 ppm each gas; 3 hours, 9 times NO_2: (564 $\mu g/m^3$) SO_2: (786 $\mu g/m^3$) O_3: (588 $\mu g/m^3$)	Reduced yield: primarily additive responses; NO_2 x SO_2 and O_3 x SO_2 interactions antagonistic (Reinert and Sanders 1982)
	Turfgrass, 6 species, 18 cultivars	0.15 ppm (294 $\mu g/m^3$) O_3 for 6 hours/day; 0.15 ppm (393 $\mu g/m^3$) SO_2 and 0.15 ppm (282 $\mu g/m^3$) NO_2 continuously; 10 days	Foliar injury and reduced leaf area, primarily additive with some antagonism (Elkiey and Ormrod 1980)

Table 5. (continued)

Mixture	Test plant	Exposure information	Plant response to the mixture
	Sunflower	0.2 ppm each gas, 2 hours NO_2: (376 $\mu g/m^3$) SO_2: (524 $\mu g/m^3$) O_3: (392 $\mu g/m^3$)	Reduced net photosynthesis: NO_2 x O_3 and SO_2 x O_3, synergistic response; three way similar to two-way responses (Furukawa and Totsuka 1979)
O_3 + Acidic Rain	Soybean	O_3 - CF, NF, ambient; simulated rain pH 2.8, 3.4, 40	Ozone reduced a stimulatory effect of rain acidity (Troiana et al. 1983)
	Soybean, red clover, winter wheat	0.04-0.07 ppm (78-137 $\mu g/m^3$) O_3 simulated rain pH 3.0-5.6	No effect of acidic rain as low as pH 3.0 on the ozone response of test plants (Reich and Amundson 1985)
	Radish	Rain - pH 4.0, 3.3, 3.0; ozone - 0.1, 0.2, 0.4 ppm (196, 392, 784 $\mu g/m^3$) 3 h/week, 3 weeks	No effect of acidic rain; no interaction between O_3 and acidity (Johnston et al. 1986)
SO_2 + Acidic Rain	Soybean	Rain: pH 3.1 and pH 5.3 SO_2: year 1, 4 h/event, 24 events, mean SO_2 of 79 ppm (2070 $\mu g/m^3$); year 2, 4 h/event, 17 events, mean SO_2 of 0.19 ppm (498 $\mu g/m^3$)	No significant interaction on yield (Irving and Miller 1981)
SO_2 + O_3 + Acidic Rain	Soybean	Rain - pH 2.6 vs pH 5.6; 0.10 ppm (196 $\mu g/m^3$) O_3 and 0.20 ppm (524 $\mu g/m^3$) SO_2	No interactive effects of the pollutant treatments on growth (Norby and Luxmoore 1983)

From Shriner et al. (1990) as adapted from Heck et al. (1986).

Table 6. Crop production data for Tennessee's District 6, which lies primarily east of a line from Chattanooga in Hamilton County to Jacksboro in Campbell County, and includes 9,640 mile^2

Crop	Acres planted (1988)	Fraction of district[a]	Yield per acre (average for 1988)
Soybeans	23,000	0.0037	24 bushels
Tobacco	24,657	0.0040	1,760 lb
Wheat	51,000	0.0083	40 bushels
Corn	81,000	0.0131	70 bushels
Sorghum	3,300	0.0005	54 bushels
Hay (statewide)			
Alfalfa	110,000	0.0041[b]	2.6 tons
Other	1,500,000	0.0555[b]	1.7 tons

[a] Estimated fraction of planted acreage falling within a 50-km radius of the reference site. These fractions are used to estimate the number of acres of the respective crop in the area of 50-km radius being considered for ozone impacts.
[b] Statewide acreage of hay divided by the total area of the state, which is 42,244 mile^2.

of the plant. Then the estimated reduction in corn production (bushels) associated with that increment would be

$$\mathbf{R = 0.004 \times 1.0 \times 1{,}061 \times 70 = 297\ bu}$$

If the increase took place over all of Tennessee District 6, then the estimated reduction in corn production would be 22,680 bu.

5.4.3 Impacts of Acid Rain on Crops

Research studies of the impacts of acid rain on crops have generally found no significant effects on crop yield.

Research studies of the impacts of acid rain on crops have generally found no significant effects on crop yield. The results of these studies, as thoroughly reviewed by Shriner et al. (1990), are summarized in Table 7. No consistent yield reductions were found in crops exposed to levels of acid rain representing average ambient levels (pH 4.1 to 5.1) or rain events with relatively high acidity (pH 3.0 to 4.0) in the eastern United States (see also Fig. 10). In addition, many studies investigated the effects of acidic fog; the yield of crop plants was reduced only when fog pH was 2.0 or less. However, in some cases the market value of the

Table 7. Summary of research results: Impacts of acid rain on crop yield

Crop (cultivar)	Results (references)
Radish	Greenhouse radishes indicated a threshold for significant yield reductions between pH of 3.0 and 3.4 (Irving 1985; Jacobson et al. 1988).[a]
Radish	Field-grown radishes in four studies showed no negative effects from rain acidity levels as low as pH 3.0 (Irving 1985).
Corn (Pioneer 3377, B73 x M017)	No effect at pH 3.0 to 4.6 for 2 cultivars grown for 3 years (Banwart 1987). No effect was seen with 2 cultivars at pH 3.0 and 3 different total rainfall amounts, except yield was reduced in 1 of the 2 exposed cultivars in only 1 of 3 years. (Banwart 1986).
Corn (FS 854)	No effect was seen at pH 3.0 at three rainfall amounts (Banwart et al. 1990).
Soybean (Williams)	No effect was seen in the Williams cultivar at pH 2.7 to 4.1 (Evans et al. 1984, 1985).
Soybean (Amsoy, Asgrow, Corsoy, Hobbit)	Results were inconclusive, although yield reductions appeared to occur frequently. Yields were sometimes reduced at the highest acidity levels, but sometimes reductions occurred at medium acidity levels rather than at the highest levels (Evans and Curry 1979; Evans et al. 1984, 1985, 1986).
Soybean (Amsoy 71)	Four studies found no effect (Banwart 1984, 1985, 1986; Irving et al. 1986; Miller and Irving 1984), but a fifth showed reduced yield from pH of 3.0 to 4.2 (Banwart 1987).
Soybean (Davis, Forrest, Hodgson)	Five studies found no effects (Heagle et al. 1983; Johnston and Shriner 1986; Dubay et al. 1984; Irving 1986b).
Hay	One study found no effect (Irving et al. 1984).
Wheat	One study found no effect on foliage biomass (Shriner and Johnston 1984).
Alfalfa	One study found no effect from treatments of pH 3.2 and 4.4 (Temple et al. 1987).

Table 7. (continued)

Crop (cultivar)	Results (references)
Tobacco	One study found no effect on yield at pH from 2.0 to 4.0, but quality was reduced at pH 2.0 (Rather and Frink 1984).
Potato	One study for 2 years found no effect at pH 2.8 to 4.6 (Pell et al. 1986).
Oat	One study found no effect at pH 3.0 to 4.8 (Pell and Puente 1987).
Snap bean	One two-year study found no reduction in yield, but found an increase in yield at pH 2.6 and 3.4 during the second year (Troiano et al. 1984).

[a]The references given in this table were not consulted first hand but were cited by Shriner et al. (1990), from which the information in this table was obtained. Please see Shriner et al. to obtain the complete reference citations.

yield of fruit or foliage was reduced as a result of visible injury from pH levels up to 2.6 (Shriner et al. 1990). The levels of acidic deposition required to impact crop yield are for the most part between 10- and 100-fold greater than average ambient levels of acidity. For this reason, the increased increment of deposition resulting from the siting of a hypothetical new power plant using best available control technology is not anticipated to result in any reduction in crop yields associated with increased acid rain.

Although extensive quantitative estimates of the relative importance of atmospherically-derived sulfur to meeting plant nutritional requirements are not available, several studies provide some insight in this regard. Results of experiments with simulated acid rain exposures to a forage mix (timothy, red clover) suggested that these species might benefit from levels of sulfur and nitrogen increased above ambient levels in rain (Irving 1986). Noggle (1980) estimated that soybeans growing at various distances from sources of atmospheric sulfur obtained between 10 and 50% of their sulfur requirement from the atmosphere. Jones and Suarez (1980) reported increased yield of corn grain and silage with 9 and 18 kg/ha of sulfur added in fertilizer trails. Atmospheric sulfur deposition at the sites was approximately 11 kg/ha per year. The authors concluded that the probability was low that plant health was being influenced by either too much or too little atmospheric or soil sulfur. In their South Carolina studies, only one crop (corn) out of eight studied, and one soil (a loamy sand) out of five studied, showed positive responses to sulfur additions. At none of the 15 locations studied was there an indication of too much atmospheric sulfur for healthy plant growth.

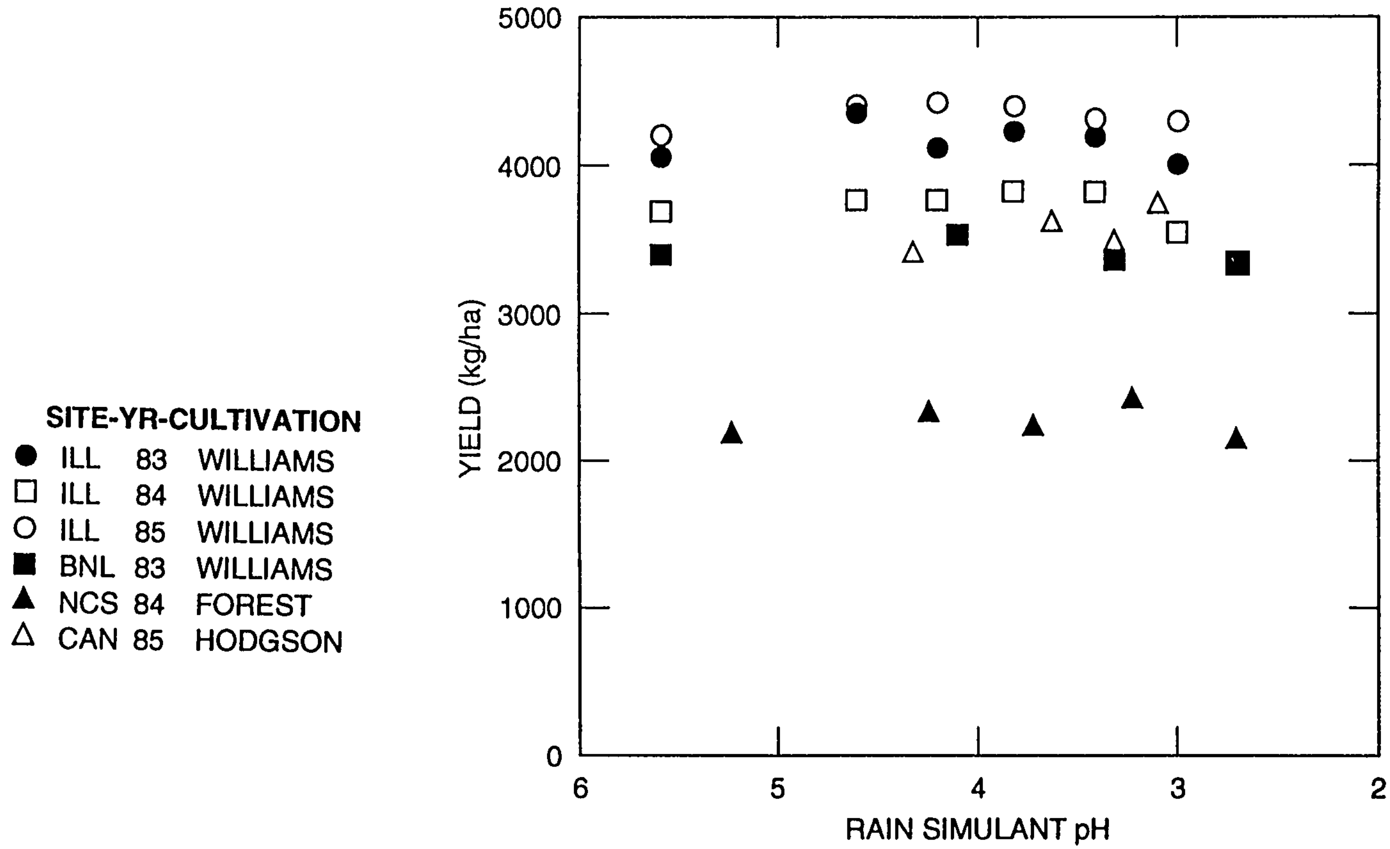

Fig. 10. Yield of field-grown "Williams," "Forrest," and "Hodgson" soybeans as a function of simulated rain acidity. Differences among means for a particular hybrid and year are not statistically significant (Banwart 1984, 1985; DuBay et al. 1984; Evans et al.; Kuga 1986). *Source*: NAPAP 1987.

5.4.4 Impacts of Air Pollution on Trees

The following sections are quoted directly from NAPAP 1990:

"Forests and crops are exposed to a complex mixture of pollutants through a variety of pathways. Wet and dry deposition of pollutants follow separate pathways of interaction with plants. Wet deposition is primarily a foliage surface phenomenon involving the interception of rain, clouds, or fog by plant canopies, whereas dry deposition of gases affects plants primarily by the uptake of the pollutant within the interior of the leaf where physiological processes such as photosynthesis can be affected. Wet and dry deposition can also affect plants by modifying soil chemistry and, as a result, plant nutrient status, symbiotic relationships, and other functions associated with the root system. Because ozone is the only gaseous air pollutant for which potentially phytoxic concentrations occur consistently across large geographic regions and because many of those geographic regions coincide with regions where acidic deposition is also potentially of concern to forests and crops, ozone is highlighted as an important stress contributing to the overall condition of terrestrial ecosystems which must be included in an overall assessment of acidic deposition.

"Single short-term exposures of plants to an air pollutant rarely cause problems, unless extensive tissue death (acute injury) occurs. More important is the long-term accumulated impact of multiple exposures and multiple pollutants over a growing season or several growing seasons. For this reason, perennial plants (such as forest trees or pasture and orchard crops) are at greater risk of long-term air pollution injury than are annual plants (such as most agricultural crops).

"Plants may respond differently to acute (short-term, high-concentration) and chronic (long-term, low-concentration) exposures to gaseous pollutants depending on the ability of the plant to assimilate or detoxify the pollutant. For purposes of this assessment, long-term repeated exposure to a pollutant or pollutants at low concentrations is of greatest interest. Acute exposures to pollutants at high concentrations near point sources (such as a smelter, or other individual industrial source) are of less concern because at the regional scale, pollutant concentrations in the ambient atmosphere in North America rarely reach levels capable of causing acute injury.

"Chronic exposures result in assimilation of the pollutant by the plant and in either subsequent toxic effects (e.g., damage to membranes) or metabolism of the compound. In either case, physiological processes within the plant are typically altered. Natural stresses, such as drought, insects, disease-causing organisms, competition, and temperature extremes depress the baseline health of vegetation communities. Levels of air pollutant stress are superimposed upon these natural stress factors. In some cases, one or more natural stress factors may either exacerbate or dampen plant response to periodic natural stress events. In general, altered physiological processes lead to the inability of the plant to respond to existing environmental conditions or to additional stress. In perennial vegetation, this accumulation of subtle responses may not become detectable for many years.

"As acidic deposition contacts the forest canopy, a series of interactions occurs between the deposition components (primary ions of concern are hydrogen, sulfate, and nitrate ions) and the leaves, stems, and branches of trees, the litter layer on the forest floor, and the soils and surface which acidic deposition influences the physiological processes or growth of individual forest trees is determined by a complex set of interacting factors, including the inherent sensitivity of the tree species, the growth stage of the vegetation, the chemical composition of the precipitation (i.e., the relative contributions of the sulfate, nitrate, and ammonium ions), the timing, duration, and concentration of exposure, and the duration of respite between exposures. Soil factors (cation and anion nutrient status, pH, sulfate adsorption capacity) are also potential modifiers of plant response, depending on the specific mechanisms of action which are involved. Cation-exchange reactions between precipitation and the forest canopy (leaves) may temporarily neutralize incoming precipitation acidity by exchanging hydrogen ions for basic cations (calcium [Ca], magnesium [Mg], potassium [K]) at the leaf surface. However, this process of cation removal in throughfall (the precipitation that washes over foliage and falls to the forest floor) results in a demand for uptake of additional cations from the soil to replace them (an acidifying process in the soil whereby the plant gives off hydrogen ions in exchange for nutrient cations), and as a consequence, causes a net acidification of the system (Fig. 11). Similarly, uptake of ammonium ion by vegetation is also an acidifying process. Hydrogen ions in the rooting zone may in turn leach essential cation nutrients through the soil into surface waters (making them unavailable for plant uptake) or mobilize metals such as aluminum (Al) or iron (Fe) into the soil solution, which may compete with nutrient cations for uptake at the root surface or which may be toxic to the physiologically active fine root system of the tree.

"Figure 12 illustrates the complex nature of the interactions between wet and dry deposition of pollutants and a forest canopy. Incoming precipitation either washes off the canopy surfaces, is taken up by the foliage, or evaporates from the surface. Ions in solution which are absorbed by the foliage may be translocated within the plant and metabolized, stored, or subsequently lost through leaching. Ions reaching the forest floor may be taken up by vegetation, or lost to ground and surface waters through soil leaching processes. The processes of litterfall and root turnover return plant organic matter to the soil where decay of organic compounds releases mineral nutrients to the soil. In each of the above transfers, the chemistry

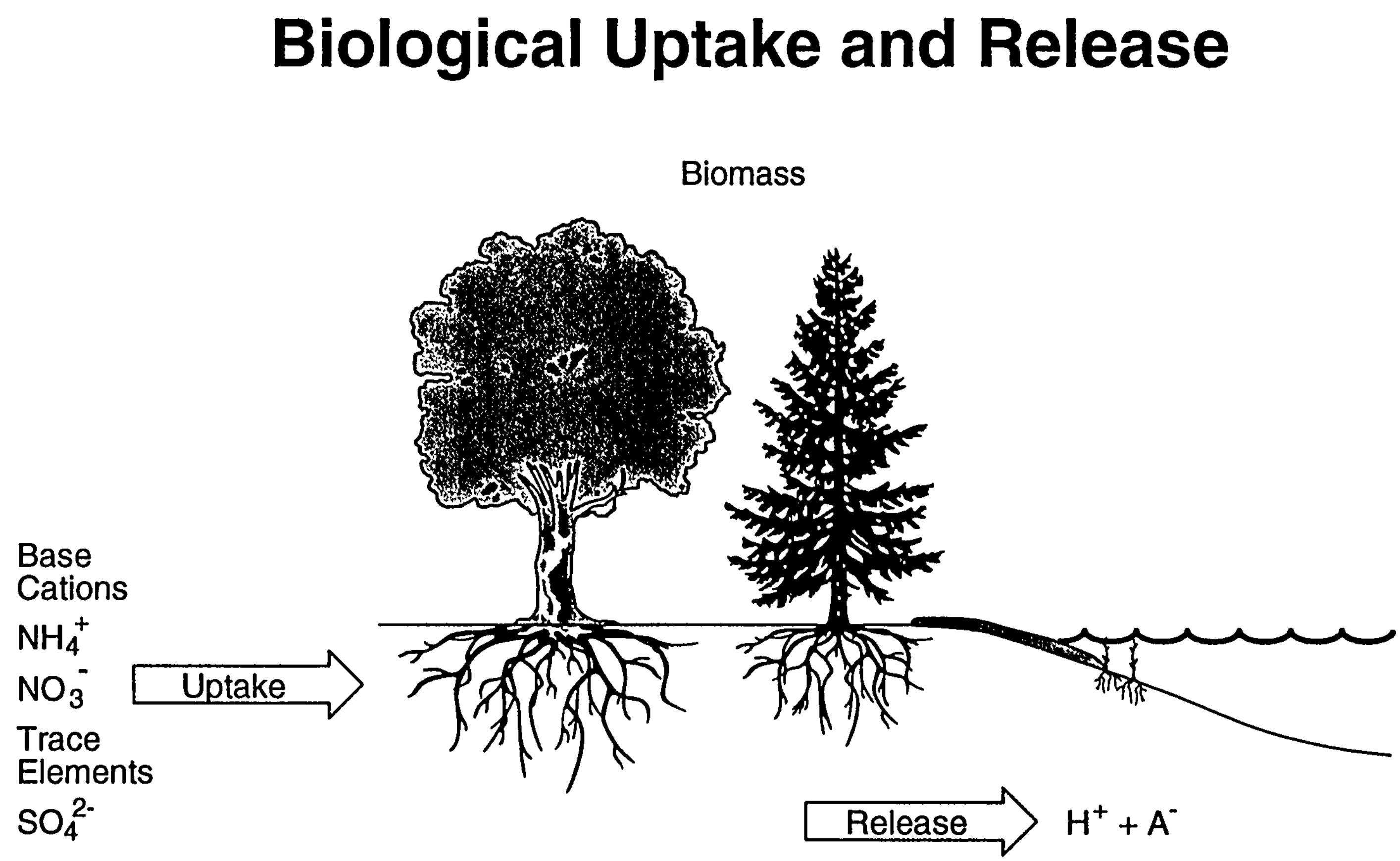

Fig. 11. Biological uptake of nutrients is an acidifying process in soils.

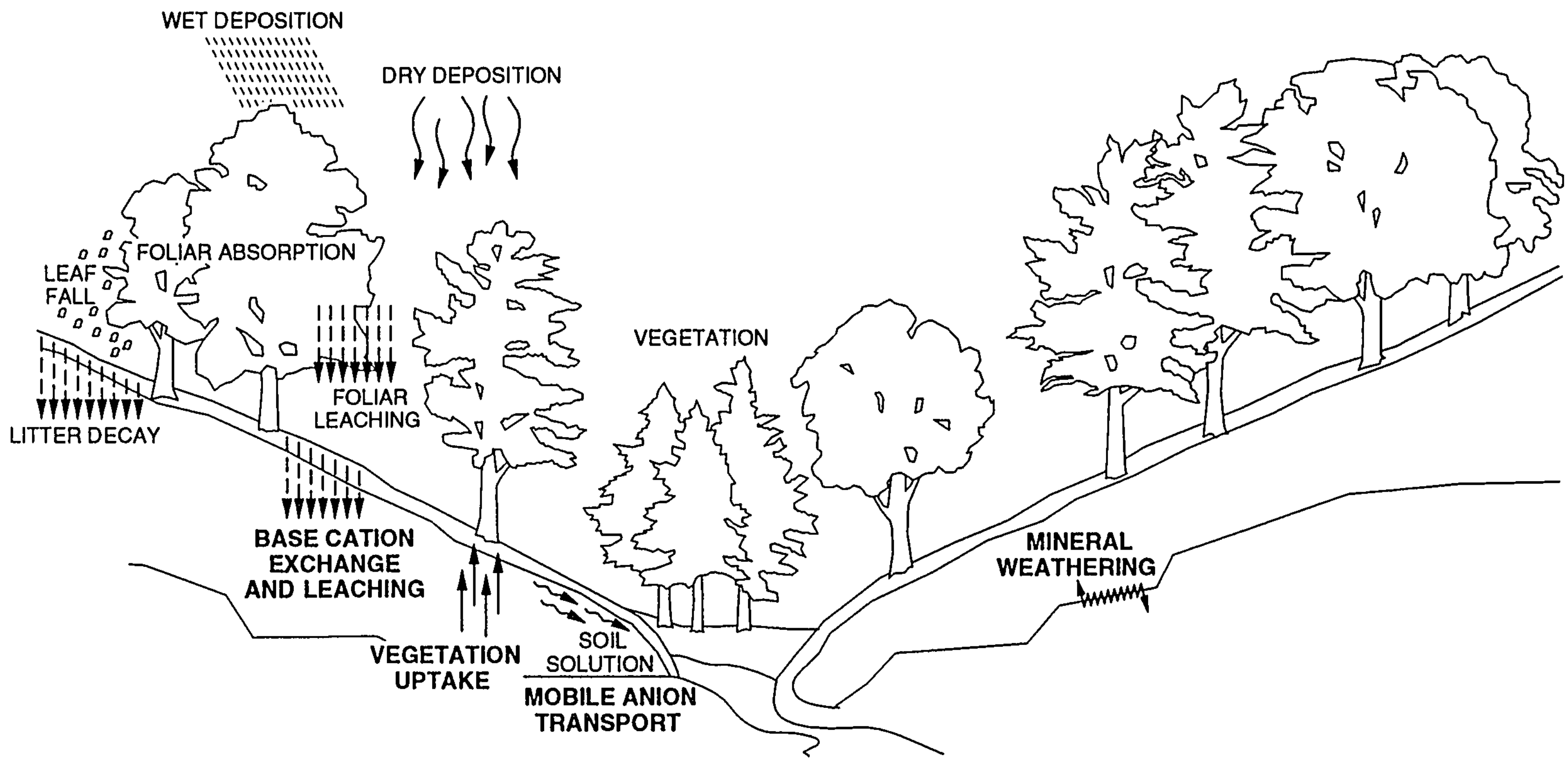

Fig. 12. Schematic representation of the interaction of wet and dry atmospheric deposition with components of the forest ecosystem.

of the solution moving through the system influences process rates and magnitudes. These types of modifications of nutrient cycling processes are an important pathway by which ambient levels of acidic deposition impact forest trees.

"Because of these ambient levels are typically too low to cause direct damage to physiologically active tissue, the subsequent loss of growth or other plant functions by shifting allocation of photosynthetic products from growth to increased respiration and repair of damaged tissues is a less important pathway of acidic deposition effect on forest trees.

"The most important form of dry deposition for regional-scale forest effects is that of the photochemical oxidant, ozone...

"The most important form of dry deposition for regional-scale forest effects is that of the photochemical oxidant, ozone. Ozone is considered a secondary pollutant because it is formed in the atmosphere as the result of photochemical reactions between the primary (emitted) pollutants in the general class of volatile organic compounds and oxides of nitrogen.

"A primary physiological effect of ozone on plants is to disrupt carbon allocation pathways. Reduced photosynthesis, increased respiration, and reduced allocation of photosynthate to root growth have all been shown to occur in response to ambient ozone exposure in sensitive species.

"Forests and their ecosystems are constantly changing. Change occurs on many spatial and temporal scales through natural and anthropogenic (generated by human activity) disturbance and through the processes of succession and biomass accumulation. Tree and stand growth rates, mortality rates, and all other forest attributes are subject to change.

"Changes in forest health such as those associated with insects, pathogens, climatic stresses, and normal maturation and senescence (aging) of trees and stands are natural features of forest ecosystems. Such changes may develop gradually or suddenly, depending on environmental conditions and the characteristics of the species involved. Sudden (over a period of several years) dieback is a normal phenomenon in some forest stands and may be confused with the dieback of stands caused by external stress factors such as acidic deposition.

"Forest health changes not attributable to normal stand development may be related to natural and/or anthropogenic factors, including acidic deposition and ozone. Such forest health problems are referred to as "declines" if they are (a) not clearly attributable to a single natural factor and (b) sufficiently severe or extensive as to be detected by routine forest monitoring or other observational means. Symptoms of change in forest health may include loss and discoloration of foliage, leading to reduced growth, and increased rates of branch and whole tree mortality. Some or all of these symptoms may be present, and they may be associated with a variety of natural and anthropogenic factors.

"Several recent reports of changes in forests of North America have received considerable attention in the debate over the ecological effects of acidic deposition and associated air pollutants. The geographic distribution and extent of forests in major regions of the United States are summarized in Table 8. Conclusions regarding the current status of forests are summarized in the following section."

Table 8. Area of timberland in the United States by forest type, 1987

Forest type	Total area (million hectares)	Proportion (%)
Eastern conifer forests		
Eastern spruce-fir forests		
High elevation	0.2	0.1
Low elevation	6.6	3.5
Southern pine forests	26.0	13.7
Other softwoods	5.6	3.0
Total	38.4	20.3
Eastern hardwood forests	104.7	55.3
Western conifer forests (all species)	46.1	24.4
Total, all forest types	189.2	100

5.4.4.1 Eastern Spruce-Fir Forests

The following is quoted directly from NAPAP 1990:

"Surveys conducted throughout the Appalachian Mountains indicate that many high-elevation spruce-fir stands are exhibiting high rates of mortality, excessive foliage loss, and reduced growth. Damage to red spruce in the northern Appalachians increases with increasing elevation and does not appear to be correlated with stand basal area or tree age. Analyses of forest history and weather records show, however, that widespread episodic mortality of spruce has occurred before in the northern Appalachians and that water and temperature stresses may have been responsible for those earlier periods of mortality. Tree seedlings, especially fir, are usually abundant and apparently healthy wherever mortality in the overstory trees allows direct sunlight to reach the forest floor, suggesting that seedlings are not limited in their requirements for growth. No single factor accounts for recent changes in spruce-fir forests at high elevations. Air quality and cloud chemistry monitoring data suggest that ozone and cloud water acidity can reach potentially harmful levels. Experimental evidence suggests that acidic deposition can increase the susceptibility of red spruce trees to winter injury at high elevations in the northern Appalachians."

"High-elevation red spruce are generally in better condition in the southern Appalachians than in the norther Appalachians with regard to mortality but are experiencing some growth reduction. In the southern Appalachians, most of the numerous dead trees at high elevations are Fraser fir killed by an insect called the balsam wooly adelgid."

"At low elevations in upstate New York and norther New England, radial growth rates of red spruce and balsam fir trees have decreased during recent decades. The decreases, however, are within the range of differences expected due to changes in historical land use patterns, insect outbreaks, and normal stand development processes. Overall, average growth rates of eastern white pine, northern red oak, sugar maple, and eastern hemlock are increasing in these same areas."

5.4.4.2 Southern Commercial Forests

The following is quoted directly from NAPAP 1990:

"The U.S. Forest Service has reported reductions in tree and stand growth rates of natural pine stands in the southeast. The reductions generally began in the 1950s and 1960s and may be at least partially attributable to changes in land use patterns, increases in age and competition, and other natural factors. Unfortunately, available data are not adequate to determine whether the growth reductions are greater or less than would be expected in the absence of acidic deposition and associated pollutants. Widespread growth rate reductions have not been found on intensively managed forest industry lands."

"Whether acidic deposition or other air quality factors are significantly reducing the health and productivity of southern pines remains to be determined. Ambient levels of ozone have altered seedling growth and physiology of some southern pine families in controlled experiments. In contrast, effects of acidic deposition on pines in controlled experiments are mixed but generally without significant negative growth response, and some positive growth responses were noted."

"Modeling studies suggest that acidic deposition could have measurable effects on the chemistry of some southern forest soils within 50 years. Whether such changes will (a) actually occur and (b) affect pine growth remains to be determined. Beneficial effects on soil fertility due to nitrogen and sulfur deposition may offset adverse effects due to accelerated leaching of base cations, in the short term, followed by long-term reductions in nutrient status of the soils."

5.4.4.3 Eastern Hardwood Forests

The following is quoted directly from NAPAP 1990:

"Sugar maple is the only major eastern hardwood forest species for which atmospheric deposition has been hypothesized to be linked to regional decline. Sugar maple decline (e.g., crown thinning, branch dieback) has recently affected many stands in southern Quebec and scattered trees and stands in the other eastern Canadian provinces and northeastern United States. Symptoms of decline are similar to those reported intermittently since the early 1900s. The number of reports of declining stands has increased in recent years, but is not clear whether this reflects a real increase in the incidence of decline or an increase in public awareness of forest conditions. Future long-term monitoring of forest condition will draw upon baseline measurements made as the result of recent interest in acidic deposition to evaluate this important question."

"Maple decline has generated greatest concern in Quebec. Many stands are affected, especially in the region south of the St. Lawrence River. Insect defoliation and nutrient deficiency are important contributors to the decline. Many soils in the region are naturally low in potassium. Some of these soils are also low in phosphorus. Some soils in the region north of the St. Lawrence River are magnesium deficient. Unusual weather-related stresses during the early 1980s may also have contributed to decline. It has been hypothesized that acidic deposition has contributed to decline by accelerating leaching of basic cations from soils and foliage. However, nutrients other than potassium are not deficient in most declining stands in Quebec. In those limited areas where nutrient deficiency symptoms already occur, acidic deposition could further exacerbate their expression."

"Preliminary results of a joint U.S.-Canadian sugar maple monitoring project suggest that sugar maple decline is widely scattered outside of Quebec and is not spatially correlated with acidic deposition patterns. Analyses of increment cores taken from sugar maple and red maple trees throughout the northeastern United States in the late 1970s and early 1980s found no evidence of a decline in average radial growth rate."

"Ambient levels of rainfall acidity and soil solution aluminum have not been shown to reduce growth or damage tissues in controlled experiments with sugar maple seedlings. Field studies have not found symptoms or correlations which provide substantial support for hypothesized linkages between sugar maple decline and acidic deposition. The most recent Forest Survey statistics show that growing stock volumes for sugar maple and red maple have been increasing in New York and Maine."

"Absence of a simple spatial correlation between deposition and decline is, by itself, inadequate evidence to discount the importance of acidic deposition. The decline phenomenon is thought to be the result of several increasing stresses, the spatial pattern of which may mask a relationship between the pattern of deposition and decline. There is currently inadequate information available to rigorously test the relationship between these multiple stress factors and sugar maple decline. As a result, the available information does not rule out hypothesized subtle effects of acidic deposition and ozone which might predispose sugar maples to damage from natural stresses. Such hypothesized effects merit further investigation."

"Results of investigations in other eastern hardwood forests suggest that atmospheric deposition of sulfur has altered nutrient cycling in northern hardwood forest stands from Minnesota to Michigan, had reduced lichen species richness at the high end of a deposition gradient across red maple stands in western Pennsylvania, and is associated with higher rates of mortality in black oak across similar gradient from northwest Arkansas to southern Ohio. Changes in overall productivity to these forests is not apparent at the present time."

5.4.4.4 Western Conifer Forests

The following is quoted directly from NAPAP 1990:

"Ozone stress is the predominant factor in a decline of ponderosa and Jeffrey pines in the San Bernardino Mountains near Los Angeles and the central Sierra Nevada of California. The impacts of ozone on forests in California range from slight to severe. Ozone damage is relatively difficult to detect, but the cumulative effects are evident by higher mortality rates due to insect (such as bark beetles) infestation, especially during periods of drought stress. Ozone and/or drought stress can make trees more susceptible to bark beetle attack. Mortality is relatively low over a short term. Over an 8-year period, the mortality of trees with ozone injury symptoms was 1% in the central and southern Sierra Nevada. In contrast, major insect and disease pests cause rapid and, easily detectable damage. Ozone in combination with other environmental stresses such as drought, insects, and/or disease leads to the most severe problems. This combination of long-term and intermittent stresses can not only reduce productivity, but has also resulted in long-term changes in community composition in areas such as the San Bernardino Mountains."

5.4.5 Other Major Air Pollutants of the Coal Fuel Cycle

Gaseous sulfur and nitrogen oxides have dual potentials as both nutrient sources as well as toxic agents. As a result, plants and plant communities are influenced in a variety of ways depending on concentrations, exposure dynamics, and relative species' sensitivities. Separation of nutrient and toxic responses is difficult, because processes impacted positively by enhanced sulfur or nitrogen nutrition during a growing season be negatively impacted by the same concentration over a prolonged exposure period (e.g., decades).

Although historical examples of effects of high levels of sulfur dioxide on crop and forest vegetation are abundant near point sources of the pollutant such as smelters and large coal-fired power plants, most of these examples predate compliance with the Clean Air Act of 1970. Currently, ambient SO_2 concentrations by themselves are not responsible for large-scale regional crop yield reductions in the United States (Shriner et al. 1990). Occasional to rare instances of vegetation damage and potential for yield loss may occur in the local vicinity of point sources of SO_2 under unusual meteorological conditions. In forest ecosystems, growth reduction may occur near sources of SO_2 as a result of the cumulative effects of low-level exposures over long periods of time (e.g., decades).

Nitrogen dioxide, however, is not a direct cause of regional-scale growth or yield reduction in U.S. agricultural crops or forest trees. Much less is known of the role that NO_2 plays in plant stress than is known for SO_2 or ozone, and much less is known in the response of trees to NO_2 than for the response of crops. However, most preliminary studies have indicated that NO_2 is not toxic to plants within normal ambient concentration ranges that occur even in the most polluted environments. As a result, plant effects research with NO_2 has been of low priority. Some evidence suggests the potential for interactions between NO_2 and SO_2 and/or O_3, but dose-response functions for such interactions do not exist.

5.4.6 Factors Constraining Quantification and Analysis of Externalities

Although much is known about how plants function and how individual plants respond to stress, there are many gaps in our knowledge. These gaps constrain an analysis of externalities of the coal fuel cycle in a number of important ways. Many details are lacking in translating the known and understood response of seedlings exposed to pollutants under controlled experimental conditions to an understanding of the response of a mature tree in a forest, a stand of mature trees, or a forest landscape composed of many such stands. The greatest single challenge of future assessment efforts will be to deal with this issue of **scale**. In order to successfully analyze meaningful economics and policy options, or make forest management decisions, the impact of atmospheric deposition must be understood at a geographic scale large enough to be relevant to those decisions. In most cases, even predictions at the level of a forest stand are of little value without some way to extrapolate to the region, or perhaps at least to the forest type within a region. Regional forest assessment is very difficult because of the natural variability of forests and soils across large regions, and because of the difficulty in adequately sampling and measuring their response (Shriner et al. 1990).

Regional forest assessment is very difficult because of the natural variability of forests and soils across large regions...

For assessment purposes, currently available techniques for extrapolation to the regional scale are inadequate to capture the complexity of the number of mechanisms which might be involved within a single region. New theory and methods are necessary in disciplines such as landscape ecology in order to adequately address this **missing knowledge base**.

Also for assessment purposes, a mechanism of long-term monitoring and inventory must be established to provide a baseline against which future change can be measured. The absence of such baselines for many resource response variables at the present time represents a significant **measurement** issue limiting economic analysis. As such baselines are established, major emphasis should be placed on identification of the biologically meaningful indicators of ecosystem health, where and how frequently they should be measured, and their relationship to relevant economic variables.

The most serious limitations associated with the crop productivity estimates and the analysis of the effects of changes in ozone concentration on crop yield are the inadequate sampling of environmental conditions (including pollutant concentrations in rural environments) and an inadequate number of experimental test sites to represent the populations of sites and environments of inference in regional or national assessments. Where inferences are made for randomly sampled environments, the underlying assumption is required that the environments are representative of the populations of interest. It must also be assumed that past and present studies can be used to predict future response environments. The best set of currently available data, that generated by the NCLAN program (Heagle et al. 1988), has multiple years of data for multiple sites for only a limited number of species. Although a single site within a region is inadequate to represent the entire region, the relative consistency of responses across the NCLAN test sites and years lends confidence to the general results used in this analysis (NAPAP 1990).

Inferences for national assessment purposes are also limited by the small number of crop species included in the analysis. However, the four major species included by the NCLAN program in their analyses, corn, soybean, wheat, and sorghum rank first, second, third, and fifth, respectively, in value of crop production, totaling 182.5 million acres harvested, and $38 billion. The four species combined represent approximately 61% of the total crop value for all crops in the United States, and approximately 62% of the total crop area harvested for all crops in 1987. Each of the four species is broadly distributed geographically. Wheat and corn are each produced commercially in 41 states in the United States, soybean in 29 states, and sorghum in 20 states. Ozone exposure-response functions are only available for a total of 14 crop species. As a result, many locally and regionally important crop species, any one of which represents a small fraction of national crop value or acres harvested, are not included in any analyses of crop impacts to date (NAPAP 1990).

Growing seasons vary for each crop and for each agricultural region. Even within geographic areas as small as a State, there are a range of growing seasons because farmers have flexibility in planting and harvesting dates. Furthermore, plants are more sensitive to pollution impacts at certain stages during their development (e.g., flowering fruit set). These factors combine to result in a range of sensitivity of plants to yield impacts during the growing season, and have been accounted for in the development of the exposure-response functions used in these analyses (Heagle et al. 1988).

In the final characterization of economic effects, it is insufficient to document and describe changes in physical yield (e.g., bushel/acre) on a percentage basis. A 10% loss of one crop may be much more or less important than a 10% loss of another crop depending on the region and the value attached to those crops in the market place. Substitutions available to the farmer and other forms of market elasticity are important factors in fixing the final magnitude of the external cost for the agricultural and forest commodities discussed in these analyses.

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PART III

ESTIMATING HEALTH EFFECTS

PAPER 5[1]

AIR POLLUTION AND MORTALITY: DETERMINATION OF A QUANTITATIVE ASSOCIATION

1. INTRODUCTION

Over the last few decades, several epidemiologic studies have reported associations between daily concentrations of ambient particulate matter and mortality among the general population. These associations were not only observed at the extremely high concentrations usually associated with the "killer fogs" of London or Donora, Pennsylvania, in the 1940s and 1950s but also at concentrations currently and commonly experienced in metropolitan areas throughout the United States.

Among these studies, statistically significant relationships have been found using several alternative measures of particulate matter including total suspended particulates (TSP) [or particles of all sizes], fine particles (particles less than 2.5 microns in diameter), British smoke (BS), coefficient of haze (COH), and sulfates. None has involved measurement of the mass of particulate matter less than 10 microns in diameter (PM_{10}), the metric used by the U.S. Environmental Protection Agency (EPA) in the National Ambient Air Quality Standards. The studies have been conducted in several different cities and seasons, thereby incorporating a wide range of climates, chemical compositions of particulate matter, and populations. The epidemiologic investigations have used two principal study designs: time-series and cross-sectional. Time-series analysis examines changes in daily mortality rates within a specific area as air pollution levels fluctuate. A cross-sectional analysis compares differences in mortality rates across several cities at a selected point or period of time. The time-series studies have the distinct advantage of reducing or eliminating the problems associated with confounding or omitted variables, a common concern in the cross-sectional studies. Because the population characteristics are basically constant over the

[1]Based largely on a working paper by B. Ostro.

study period, the only factors that may vary with daily mortality are environmental and meteorologic conditions. Thus, researchers are able to more easily elicit the effects of air pollution and weather on mortality.

The inference of causality from statistical associations has been a controversial issue long debated in the literature and among expert panels (Hill 1965; EPA 1982). Causal inferences in epidemiology is an informal process, involving consideration of several guidelines or criteria. Ultimately, the satisfaction of several criteria has been proposed including: (1) consistency of the association, (2) specificity of the association, (3) existence of a dose-response curve, (4) strength of the association, (5) coherence of the association with other known facts, and (6) biological plausibility of the association. The expert panels emphasized "that no one criterion is definitive by itself nor is it necessary that all be fulfilled in order to a determination of causality" (EPA 1982).

This paper seeks to examine the air pollution-mortality association in view of these criteria, with particular focus on the consistency of the association. Replication of results in other environments and populations is a powerful test before causality can be inferred. The check for consistency will be accomplished by reviewing and comparing the results of the more recent empirical studies. Ultimately, the results of these calculations are presented in terms of the percent increase in mortality associated with changes in PM_{10}. After examining the consistency of the associations, some of the other criteria will be briefly discussed.

Several criteria had to be met for a paper to be included in this review and analysis. First, a proper study design and methodology were required. Therefore, there was a focus on time-series regression analyses relating daily counts of mortality to air pollution in a single city or metropolitan area. Second, studies that minimized confounding and omitted variables were included. For example, research that compared two cities or regions and characterized them as "high" and "low" pollution areas were not included because of potential confounding by other factors in the respective areas. Third, concern for the effects of seasonality and weather had to be demonstrated. This could be accomplished by either stratifying and analyzing the data by season, by prefiltering to reduce patterns in the data, by examining the independent effects of temperature and humidity, or by correcting the model for possible autocorrelation. A fourth criterion for study inclusion was ensuring that the study included a reasonably complete analysis of the data. Such analysis included a careful exploration of the primary hypothesis, as well as examination of the robustness and sensitivity of the results to alternative functional forms, specifications, and influential data points. When studies reported the results of these alternative analyses, the quantitative estimates that were judged as most representative of the overall findings were those that were summarized in this paper. Finally, for inclusion in this review, the study had to provide a measure of particulate matter that could be converted into a common metric. For example, studies that used weekly or monthly average concentrations or that involved particulate measurements in poorly characterized metropolitan areas (e.g., one monitor representing a large region) were not included in this review.

2. ANALYSIS OF CONSISTENCY

For the calculations, we must convert alternative measures of particulate matter into PM_{10}. Ideally, this would be accomplished by comparing co-located monitors at each study site. Unfortunately, for many of the measures, these data are not available and we are forced to use broad estimates of the relationships between alternative measures of particulate matter. The results of our analysis of consistency, however, indicate that the findings are generally robust to these assumptions. To convert from TSP to PM_{10}, we relied on the estimate of EPA (1982) suggesting that PM_{10} is between 0.5 and 0.6 of TSP, and use the mean of 0.55. Using the reported averages from 100 cities in 1980, we assumed that sulfates constitute approximately 0.14 of TSP (Ozkaynak and Thurston 1987). Therefore, the ratio of sulfates to PM_{10} is 0.25. The BS measurement is based on the amount of light reflected through a filter paper stained by ambient air flowing through the paper. Because monitors for BS do not admit particles less than 4.5 microns in diameter, they indicate concentrations of fairly small particles, but they do not measure particle mass like TSP and PM_{10}. However, available data for co-located BS and TSP monitors in London indicate an average ratio of BS/TSP of 0.55, the same ratio of PM_{10} to TSP (Cummings and Waller 1967). Therefore, based on this and additional analysis by the California Air Resources Board (California Air Resources Board 1982), it is assumed that PM_{10} is roughly equivalent to BS.

2.1 LONDON

Among the earliest empirical estimates of mortality outcomes associated with particulate matter is the analysis of data from London for the winter of 1958–59, where a statistically significant relationship was found between daily deaths and daily levels (24-h average) of BS (Martin and Bradley 1960). London data for 14 winters, 1958–59 through 1971–72, have been analyzed by Mazumdar et al. (1982), Ostro (1984, 1985) and Schwartz and Marcus (1986, 1990), and reviewed by EPA (1986). In the earlier winters, the levels of BS were extremely high; the mean for the first seven winters was 270 $\mu g/m^3$. However, the mean for the last seven winters was 80 $\mu g/m^3$, and in 3 of the last 4 years the mean concentrations were below 70 $\mu g/m^3$. Therefore, the concentrations of BS in London in the last few years are comparable to those commonly found in the United States.

Although these analyses involve several different statistical methods, the following general conclusions can be drawn: (1) there is a strong relationship between particulate concentrations and daily mortality in London, which holds both for the entire data set and for individual years (the later years exhibited almost an order of magnitude decrease in air pollution concentrations); (2) there is no indication of a "no-effects level" (i.e., a threshold) at the lower concentrations of air pollution experienced in London; (3) the association between air pollution and mortality cannot be explained away by meteorologic factors or by serial correlation in the data; and (4) regardless of the model specified, the quantitative implications of the studies are similar. For example, comparing results of linear models, Ostro (1985) obtains a slope relating BS to mortality of 0.088 (without controlling for autocorrelation) versus 0.0898 or 0.0808 from Schwartz and Marcus (1986) (both estimates controlling for autocorrelation and the first including temperature and humidity as explanatory variables).

For this review, quantitative estimates of the London data are taken from Schwartz and Marcus (1990) that involves the most complete examination of the effects of temperature and humidity, autocorrelation, and functional form. Their results over the 14 years suggest the following:

$$\text{Daily mortality} = 2.31 * (\text{daily average BS})^{0.5} \text{ in London.} \qquad (1)$$

The standard error of the estimated regression coefficient is 0.160. During the period of study, there was an average of 280 deaths per day and a mean concentration of BS of 174 $\mu g/m^3$ in London. After taking the derivative of equation 1 and substituting in the mean of daily deaths, the effects of PM_{10} can be expressed as:

$$\%\text{ change in daily mortality} = (0.004125\ \text{BS}^{-.5})\ (100) * \text{change in BS.} \qquad (2)$$

Thus, at the mean level of BS, a 10 $\mu g/m^3$ increase in PM_{10} is associated with a 0.31% increase in mortality. Using plus or minus one standard error from the estimate, a confidence interval of 0.29 to 0.33% is obtained. The 0.31% increase is the same that would be predicted from the linear models described above.

2.2 ONTARIO, CANADA

Plagiannakos and Parker (1988) used pooled cross-sectional and time-series data for nine counties in Southern Ontario, Canada, for the period of 1976 to 1982. Their model attempted to explain mortality as a function of several socioeconomic factors (education, population over age 65, alcohol consumption); time; meteorology; and air pollution including TSP, sulfates, and sulfur dioxide. There was no correction for autocorrelation. Because mean ambient concentrations were not provided by the authors, graphical displays were used to estimate pollution levels. The mean for TSP appeared to be approximately 70 $\mu g/m^3$, while the mean for sulfate was approximately 12 $\mu g/m^3$.

A statistically significant association was found between all-cause mortality and sulfates and sulfur dioxide, but not TSP, based on separate regressions for each pollutant. Also, an association was reported between respiratory-related mortality and TSP, sulfur dioxide, and sulfates, each estimated separately. For all-cause mortality, the model with the highest association between air pollution and mortality, is represented by the following quantitative relationship:

$$\text{log(annual mortality in Ontario)} = 0.047 \text{ log (sulfate)}. \qquad (3)$$

The standard error of the estimated coefficient is 0.0235. To estimate the change in mortality per $\mu g/m^3$ of PM_{10}, we take the total derivative of (3) and obtain:

$$\text{Percentage of change in mortality} = 0.047 \text{ (change in sulfate/mean sulfate)}.$$

Substituting the mean concentration of sulfates and the ratio of sulfate to PM_{10} we obtain:

$$\text{Percentage of change in mortality} = 0.098 \text{ (change in } PM_{10}). \qquad (4)$$

This indicates that a change in PM_{10} of 10 $\mu g/m^3$ corresponds to a 0.98% change in all-cause mortality. Applying plus and minus one standard error, a 10 $\mu g/m^3$ change in PM_{10} generates an effect ranging from 0.49 to 1.47%.

2.3 STEUBENVILLE

A recent time-series study (Schwartz and Dockery 1991) examined the association between daily mortality and air pollution in Steubenville, Ohio, from 1974 to 1984. This study reported a mean TSP of 111 $\mu g/m^3$ with a median of 91. The analysis controlled for the effects of weather and seasonality. A statistically significant relationship between TSP and mortality was reported and the estimated coefficient was robust with respect to the included explanatory variables. The authors also found that when sulfur dioxide was used in place of TSP as the single measure of outdoor air pollution, it too was significant. Regarding particulates, the most general finding is expressed as:

$$\log(\text{mortality}) = 0.000352 \text{ (change in TSP)}. \qquad (5)$$

The standard error of the estimate was 0.00011. In terms of PM_{10}, this converts to:

$$\text{Percentage of daily change in mortality} = 0.0640 \text{ (change in } PM_{10}). \quad (6)$$

This implies that a 10 $\mu g/m^3$ change in PM_{10} corresponds to a 0.64% increase in mortality over baseline. The confidence interval, based on plus or minus one standard error, is 0.44 to 0.94%

2.4 PHILADELPHIA

A similar time-series study was conducted in Philadelphia (Schwartz and Dockery 1991) using data from 1973 to 1980. The mean of the TSP concentrations was 77.2 $\mu g/m^3$ with a median of 73. The analysis controlled for the effects of temperature, temperature extremes, and dew point. A statistically significant association was reported between TSP and all-cause mortality. The association was also significant for both the populations under and over 65 years of age. In addition, mortality due to chronic obstructive pulmonary disease, pneumonia, and cardiovascular disease were each separately associated (at $p < 0.05$) with TSP. When sulfur dioxide was used in place of TSP as the exposure measure, it was also statistically significant. The results for the association of all-cause mortality and TSP are represented as:

$$\log(\text{mortality in Philadelphia}) = 0.000661 \text{ (change in TSP)}. \qquad (7)$$

The standard error of the estimated coefficient was 0.000131.

Converting into PM_{10}, we obtain:

$$\text{Percentage of daily change in mortality} = 0.120 \text{ (change in } PM_{10}). \quad (8)$$

This indicates that a 10 $\mu g/m^3$ in PM_{10} is associated with a 1.20% increase in mortality, with a one standard error confidence interval of 0.96 to 1.44%.

2.5 SANTA CLARA COUNTY, CALIFORNIA

A recent time-series analysis examined the relationship between COH and mortality for the metropolitan area surrounding San Jose, California (Fairley 1990). Daily mortality and suspended particles measured as COH were compared between 1980 and 1986. A statistically significant association was found between COH and both all-cause mortality and respiratory-related mortality, after controlling for temperature and humidity. The models were also tested for the influence of year, season, day, and weather, with little change in the overall results.

The general model for all-cause mortality, chosen by the author as most representative, is indicated the following:

$$\text{Daily mortality in Santa Clara} = 0.0084 \text{ COH}. \qquad (9)$$

The standard error of the estimated coefficient was 0.0029.

To obtain the effect of a 10 $\mu g/m^3$ change in PM_{10} on the percent change in mortality, several adjustments must be made. First, as discussed by the author, only readings from the central city monitor were used in the study. This monitor averaged about one-third higher concentrations than the metropolitan-wide average, a metric typically used in the other studies. The author also found that the COH to TSP ratio was at least one. Therefore, we assume that PM_{10} is approximately 0.50 $*$ COH. Finally, the study population averaged about 20 deaths per day. Therefore, we adjust the coefficient by (4/3)(1/0.5)(1/20) to obtain:

$$\text{Percentage of change in mortality} = 0.112 \text{ (change in } PM_{10}). \qquad (10)$$

This implies that a 10 $\mu g/m^3$ change in PM_{10} results in a 1.12% change in mortality. Applying the standard error, we obtain a range of effect of 0.73 to 1.51% associated with a 10 $\mu g/m^3$ change in PM_{10}.

2.6 LOS ANGELES COUNTY, CALIFORNIA

A time-series analysis of daily mortality and air pollution between 1970 and 1979 was conducted by Shumway et al. (1988). Again, the influences of several meteorologic factors, such as temperature and humidity, were incorporated into the analysis. Particulate air pollution was measured as KM, a measure of light transmittance that is sensitive to smaller particles. After correcting for autocorrelation, a dose-response between daily KM and all-cause mortality was found. The association is represented as:

$$\text{Daily mortality in Los Angeles} = 0.438 \text{ KM}. \tag{11}$$

The standard error of the estimate is 0.034. Unfortunately, the relationship between KM and PM_{10} is not well understood and the study did not report means for the variables of interest. Therefore, interpretation of these results is more difficult that those reported above. However, graphical presentations indicate that during the study period, the mean concentration of KM was approximately 60, and there was an average of 175 deaths per day. Because the annual mean PM_{10} for the county during the study period was approximately 65 $\mu g/m^3$, we assume for our purposes a KM to PM_{10} ratio of 0.92. Therefore, after conversion, the results are expressed by :

$$\text{Percentage of change in mortality} = 0.231 \ (\text{change in } PM_{10}) \ . \tag{12}$$

Therefore, a 10 $\mu g/m^3$ change in PM_{10} is associated with a 2.3% change in mortality, with an interval of 1.9 to 2.7%.

2.7 MORE RECENT DAILY TIME-SERIES STUDIES

Recently, several additional studies relating daily mortality to particulate matter have been published. This association has been reported for Detroit (Schwartz 1991), St. Louis and Easter Tennessee (Dockery et al. 1992), Utah Valley (Pope et al. 1992), Minneapolis (Schwartz 1992), and Birmingham, Alabama (Schwartz 1993). While there will continue to be some uncertainty about the precise magnitude of the effect, these studies tend to support the quantitative estimates of the studies reported above.

2.8 CROSS-SECTIONAL STUDIES

Additional evidence is provided by studies using cross-sectional data such as Ozkaynak and Thurston (1987), Evans et al. (1984), and Lipfert et al. (1988). Ozkaynak and Thurston examined roughly 100 metropolitan areas in the United States using the 1980 vital statistics. This study controlled for socioeconomic characteristics and conducted additional sensitivity analysis to determine the impact of certain cities and alternative model specifications. The authors found statistically significant relationships between mortality rates and alternative measures of ambient particulate matter including sulfates and fine particulates. Specifically, the study reports that existing sulfate concentrations (mean of 11.1 $\mu g/m^3$) correspond to a 4 to 9% increase in mortality. Assuming a ratio of sulfates to PM_{10} of 0.25 as above, this suggests that a 10 $\mu g/m^3$ change in PM_{10} corresponds to a 0.092 to 2.06% change in all-cause mortality, with a mean of 1.49%.

The cross-sectional study conducted by Evans et al. (1984), reanalyzing data originally used by Lave and Seskin (1977), also reports an association between mortality rates and TSP or sulfates. Using 1960 data for 117 cities, the mean concentrations of sulfates and TSP were 10.1 $\mu g/m^3$ and 118 $\mu g/m^3$, respectively, and the mean mortality rate was 9.12 deaths per thousand. The general results of the analysis indicate that:

$$\text{Annual mortality/100,000} = 2.63 \text{ Annual average sulfate.} \qquad (13)$$

The reported standard error of the regression estimate is 1.40. Converting from sulfates to PM_{10}, we obtain:

$$\text{Percentage of change in mortality} = 0.0721\ PM_{10}. \qquad (14)$$

This generates an estimated change of 0.72% in response to a 10 $\mu g/m^3$ change in PM_{10} with a range, based on one standard error, of 0.34 to 1.10%.

The final cross-sectional study reviewed is the analysis of 1980 data by Lipfert et al. (1988). Sulfate measurements from the SAROAD network were obtained for 197 cities throughout the United States. The mean annual average concentration of sulfates and TSP was 9.5 $\mu g/m^3$ and 41.2 $\mu g/m^3$, respectively, with a mortality rate of 9.14 per thousand. A comprehensive analysis of influential data points, outliers, residuals, and robustness was performed. For the most general results representing the effect of sulfates on mortality per thousand, the authors report a coefficient of 0.041 with a standard error of 0.020. After conversion, this indicates that a 10 $\mu g/m^3$ change in PM_{10} would result in a 1.09% change in mortality with a range, based on one standard deviation, of 0.55 to 1.64%.

This review suggests that the recent studies linking particulate matter to mortality generate remarkable consistent results, as summarized in Table 1. The means of the estimated effect of a 10 $\mu g/m^3$ change in PM_{10} implied by these studies varies between 0.31 and 2.31%. However, the highest and lowest estimates are derived from the particulate matter measurements with the greatest uncertainty in translating to equivalent PM_{10} concentrations (i.e., BS and KM). If these two estimates are dropped, the range of mean estimates is from 0.64 to 1.49 with a mean of 1.03%. The lower confidence level varies between 0.34 and 0.96% with a mean of 0.63, while the upper confidence level varies between 0.84 and 2.1% with a mean of 1.44%.

3. OTHER CRITERIA FOR INFERENCE

Among the other criteria for inference is the existence of a dose-response, the specificity and coherence of the association, and the biological plausibility of the association. Because the existence of a dose-response relationship is clearly indicated in the previous discussion, we focus on the remaining criteria. In examining the air pollution-mortality association, specificity may be expressed by whether PM_{10} is associated with disease-specific mortality, such as respiratory or cardiovascular diseases. For some of the time-series studies, this is difficult to determine because the low counts of daily disease-specific mortality preclude independent hypothesis testing. However, several of the studies do test for these associations. The time series analysis in Ontario, Canada (U.S. EPA 1986) reported statistically significant associations between respiratory-related mortality and both TSP and sulfates.

In Philadelphia (Schwartz and Dockery 1991), significant associations were found between TSP and several different causes of mortality including COPD, pneumonia, and cardiovascular disease. In addition, that study reported a greater effect for those above age 65, as expected. Finally, in Santa Clara County, a statistically significant relationship was found between COH and both respiratory-mortality (including emphysema, chronic bronchitis, and pneumonia) and cardiovascular-mortality (including myocardial infarction, arteriosclerosis, and stroke), and not found between COH and cancer, other nonaccidental mortality, and accidental mortality (Fairley 1990). These studies, therefore, provide ample evidence of specificity. It is also important to note that statistical bias is not introduced by using all-cause mortality as a dependent variable in the regression analysis. If factors not reasonably related to the explanatory variable are included in the dependent variable, they are essentially random errors that will not bias the estimated regression coefficient.

Coherence can be demonstrated by evidence that PM_{10}, or small particles in general, are associated with an entire range of health effects. That association has been reported amply in literature. For example, particulate matter has been reported to be associated with emergency room visits (Samet 1981; Plagiannakos and Parker 1988), hospital admissions (Pope 1991; Thurston et al. 1992), respiratory symptoms severe enough to cause restrictions in activity (Ostro 1990), acute bronchitis (Dockery et al. 1989), and asthma attacks (Whittemore and Korn 1980; Ostro et al. 1991).

Currently, the biological mechanism for the air pollution-mortality association is unknown. In part, this is a result of the heterogeneous mix of chemicals that constitute particulate matter, making it difficult to test for a mechanism in a clinical or experimental setting. However, current evidence indicates that small particles can penetrate deeply into the lung and result in bronchoconstriction and an alteration in respiratory mechanics (EPA 1986). In addition, constituents of particulate matter, such as acid sulfates, may irritate the upper airway and deep lung, reduce bronchial clearance, and modify the lung's resistance to infection (Ostro et al. 1991). Pre-existing respiratory infection may also be an important part of the mechanism. Various respiratory viruses cause prolonged bronchial hyperresponsiveness via multiple mechanisms involved epithelial cell damage and inflammation (Zoratti and Busse 1990; Empey et al. 1986).

In this setting, exposure to pollution represents an additional burden of inflammation, which may exacerbate the conditions of individuals already compromised with respiratory infections. For example, Bates (1992) has suggested several mechanisms by which air pollution may increase respiratory and cardiovascular mortality. Among these mechanisms are: (1) increases in lung permeability that precipitate pulmonary edema in people with myocardial damage and increased left atrial pressure, and (2) air pollution-induced bronchiolitis in the presence of pre-existing heart disease that may precipitate congestive heart failure. Additional support for this possibility is obtained from several recent epidemiologic studies with large enough sample sizes to examine disease-specific mortality. For example, Schwartz and Dockery (1991) found that air pollution was associated with elevated risk for mortality associated with chronic obstructive pulmonary disease, pneumonia, and cardiovascular disease. Likewise, Pope et al. (1992) found higher associations between PM_{10} and respiratory- and cardiovascular-specific mortality relative to all other causes of death.

Thus, considering all of the evidence, the criteria for causality appear to hold. In addition, the empirical observations of mortality during air pollution episodes in London, England and Donora, Pennsylvania provide confirmation of an association between air pollution and mortality. The delineation of the precise mechanism awaits further animal studies using various constituents and mixtures of particulate matter.

4. CAVEATS AND UNCERTAINTIES

Numerical estimates reported elsewhere in this study suggest that the impacts of particulate matter on human health are one of the major sources of externalities in several of the fuel cycles (e.g., ORNL/RFF 1994). Nevertheless, as is usually the case in scientific inquiry, there is some uncertainty about the exact extent of the health impacts and about the biological mechanisms through which they occur. This uncertainty gives rise to several issues about the methods that this study recommends for estimating the impacts of particulates on human health. This section summarizes these issues and the study's position on each of them.

4.1 UNKNOWN BIOLOGICAL MECHANISM

One concern about using exposure-response relationships to estimate the effects of particulate matter on health is that science has not yet identified the biological mechanism by which exposure to particulate matter increases the risk of mortality. Samet and Utell (1993) reviewed recent epidemiologic studies and concluded that, although they were suggestive, they did not prove that a causal association exists between particles and mortality. Specifically, they state that:

> "These toxicological considerations suggest that the associations of adverse health effects with current levels of particulate matter should not yet be accepted as causal (p. 1335)."

Part of the difficulty in determining the potential biologic mechanisms, typically deduced from toxicologic or human clinical studies, results from the uncertainty about the responsible constituent of particulate matter. Researchers are faced with several candidate pollutants including PM_{10} mass, fine particulates, sulfates, acidic aerosols, and various elements typically found in particulate matter. Fairley (1990) and other researchers indicate that the precise constituent of total particles that are the real cause of the health effects are unknown and that there is a danger of imposing blanket regulations on total particulates when only specific ones are of actual concern. Regardless of the specific constituent, however, the agent of concern is likely to be strongly correlated with TSP or PM_{10}, the measures typically used in the epidemiologic studies. Otherwise, these pollutant measures would not in themselves be statistically associated with mortality and morbidity so consistently.

However, it is of continued interest to determine the specific constituent of particulate matter that may be responsible for these and other health effects. Different types of particulates have different chemical and physical properties -- implying that they have different impacts on human health, and thus different externality values. However, the evidence from the current literature is mixed on this issue:

- On the one hand, Schwartz and Marcus (1990) report an effect from particulate matter and a lesser effect from a higher covarying concentration of sulfur dioxide. Also, Dockery et al. (1992) find that PM_{10}, but not sulfates and acid aerosols (two of the smaller types of particles), are associated with premature mortality. Likewise, Frampton et al. (1992) found no statistically significant adverse impacts to twelve healthy non-smokers exposed to very high levels of sulfuric acid aerosols. Lipfert (1993) generally confirms Dockery et al.'s results, finding that with a smoking variable included in the models, the broad measure of particulates was highly correlated, but that sulfates were not.

- On the other hand, Plagiannakos and Parker (1988) report effects from sulfates and sulfur dioxide, but not from TSP. This result supports that of Hatzakis et al. (1986), who found a mortality effect from moderate levels of sulfur dioxide, but no effects from particulate matter, measured as British Smoke. Ozkaynak and Thurston (1987) found sulfate to be the most important air pollutant. Another report indicates that the acidic aerosols may be of greater consequence than particulate matter (Thurston et al. 1989).

Taken together, past studies are inconclusive about the specific types of particulates that may result in elevated mortality. While studies (refer to Sect. 4.2) have identified associations between particulates and mortality, some would argue that these associations be regarded as "high" estimates in that the precise biological mechanism is unknown and no studies suggest that the associations should be higher than those used in our study.

However, as suggested by Ito et al. (1993), the inability to assign the "effect" to a single pollutant is a function of, among other things, the mis-measurement of the pollutants. This mis-measurement arises from both the analytical errors in the monitoring technique employed and the errors in the spatial representativeness of the samples in relation to the population distribution. Thus, for example, studies using both H^+ and other measures of pollution may find H^+ to be less significant simply because the analytical measurement errors are likely to be higher. This may partly explain the results of Dockery et al. (1992) where an association was observed between mortality and PM_{10}, but not with sulfates or H^+ in St. Louis, or that of Hatzakis (1986) which found an effect of sulfur dioxide but not particles, measured as British Smoke. Hatzakis' (1986) finding is contrary to most studies such as the studies of the London winters, where British Smoke was strongly associated with mortality. This suggests that the correlation between British Smoke and PM_{10} varies greatly and depends on climate and atmospheric chemistry at the particular site. Any random errors in measuring the exposure typically will reduce the size of the estimated effect. Therefore, it is quite possible that the magnitude of the effect of particles on mortality is consistently underestimated.

It is important to note that, using the daily time-series studies of mortality, PM_{10} or TSP is consistently associated with mortality across a wide range of climates, seasons, covariates, and populations. The association appears to hold both in areas where particles peak in the summer and in those where it peaks in the winter; where there is high or low humidity; in areas of high and low acidity; and in areas where there are temperature extremes and where the temperature is fairly moderate. Regarding the time-series studies that reported associations between mortality and particulate matter, Pope et al. (1992) state, "Steubenville, Philadelphia, St. Louis and Kingston are in the eastern United States where there is a humid, continental climate. Santa Clara has a warm, dry climate and Utah County has a semi-arid climate and cold winters. Moreover, in contrast to the eastern locations where particulate matter concentrations are highest in the summer -- when the weather is warm and humid - highest particulate pollution concentrations in Utah County occur during cold winter weather. The substantial differences in the coincident weather patterns that accompanied particulate exposure in these different locations, coupled with the similarity in observed associations between particulate pollution and mortality, make it unlikely that confounding weather effects are the sources of the apparent pollution effects." Overall, the evidence is fairy compelling that increases in particles which contribute to PM_{10} mass are associated with increased risk of mortality. The above findings are both internally consistent across many time-series and cross-sectional mortality studies and consistent with several different morbidity endpoints, as indicated above. This suggests that the findings are not spurious.

For our quantitative estimates, we are forced to assume that the existing pollution mix (composition and particle size) that was observed in the original study will remain relatively constant. Changes in these factors also will have an impact on the estimated effect. The fact that the same general magnitude of effect was observed in many different locations, with varying compositions, suggests that the assumption about composition does not seriously bias our estimates, and that PM_{10} is, at a minimum, a good surrogate measure.

Finally, we have indicated above that most of the criteria for causality established by Hill (1965) have been met. In his discussion, Hill indicates that it was not necessary to meet all of the criteria to reasonably infer a causal relationship. While it is important to understand the actual mechanism responsible for mortality and morbidity, the lack of a mechanism is not sufficient grounds for ignoring the considerable body of evidence and for assuming a zero risk.

4.2 EXPOSURE-RESPONSE THRESHOLDS

A clear threshold for effects has not been identified from currently available times-series data on air pollution and mortality. However, the cross-sectional data are more suggestive of a potential no-effects level. Unfortunately, it is not clear whether this inconsistency is a result of differences in study design and data, or because a real threshold exists. Current research is unclear about whether a threshold level of particulates must be reached before there are effects on human health. Given the current uncertainties, and the limited evidence available from the cross-sectional studies, we assume that a threshold exists for both mortality and morbidity at 30 $\mu g/m^3$.

A decision not to use thresholds implicitly assumes that existing, background concentrations of particulate matter exceed any possible threshold that may exist, and that the incremental increase in concentration from the electric power plant results in increased likelihood of morbidity and mortality. A major reason for the concern about the issue is that if we assume that there is no threshold and if one actually exists, then the numerical estimates in our study would overestimate the externalities associated with particulate matter.

Most studies have not used statistical tests to determine whether the exposure-response relationships hold at lower levels of exposure. Specifically, these studies do not assess whether the strength of the association at very high levels of concentration might statistically overwhelm a weak connection at low levels.

An exception, however, is Ostro (1984) which specifically examined the lower concentrations in the data on London winters. Also, more recent studies have used two different approaches to deduce the lack of a threshold. They include: (1) dividing the pollution data into quartiles or quintiles and plotting the dose-response relationship and (2) using locally-weighed smoothing techniques to provide an assumption-free indication of the shape of the dose-response curve. Most studies applying these approaches to the time-series data from one city appear to indicate effects down to very low ambient concentrations. These studies may suggest either the lack of a threshold or effects down to levels of around 30 to 45 $\mu g/m^3$. Specifically, we note that:

(a) Results from the Schwartz and Marcus (1990) study generally support the likelihood of effects at low ambient concentrations with the possibility of no threshold. Schwartz and Marcus (1990) show that the slope, indicating the relationship between air pollution and mortality, was fairly consistent across all of the 14 years of data. The London data are highly seasonal, notably peaking in winter. Since the winter averages are about 70 $\mu g/m^3$ and below, the annual averages are much less than 70 $\mu g/m^3$. Thus, the statistical findings are consistently significant at these lower levels of concentration. Many metropolitan areas in the U.S. have similar levels, so that the exposure-response relationship is expected to hold in the U.S. as well.

(b) Likewise, Ostro (1984) examined the possibility of thresholds at continually lower levels of concentration during winters in London. The results indicate that even in the final few winters, when mean British Smoke levels were 70 $\mu g/m^3$ and below, a relationship between low levels of particles and mortality appears to exist.

(c) Time series analyses are generally regarded as being more reliable than cross-sectional studies. As discussed later in this section, one of the advantages of the time-series approach is that a given set of individuals are examined over a period of time during which many factors (location, activity, occupational and indoor residential exposures, diet and exercise, etc.) are relatively constant. In contrast, cross-sectional studies [such as those by Ozkaynak and Thurston (1987) and by Evans et al. (1984)] use data collected during a single year from many cities. Locations, activities, occupations, diets, etc. vary across cities, thus making it difficult to control for the effects of these potentially confounding factors in the statistical analyses. For this reason, we have relied more heavily on the time-series studies for our estimates since the air pollution-mortality association is less likely to be confounded by other variables, and omitted variable bias is also less likely.

Many of the recent time-series studies divide the data into quartiles (or quintiles) and produce statistical and graphical relationships between increasing levels of particulate matter and the associated risks of mortality. For the studies that involve daily time-series data, the results generally indicate that mortality risks rise monotonically with particles. Data in the references cited below indicate continuous dose-response relationships in several locations including: (1) Utah Valley [Figure 1 of Pope et al. (1992)] with a mean PM_{10} of 47 $\mu g/m^3$; (2) Steubenville, Ohio [Figure 2 of Schwartz and Dockery (1991)] with a mean TSP of 111 $\mu g/m^3$ (PM_{10} of 61 $\mu g/m^3$); (3) Detroit [Figure 1 of Schwartz (1991)] with a mean TSP of 87 $\mu g/m^3$ (PM_{10} of 48 $\mu g/m^3$); and (4) Birmingham, Alabama [Figure 6 of Schwartz (1993)] with a mean PM_{10} of 48 $\mu g/m^3$.

(d) Additional information is provided by the results of the individual-level analysis of the data from the Harvard "six cities" cohort (Dockery et al. 1992). In this analysis, with mortality corrected for individual risk factors such as smoking status, occupational exposure and gender, a monotonic (and statistically significant) association was observed between mortality and both fine particulates and sulfates.

Some evidence is available, however, primarily from the cross-sectional studies, indicating the possibility of a threshold level. Specifically:

(a) Ozkaynak and Thurston (1987) analyzed the relationships between particulate matter and mortality among 100 cities in the U.S. Generally, an association was found between mortality and both sulfates and fine particulates. However, sensitivity analyses indicated that when several metropolitan areas with relatively high sulfates were removed from the regressions, the estimated coefficient for sulfates dropped significantly. This finding implies that the association between sulfates and mortality is influenced by the States with the highest pollution levels, and that it may be difficult to demonstrate a relationship at lower levels.

(b) In another cross-sectional study, Evans et al. (1984) did not find a significant association between sulfate concentration and elevated mortality rate. They found even less significance with total suspended particulates (TSP). Lipfert (1988) contains evidence that geographic areas with lower pollution levels do not have significant associations between pollution levels and mortality.

(c) In the analysis of Santa Clara County, Fairley (1990) does not appear to find effects in the years when mean COH levels were relatively lower. However, this finding is not certain since the analysis was conducted only for the winter months and the COH mean levels were reported for the year as a whole. In an analysis of the four winters considered separately, the association between pollution levels and daily mortality was significant in only two of the years. The PM_{10} equivalent annual average in this area is very close to the Federal standard of 50 $\mu g/m^3$.

(d) While Schwartz and Marcus (1990) generally found a significant relationship between daily particulate pollution and mortality, the statistical significance of their results declined as pollution levels decreased over the 1958 to 1971 time period. In fact, in one of the more recent years, the statistical relationship was not significant.

(e) The most complete threshold analysis performed on cross-sectional data is that of Lipfert (1993). In one analysis, the regression coefficient linking TSP with mortality is observed in data that are successively truncated to include lower levels of TSP. The regression coefficient appears consistent until the average of the maximum (not mean) TSP of the remaining cities falls to below 80 $\mu g/m^3$; the associated mean annual TSP level is not reported. The quintile analysis suggests a monotonic dose-response relationship between TSP and COPD-related mortality but the two lowest points (with TSP levels of approximately 52 and 57 $\mu g/m^3$ or roughly 30 $\mu g/m^3$ PM_{10}) are not statistically different from each other. The latter result does suggest that a threshold may exist at low levels of concentration of about 30 $\mu g/m^3$ PM_{10}. Further evidence in support of this suggestion is the non-monotonic association displayed in the data from St. Louis and eastern Tennessee (Figure 1 of Dockery et al. 1992), where mean PM_{10} levels were approximately 30 $\mu g/m^3$.

Therefore, based on the daily studies in low pollution areas and the results of several cross-sectional studies, the association becomes more uncertain at the lowest observed levels, roughly between 30 and 45 $\mu g/m^3$ PM_{10}. While we are not certain that no effects occur below this level, it does appear that effects are more difficult to detect at these lower levels. However, attributing mortality effects to the incremental contributions to ambient particulate matter resulting from combustion of certain fuels, also appears to be a reasonable extrapolation of the data even at fairly low ambient levels. Thus, to encompass both possibilities, sensitivity analyses can be carried out, with a threshold of about 30 $\mu g/m^3$ PM_{10}, and without any threshold.

4.3 POSSIBLE MODEL MISSPECIFICATION FROM FAILURE TO ACCOUNT FOR KEY FACTORS

Another question about the accuracy of the exposure-response functions that this study uses, stems from the possibility of model misspecification in studies that omit key confounding variables. The reason for this concern is that if some of the mortality is more accurately attributed to various other factors, then the mortality (and thus the externalities) attributed to particulates will decline.

As discussed above, the likelihood of confounding is much greater in the cross-sectional studies. In the time-series studies relating daily changes in air pollution to the daily incidence of a health effect, the likelihood of confounding from other factors is minimized. For example, if a study is conducted over a three month period, and daily emergency room visits or mortality was associated with PM_{10}, then it is extremely unlikely that smoking habits, occupational exposure, diet, exercise and activity patterns, indoor exposure, etc. would change on a daily basis *and* be correlated with daily particulate matter enough to drive the observed association.

In the time-series studies, temperature is most often cited as a potential confounder. However, in most of the studies utilized for our estimates, the models specifically incorporated the independent effects of temperature and season. For example, the studies of Steubenville (Schwartz 1992b) and Utah (Pope 1992) all find an independent effect of weather, with no confounding of the air pollution-mortality association. For example, in the Philadelphia study, it is stated "In the initial analysis of weather factors, hot days (mean > 80 degrees), previous day's mean temperature, mean dew point, and winter temperature were significant predictors of daily mortality." Later they report that, "The association [of mortality] with TSP was not strongly confounded by variables for season and weather."

Among the cross-sectional studies, factors that have been hypothesized to be most likely confounders include smoking, poverty, temperature and indoor air pollution. For example, Ozkaynak and Thurston (1987) consider the effect of poverty on mortality rates but do not include smoking in their model, although Dockery et al. (1993) showed that the inclusion of smoking did not alter the mortality-air pollution association. This is not surprising since there is no evidence that smoking and air pollution are correlated. Although the recent study by Dockery et al. (1993) includes smoking, it omits indicators of socioeconomic status. If in the cross-sectional studies, air pollution and poverty rates are highly correlated, then models that do not include an indicator for socioeconomic status may be misspecified. If the poverty rates vary greater across the cities, then the functional forms and estimated coefficients in the regression equations may be incorrect. The implication is that the externalities attributed to particulates may be over-estimated. To the extent that smoking and lower socioeconomic status are correlated, however, the smoking variable may be incorporating some of the effects of socioeconomic status, as well.

Fairley (1990) raises the issue of indoor air pollution as a confounding factor. He notes that days with high particles may be associated with cold days where people are more likely to stay indoors. He suggests that staying indoors might be the real cause of higher mortality. However, the coldest days are not necessarily the highest particulate matter days. In addition, the daily time-series studies include data from cities in which, in some cases, it peaks in the winter. Therefore, it is unlikely that people are inside on all of the high particulate matter days.

The underlying observation is a simple but powerful one: despite problems in measuring exposure and in controlling for all potential confounders, when the outdoor fixed site monitor measures a higher concentration of particulate matter, higher levels of mortality are consistently observed.

4.4 ACCOUNTING FOR COMPETING RISKS

The Schwartz and Dockery (1991) study of Philadelphia notes that those suffering from chronic obstructive pulmonary disease (COPD, i.e., mainly adult bronchitis and emphysema) and major cardiovascular disease (MCV) are two of the three disease categories most likely to be impacted by high particulate levels. According to the Surgeon General, about 90% of COPD and 30% of MCV cases are caused by smoking. One criticism that has been presented against applying the air pollution-mortality exposure-response functions in our study suggests that the number of COPD cases would be reduced if people stopped smoking, and thus that the number of people at risk to the effects of particulates would be reduced by the same amount. In that case, the exposure-response function would be reduced because the number of susceptible people would be smaller. The same reasoning would apply to MCV. The implication of this argument is that the externalities associated with particulates from fuel cycle activities are overestimated.

This reasoning is inconsistent with a fundamental premise of our study -- to estimate the impacts of an *incremental* addition of a resource (such as a coal-fired power plant). It is therefore appropriate to estimate the *marginal* impacts on environment and health that result from that incremental addition. Sections 1.3 and 4.3.6 of the Coal Report (ORNL/RFF 1994) provide further discussion. Invoking marginality means that, by definition, we hold everything else constant -- we take the world as given, smokers and all, and estimate the additional, marginal effect on health and environment of an incremental increase in a resource (i.e., constructing and operating a power plant and its associated fuel cycle).

To estimate these marginal effects, we use exposure-response functions whose coefficients are estimated statistically, controlling for variation in the values of the other variables in the equation. Furthermore, epidemiologic studies that include a smoking variable can account for both the separate and combined effects of smoking and particulate matter on health. For example, research by Dockery et al. (1993) suggests that former smokers are still at increased risk of mortality related to air pollution, relative to those who have never smoked.

5. SUMMARY

An association between air pollution, measured as particulate matter, and mortality, has been reported in several different cities. These studies have been conducted over a wide range of climates and populations. The time-series studies, which examine the joint occurrence of daily fluctuations in air pollution and mortality, provide the strongest evidence of a true association. After converting the different studies into common units, a striking consistency in the results was observed. The mean effect of a 10 $\mu g/m^3$ change in PM_{10} implied by these studies varies between a 0.64 to 1.49% change in mortality. The fulfillment of other criteria, including specificity, presence of a dose-response relationship, and coherence of results lends strong support to the existence of an actual association between particulate matter and mortality.

The mean effect of a 10 $\mu g/m^3$ change in PM_{10} implied by these studies varies between a 0.64 to 1.49 change in mortality.

However, several criteria, including the consistency of the results, need to be explored before causality is inferred from these studies. This association also appears to hold at low concentrations of particulate matter, possibly with a threshold on effects at about 30 $\mu g/m^3$ PM_{10}. Currently the biologic mechanism is not well understood. In addition, the precise form of the pollutant - TSP, PM_{10}, fine particles, sulfates, acidic aerosols, sulfur dioxide, or some as yet unmeasured pollutant - is unknown based on current available evidence.

This association also appears to hold at low concentrations of particulate matter, possibly with a threshold on effects at about 30 $\mu g/m^3$ PM_{10}.

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PAPER 6[1]

ESTIMATION OF MORBIDITY EFFECTS

1. INTRODUCTION

Many researchers have related exposure to ambient air pollution to respiratory morbidity. To be included in this review and analysis, however, several criteria had to be met. First, a careful study design and a methodology that generated quantitative dose-response estimates were required. Therefore, there was a focus on time-series regression analyses relating daily incidence of morbidity to air pollution in a single city or metropolitan area. Studies that used weekly or monthly average concentrations or that involved particulate measurements in poorly characterized metropolitan areas (e.g., one monitor representing a large region) were not included in this review. Second, studies that minimized confounding and omitted variables were included. For example, research that compared two cities or regions and characterized them as "high" and "low" pollution area were not included because of potential confounding by other factors in the respective areas. Third, concern for the effects of seasonality and weather had to be demonstrated. This could be accomplished by either stratifying and analyzing the data by season, by examining the independent effects of temperature and humidity, and/or by correcting the model for possible autocorrelation. A fourth criterion for study inclusion was that the study had to include a reasonably complete analysis of the data. Such analysis would include an careful exploration of the primary hypothesis as well as possible examination of the robustness and sensitivity of the results to alternative functional forms, specifications, and influential data points. When studies reported the results of these alternative analyses, the quantitative estimates that were judged as most representative of the overall findings were those that were summarized in this paper. Finally, for inclusion in the review of particulate matter, the study had to provide a measure of particle concentration that could be converted into PM_{10}, particulate matter below 10 microns in diameter.

For this report, dose-response functions have been identified and adapted from published epidemiologic and economics literature. These functions allow the estimation of the change in health effects that would be expected to occur with

[1]Based largely on a working paper by B. Ostro.

changes in ambient air pollution levels. For each health effect, a range is presented within which the estimated effect is likely to fall, based on the judgement of this author. The central estimate is typically selected from the middle of the range of results reported in a given study or is based on the best regression specification. The range of the estimate typically reflects the reported standard error of the estimating regression coefficient.

2. PARTICULATE MATTER (PM_{10})

For the estimates, alternative measures of particulate matter must be converted into PM_{10}. Ideally, this would be accomplished by comparing co-located monitors at each study site. Unfortunately, for many of the measures, these data are not available, and we are forced to use broad estimates of the relationships between alternative measures of particulate matter. The results of our analysis of consistency, however, indicates that the findings are generally robust to these assumptions. To convert from TSP to PM_{10}, we relied on the estimate of EPA (1982) suggesting that PM_{10} is between 0.5 and 0.6 of TSP, and use the mean of 0.55. Using the reported averages from 100 cities in 1980, we assumed that sulfates constitute approximately 0.14 of TSP (Ozkaynak and Thuston, 1987). Therefore, the ratio of sulfates to PM_{10} is 0.25.

2.1 RESPIRATORY HOSPITAL ADMISSIONS

Plagiannakos and Parker (1988) used pooled cross-sectional and time-series data for nine counties in their Southern Ontario, Canada study, for the period 1976 to 1982. Their model attempted to explain all respiratory-related hospital admissions for all ages as a function of several socioeconomic factors (education, population over age 65, alcohol consumption); time; meteorology; and air pollution including TSP, sulfates, and sulfur dioxide. There was no correction for autocorrelation. The mean for TSP was approximately 70 $\mu g/m^3$, while the mean for sulfate was approximately 12 $\mu g/m^3$.

A statistically significant relationship was found between the incidence of hospital admissions due to respiratory diseases (RHA) and ambient sulfate levels. TSP was not statistically significant. We use the results from the regression specification that used sulfate as the sole pollutant and which had the greatest explanatory power. The estimated regression coefficient (b = 0.336) provides the basis for a central estimate, and we use plus or minus one standard error from the coefficient (0.129) to generate high and low estimates. The original results of Plagiannakos and Parker indicate the following relationship:

percent change in Respiratory Hospital Admissions = b $*$ log (SO_4)

where:

RHA	=	annual Respiratory Hospital Admissions per 100,000 people
b	=	0.464 (upper estimate)
		0.336 (central estimate)
		0.207 (lower estimate)
SO_4	=	annual arithmetic average sulfate in $\mu g/m^3$.

During the study period in Ontario, the annual average sulfate level was approximately 12 $\mu g/m^3$, and the recent mean level of annual respiratory hospital admissions per 100,000 was 1,450. These means are used to linearize the function. Thus, the change in RHA per 100,000 per unit of PM_{10} is (0.336)(1450)(.25)/12 = 10.15. The following ranges are calculated for the change in annual admissions:

upper change in annual Respiratory Hospital Admissions per 100,000
= 14.02 $*$ change in PM_{10}

central change in Respiratory Hospital Admissions per 100,000
= 10.15 $*$ change in annual PM_{10}

lower change in Respiratory Hospital Admissions per 100,000
= 6.25 $*$ change in annual PM_{10}.

The central estimate suggests that a 10 $\mu g/m^3$ change in PM_{10} results in a (101.5/1450) or 7.00 percent increase in RHA.

... change in annual Respiratory Hospital Admissions per 100,000 = 10.15 $*$ change in PM_{10} ...

Supporting evidence for an effect of particulates on hospital admissions is provided by a study by Pope (1991). This study assessed the association between RHA and PM_{10} between 1985 and 1989 in several mountain valleys in central Utah. Concentrations of sulfur and nitrogen dioxide were low to moderate throughout the study period, and ozone concentrations were low during the winter season, when RHA was elevated. This study indicated that a 10 $\mu g/m^3$ change in PM_{10} related to a 8.9 percent increase (range 7.9 to 9.9 percent) in one valley and a 2.7 percent increase (1.0 to 4.4 percent) in the other.

2.2 EMERGENCY ROOM VISITS

Samet et al. (1981) analyzed the relationship between emergency room visits (ERV) and air pollution levels in Steubenville, Ohio, an industrial town in the midwestern United States. Daily all-cause as well as respiratory-related ERVs at the primary hospital in the area were matched with daily levels of TSP, sulfur dioxide levels, and nitrogen dioxide levels for March, April, October, and November of 1974 through 1977. Daily all-cause and respiratory-related ERV were regressed on maximum temperatures and on each of the pollutants in separate regressions. For each of these endpoints, both particulates and sulfur dioxide coefficients were statistically significant in separate regressions, but these measures were highly correlated.

... central change in annual Emergency Room Visits per 100,000 = 23.54 * change in PM_{10} ...

We have selected the estimated regression coefficient relating TSP to all-cause ERV for this effect and have used plus or minus one standard deviation from the coefficient to generate high and low estimates. All-cause ERV is reasonable to use since (1) some actual respiratory-related ERV may be misclassified into other categories, (2) air pollution may relate to nonrespiratory effects, and (3) the inclusion in the dependent variable of random elements not related to air pollution will not bias the estimated regression coefficients. The results obtained by Samet et al. (1981) indicate the following relationship:

change in daily ERV = 0.011 $*$ change in daily TSP in Steubenville.

The standard error of the regression estimate is not provided, but since the reported p-value is <0.05, we assume it to be 0.0055. Since the approximate population in Steubenville during this period was 31,000, and PM_{10} is 0.55 of TSP, we annualize this equation and obtain a central estimate of:

central change in annual Emergency Room Visits per 100,000 =
23.54 $*$ change in PM_{10}.

The upper and lower coefficients are 34.25 and 12.83, respectively.

2.3 RESTRICTED ACTIVITY DAYS

Restricted activity days (RAD) include days spent in bed, days missed from work, and other days when activities are restricted due to illness. Ostro (1987) examined the relationship between adult all-cause RAD in a 2-week period and fine particles (FP, diameter less than 2.5 microns) in the same two 2-week period for 49 metropolitan areas in the United States. The RAD data were from the Health Interview Survey conducted annually by the National Center for Health Statistics. The FP data were estimated from visual range data available for airports in each area. Since fine particles have a more significant impact on visual range than do large suspended particles, a direct relationship can be estimated between visual range and FP.

Separate regression estimates were obtained for 6 years, 1976–1981. A statistically significant relationship between FP and all-cause RAD was found in each year and supported earlier findings (Ostro 1983) relating RAD to TSP. The mean of the six coefficients is used for the central estimates, and the mean of the highest and lowest two estimates are derived to establish the upper and lower ranges. The form of the estimated relationship was such that the coefficient for FP gives the percentage change in RAD associated with a unit change in FP. Specifically, the results indicate that

... change in annual Restricted Air Days per person per year = 0.0575 * change in PM_{10} ...

$$\text{change in RAD per adult per year} = b * \text{annual RAD} * \text{change in FP},$$

where the high, central, and low estimate of b are 0.0076, 0.0048, and 0.0030, respectively.

To convert this function for our use, we use the following information from the original study: the annual average RAD per adult was about 19 days, and sulfates were 40 percent of FP. Therefore, we annualize the results and convert from FP to PM_{10}, to obtain the following relationship:

$$\text{upper change in annual RAD per person per year} = 0.0903 * \text{change in } PM_{10},$$

$$\text{central change in annual Restricted Activity Days per person per year} = 0.0575 * \text{change in } PM_{10}$$

$$\text{lower change in annual Restricted Activity Days per person per year} = .0356 * \text{change in } PM_{10}.$$

2.4 RESPIRATORY SYMPTOMS

Respiratory symptoms are an additional measure of acute effects of air pollution. Results of Krupnick et al. (1990) can be used to determine the effects of particulate matter. This study examined the daily occurrence of upper and lower respiratory symptoms among a panel of adults in Southern California. A Markov process model was developed to determine the effects of air pollution on health which incorporated the probability of illness on the prior day and controlled for autocorrelation. Among the pollutants examined independently, coefficient of haze (COH) was found to be statistically associated with the probability of reporting a symptom (b = 0.0126, s.e. = 0.0032). Available data (Fairley 1990) suggest a ratio of COH to PM_{10} of 1.75. The marginal effect of COH was calculated by incorporating the stationary probabilities as described in the paper. Therefore, using the results of regressions when COH was the sole pollutant included as an explanatory variable, the following ranges were determined:

... change in annual symptom days per year per person = 2.05 $*$ change in PM_{10} ...

upper change in annual symptom days per year per person
$= 2.57 * \text{change in } PM_{10}$,

central change in annual symptom days per year per person
$= 2.05 * \text{change in } PM_{10}$,

lower change in annual symptom days per year per person
$= 1.63 * \text{change in } PM_{10}$.

2.5 LOWER RESPIRATORY ILLNESS IN CHILDREN

Estimates of lower respiratory illness in children are based on an analysis by Dockery et al. (1989) of children in six cities in the United States. The study related annual concentrations of TSP, PM_{15}, $PM_{2.5}$, sulfate and sulfur dioxide to the presence of chronic cough, bronchitis, chest illness, persistent wheeze, and asthma. These outcomes were noted during a health examination and intake questionnaire taken of the sample of children in each city. A condition of asthma or bronchitis required a physician's diagnosis in the previous year, while chronic cough was defined as a cough being present for 3 months in the last year. A logistic regression analysis was used to estimate the relationship between the probability of a illness being present and the average of the 24-hour mean concentrations during the year preceding the health examination. We focused on the results using PM_{15}, $PM_{2.5}$, and sulfates since they are most closely related to PM_{10}. The analysis of these three pollutants for the five different health outcomes indicated above demonstrated that PM_{15} had a statistically significant relationship to bronchitis and chronic cough. The other pollutant-morbidity associations were positive but not statistically significant at the 95 percent level of confidence. The results are applied to the population age 17 and below (approximately 17 percent of the total population). The basic findings that are used suggest the following:

$$\text{log odds of health outcome} = \log [B/(1-B)] = 0.02368 * PM_{15}.$$

The change in the probability of the health outcome due to a change in PM_{10} can be calculated since, after taking the partial derivative of the above, the following holds:

$$dB = b\, p_o (1-p_o) * \text{change in } PM_{15},$$

where:

dB = change in probability of outcome,
b = estimated regression coefficient,
p_o = the baseline probability of bronchitis.

... central change in annual proportion with bronchitis per year = 0.00160 * change in PM$_{10}$ ...

To determine the effect of a change in PM_{10} on bronchitis, we assume a PM_{10} to PM_{15} ratio of 0.9 and use the baseline probability of bronchitis of 6.47 percent. The central estimate uses the estimated regression coefficient reported by Dockery et al. (1989) (0.02368), and the upper and lower ranges are plus or minus one standard error from this coefficient (0.03543 and 0.01197).

Therefore, the central estimate for the effect of a unit change in PM_{10} equals (0.02368)(1/0.9)(0.0647)(0.9353) = 0.00160. Incorporating the above data, the changes in the annual risk of bronchitis in children are estimated as:

upper change in annual proportion with bronchitis per year
$= 0.00238 * \text{change in } PM_{10}$,

central change in annual proportion with bronchitis per year
$= 0.00160 * \text{change in } PM_{10}$,

lower change in annual proportion with bronchitis per year
$= 0.0008 * \text{change in } PM_{10}$.

For chronic cough, the estimated coefficient is 0.0338, with a standard error of 0.0170, and a mean incidence rate of 0.0577. Therefore, the risk of annual chronic cough can be estimated by:

upper change in proportion with chronic cough per year,
$= 0.00276 * \text{change in } PM_{10}$

central change in proportion with chronic cough per year,
$= 0.00184 * \text{change in } PM_{10}$

lower change in proportion with chronic cough per year
$= 0.0009 * \text{change in } PM_{10}$.

2.6 ASTHMA ATTACKS

... central change in proportion with chronic cough per year = 0.00184 * change in PM$_{10}$ ...

Several studies have related air pollution to increases in exacerbation of asthma. For example, in a study of asthmatics in Los Angeles, Whittemore and Korn reported a relationship between exacerbations of asthma and daily concentrations of TSP and ozone (Whittemore and Korn, 1980). In a recent study Ostro et al. (1991) examined the association between several different air pollutants, including sulfates, $PM_{2.5}$, and acidic aerosols, on increases in asthma attacks among adults residing in Denver. A significant association between the probability of moderate or severe asthma (measured as shortness of breath, cough, or wheeze) and particulate matter was found after controlling for temperature, day of study, previous-day illness, and use of gas stove. The empirical relationship suggests the following:

$$\text{probability of an asthma attack} = 0.77 * \log SO_4.$$

Using the reported sulfate mean of 2.11 $\mu g/m^3$ and the sulfate/PM_{10} ratio reported above, a central estimate is obtained. The low and high range are generated using plus and minus one standard error from the estimated regression coefficient. These estimates should be applied to the approximately 5 percent of the U.S. population that are asthmatic (Evans et al. 1987). The estimates for increases in the daily probability of an attack are

upper change in asthma attacks per person
$= 0.0143 * \text{change in } PM_{10}$,

central change in asthma attacks per person
$= 0.00962 * \text{change in } PM_{10}$,

lower change in asthma attacks per person
$= 0.00487 * \text{change in } PM_{10}$.

... change in asthma attacks per person = 0.00962 * change in PM_{10} ...

Supporting estimates of the effects of air pollution on asthma are provided by Pope et al. (1991). This study examined the response to air pollution among asthmatics in a mountain valley in Utah. Separate analyses were conducted for a sample recruited through a school survey and one recruited through local private physician practices. Subjects were asked to record, on a daily basis, peak expiratory flow, upper and lower respiratory symptoms, and whether extra medication was used. Air pollution, measured as PM_{10}, was associated with diminished peak flow in both samples. Pollution was also associated with lower and upper symptoms among the school-based sample, but not with the patient-based sample. The authors attributed this to proper medical management, awareness, and averting behavior among the asthmatics recruited from active physician practices. Finally, there was a statistically significant association between PM_{10} and extra medication use among both samples.

... change in extra medication per person = 0.0074 * change in PM_{10} ...

Since the use of medication is strongly associated with symptoms, we use it as an comparison with the above estimates. The results suggest the following:

upper change in extra medication per person $= 0.0076 * \text{change in } PM_{10}$,

central change in extra medication per person $= 0.0074 * \text{change in } PM_{10}$,

lower change in extra medication per person $= 0.0071 * \text{change in } PM_{10}$.

2.7 CHRONIC BRONCHITIS

Abbey et al. (1993) compared health surveys of 3,310 healthy, over 25 years old, Seventh Day Adventists in 1977 and 1987 and matched these surveys to daily pollution concentrations over the 10-year interval based on their residential location.[2] The study found that 7.1% of the subjects had developed clinical (not doctor confirmed) cases of chronic bronchitis during the 10-year period. Logit models were used to explain whether a subject developed a new case as a function of a variety of exposure measures, including the average number of hours per year that concentrations exceeded various cut-offs, and the average of concentrations exceeding these cut-offs (including, apparently zero). The calculation of these variables is unclear, particularly as particulate data are never reported in terms of hourly concentrations. The most temporally detailed statistic is a 24-hour average and measurements of TSP (or PM_{10}) are generally taken every sixth day. Additional explanatory variables included exposure to passive smoking, the usual demographics, and occupational exposures.

The researchers found a statistically significant association between the probability of developing a case of chronic bronchitis in 10 years and the average annual number of hours exposed to TSP above 100 $\mu g/m^3$. However, no effects were found for cut-offs below 100 $\mu g/m^3$ (i.e., 75 and 60 $\mu g/m^3$) . Almost parenthetically the authors also report significant relationships between chronic bronchitis and the 10-year mean concentrations above various cutoffs (60 $\mu g/m^3$ and above).

Regarding the first set of results, the authors say that their results "indicate no significantly increased relative risks for levels [concentrations] at or below the Federal standard of 75 $\mu g/m^3$." This statement deserves scrutiny. First, 75 $\mu g/m^3$ was the annual TSP standard, but it was changed to a PM_{10} standard of 50 $\mu g/m^3$. Converting the 75 $\mu g/m^3$ TSP cut-off to PM_{10} (using the 0.55 PM_{10}/TSP ratio used throughout our health effects analyses), the threshold appears to be 41 $\mu g/m^3$ PM_{10}, below the PM_{10} annual average standard of 50 $\mu g/m^3$. Second, given that the key measure of TSP is "annual average hours over the cut-off," it is unclear if an annual average standard is an appropriate benchmark. The 24-hour average standard (260 $\mu g/m^3$ for TSP, changed to 150 $\mu g/m^3$ for PM_{10}) may be more appropriate. In this case, there may very well be elevated chronic bronchitis risk for areas in compliance with annual standards but with many hours exceeding this standard.

As the air quality models and protocols we follow do not provide estimates of the numbers of hours above the standard, the above results cannot be used to estimate damages. However, as noted above, Abbey et al. (1993) also report relative risks for different 10-year *mean* TSP exposures, which match well with our estimates of changes in annual average TSP concentrations.

[2]This matching process was very complicated. There were no requirements that subjects live "close" to an air monitoring station.

From their Table 5, the relative risk to those exposed to mean TSP concentrations of 60 $\mu g/m^3$ is 1.36. This implies that the effect of a 1 $\mu g/m^3$ change in TSP on the log odds ratio is 0.00512.[3] The effect of any change in TSP on the probability of a new case of chronic bronchitis is then $(1-0.071)*0.071*0.00512 = 338*10^{-6}$ * 10-year TSP. To convert to PM_{10} and to annual cases we divide by 0.55 (the PM_{10}/TSP ratio) and by 10 years.[4] The function is then applied to adults 25 years old and up.

The final issue concerns the threshold. Abbey et al. focused on peak TSP readings because of their underlying model that repeated exposures to high readings are the problem rather than average exposures. As noted above, Abbey et al. found significant effects of mean concentrations on chronic bronchitis risks for persons exposed to 10-year average TSP concentrations of 60 $\mu g/m^3$ but also found a cut-off of 100 $\mu g/m^3$ for effects on an hourly basis. This implies that a sufficient number of peak hours were recorded over this period to contribute to chronic bronchitis risks even though the average readings were 60 $\mu g/m^3$. The issue for our study is whether peaks and averages are related in a comparable way in our reference environments to the areas in California examined by Abbey et al. If TSP readings in our environments are narrowly dispersed around 40, 50, or 60 $\mu g/m^3$, then transfer of this dose-response function to our reference environments would be misleading. Accordingly, based on Abbey et al.'s results that significant effects are not observed unless there are at least 10 days with TSP readings of 100 $\mu g/m^3$ or more, we will insist that this requirement be met in the reference environments before the dose-response relationship based on the mean is applied.

[3]This is calculated from $e^{\beta X}$ = relative risk = 1.36, where β is the estimated coefficient from the logit regression and X is the TSP concentration (60). Taking logs of both sides, $\beta X = \ln(1.36)$, and $\beta = 0.00512$.

[4]It is likely that the cases of chronic bronchitis were bunched up towards the end of the 10 year interval rather than evenly spread through it, as is implied by dividing by 10 to obtain an annual incidence rate. This procedure results in an upward bias to annual damage estimates given the effect of discounting.

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3. SULFUR DIOXIDE

3.1 REVIEW OF STUDIES

Effects of sulfur dioxide (SO_2) on the respiratory system have been observed after either short-term (less than 1-hour average) or longer term (24-hour average) exposures. In controlled laboratory experiments of human volunteers with asthma and others who appear to be sensitive to SO_2, short (5-10 minutes to 1 hour) exposures to SO_2 result in bronchoconstriction and increased lower respiratory symptoms at concentrations of 0.20 ppm and above (Sheppard et al. 1980, 1981). At concentrations near 0.20 ppm SO_2, this effect is observed only with exercise or voluntary hyperventilation. The current 1-hour standard in California is 0.25 ppm, while the federal 3-hour standard is 0.5 ppm. For a 24-hour averaging time, the California standard is 0.04 ppm, versus the Federal standard of 0.14 ppm.

Several recent epidemiologic studies indicate that changes in 24-hour average exposure to SO_2 may affect lung function, the incidence of respiratory symptoms and diseases, and risks of mortality. These studies have been conducted in different geographic locations and climates, and with different populations and covarying pollutants. Although many of these investigations also indicate that particulate matter or ozone was associated with these adverse health outcomes, several studies appear to show an effect of SO_2 where one from the other pollutants cannot be demonstrated. Furthermore, in some of the publications reporting an effect of both SO_2 and particulates, they are highly correlated, but in others, the correlation of the daily levels is only weak to moderate. Thus, it is possible to infer the possibility of an effect of SO_2 independent of particulate matter. This section focuses on effects related to 24-hour exposure to SO_2.

Epidemiologic studies undertaken in several locations indicate that SO_2, acting alone or as a surrogate for other sulfur-related species, is associated with increased risk of mortality. This includes studies in Athens (Hatzakis et al. 1986), France (Derriennic et al. 1989; Loewenstein et al. 1983), England (Chinn et al. 1981), and Poland (Krzyzanowski and Wojtyniak 1982). Respiratory symptoms and asthma exacerbation have also been associated with exposure to 24-hour average concentrations of SO_2. Unfortunately, most of the available studies do not provide dose-response functions. Below, a brief review of several relevant studies and available dose-response relationships are provided from respiratory morbidity.

Recent studies that provide quantitative evidence of an effect of SO_2 on symptoms include Charpin et al. (1988), Bates and Sizto (1983, 1987, 1989, 1990), Ponka (1990), Dodge et al. (1985), Schwartz et al. (1988) and Schwartz et al. (1991). Dose-response information can be generated from the latter two studies.

Charpin et al. (1988) examined the effects of SO_2 on respiratory symptoms in children in an area in proximity to a coal-fired power plant in France. The study involved 450 children aged 9 to 11 in eight communities, whose respiratory symptoms were logged in a daily diary for one month by their parents. Symptom scores were correlated with SO_2 and particle readings in each city. This analysis showed that only in the two most polluted cities (mean 24-hour SO_2 levels = 146 $\mu g/m^3$) were the correlations statistically significant. The incidence of both cough ($p < 0.05$) and wheezing ($p < 0.01$) were significantly related to 24-hour SO_2 readings. In the other six cities, with lower SO_2 readings (but lower, equal, and higher concentrations of respirable particulates relative to the first two cities), SO_2 was not correlated with symptoms. Annual averages of respirable particulate concentrations (particulate matter approximately 3.5 microns in diameter) were 67 $\mu g/m^3$ in the highest towns and ranged from 77 to 22 $\mu g/m^3$ in the other towns. However, neither particles nor daily temperature were correlated with symptoms during the one-month period of analysis. Since the particulate matter levels were similar in the two areas, this study suggests that SO_2 (or perhaps other sulfur compounds) was associated with adverse health outcomes. Unfortunately, dose-response relationships were not provided and cannot be estimated from the available data.

In a series of studies of hospital admissions in Southern Ontario, Canada, Bates and Sizto (1983, 1987, 1989) examined the daily effects of sulfur compounds (including both SO_2 and sulfates) and ozone. Summer and winter effects were disaggregated to reduce the impact of seasonal variations and influences. The 1983 study indicated an association in the summer months between hospital admissions related to respiratory disease and concentrations of SO_2 and ozone. This study took into account the potential confounding of temperature, humidity, and day of the week. Unfortunately, the high covariation between these two pollutants makes it difficult to determine the extent of any independent impact of SO_2. Subsequently, Bates and Sizto (1987) incorporated additional years of data and the measurement of ambient sulfates. The correlation analysis of the daily data indicated that during the summer, SO_2, ozone, and sulfate were related to both asthma and other respiratory-related hospital admissions. Again, however, daily concentrations of ozone, SO_2 and sulfates were highly correlated.

More recently, Bates et al. (1990) analyzed hospital emergency visits in Vancouver, British Columbia. The study involved emergency visits over a 28-month period from 1984 to 1986, including 8,300 visits per year for acute respiratory conditions. The data for both asthma and other respiratory-related visits were correlated with several air pollutants for each of three age groups (ages

1–14, 15–60, and 60+). Lower correlations among the pollutants were reported in this study, since Vancouver has much lower levels of ozone and sulfates (and most likely acid aerosols, as well) than Southern Ontario. Specifically, the correlations between SO_2 and three pollutant measures - ozone, sulfates, and coefficient of haze (COH) - were 0.23, 0.46, and 0.34, respectively. Analyses of these data indicate that for the summer months, the more significant correlations for the pollutants are with the age 15 to 60 subgroup, and indicate an association between asthma and total respiratory admissions and both SO_2 and sulfate. In the winter, SO_2 was associated with respiratory-related emergency room visits across all three age groups and is related to asthma admissions for the oldest subgroup. Sulfates were associated with asthma in the oldest subgroup, while ozone had little effect on emergency room visits. This study provides greater evidence for a sulfur-specific effect than the Ontario studies, which were more subject to confounding from ozone. No dose-response relationship is provided by the authors.

Pönkä (1990) investigated the association between weekly averages of air pollution and respiratory infections and absenteeism in children and adults in Helsinki, Finland. The mean of the 1-week average of 1-hour daily maxima for SO_2 was 50 $\mu g/m^3$, with a range of 25 to 125 $\mu g/m^3$, while TSP ranged from 48 to 123 $\mu g/m^3$. Temperature was inversely correlated with SO_2 ($p < 0.0001$). Health data were from 25 communal health centers handling 54% of all acute care problems. All social classes and age groups were included. Among the health outcomes measured were upper respiratory tract infections diagnosed at communal health centers, upper and lower respiratory tract infections in children attending day care centers, and absenteeism due to febrile illness among day care children, schoolchildren, and adults. Weekly averages of SO_2 and temperature were both associated with upper respiratory illness in children and with absenteeism in day care attendees and adults. After an attempt was made to standardize for temperature, only respiratory illness diagnosed at health centers remained correlated with SO_2. Analysis using TSP was not reported, although its correlation with SO_2 was high.

Samet et al. (1981) examined the daily association between air pollution and emergency room visits in Steubenville, Ohio. The 24-hour averages for SO_2 and TSP, which were highly correlated ($r = 0.69$), were 85 $\mu g/m^3$ and 156 ($\pm$ 123) $\mu g/m^3$, respectively. Small but significant associations were observed between emergency room visits related to respiratory disease and both SO_2 and TSP. Because of the high covariation between these two pollutants, an independent effect from SO_2 cannot be isolated. The effect on emergency room visits is calculated under the section for particulate matter, but there is some possibility that visits were related to some sulfur species.

Dodge et al. (1985) examined respiratory symptoms in an area with moderate levels of SO_2 and low levels of particulate sulfate, total suspended particulates (TSP) and fine particulates near an Arizona smelter. This study compares the health of 343 Mexican-American and non–Mexican-American children in third through fifth grades, living in four towns with varying pollution concentrations. The annual mean 24-hour average for SO_2 in the highest pollution area was 103 $\mu g/m^3$ versus less than 4 $\mu g/m^3$ in the cleanest area. TSP levels were approximately similar in the four towns ($\approx$ 52 $\mu g/m^3$) while sulfates varied from 10 $\mu g/m^3$ to 4.4 $\mu g/m^3$. The point prevalence (i.e., the number of persons with the condition at the time of the survey) of persistent cough and wheeze in each of the towns were reported on an intake questionnaire.

Among the sample of non–Mexican-Americans, persistent cough correlated significantly with the annual average of 24-hour SO_2 concentrations, but not with other air pollutants. Since the prevalence of parental smoking and gas stove use were not different across the cities, these factors could not explain the observed difference in the prevalence of cough. Thus, this study suggests that repeated exposure to low to moderate levels of SO_2 and sulfate concentrations are associated with a persistent cough. This study has particular relevance for determining an effect of SO_2 since particulate matter, a potential confounder, was at low concentrations and because in the arid Arizona climate, it is unlikely that SO_2 was a surrogate for sulfuric acid or fine particulate sulfates. Again, no dose-response function was provided by the authors.

3.2 ESTIMATES OF HEALTH EFFECTS

Schwartz et al. (1991) relate daily levels of SO_2 to respiratory symptoms among a sample of approximately 280 children in Watertown, Massachusetts, who were part of the Harvard Six-Cities Study. A daily diary completed by parents recorded several acute symptoms of their children including upper respiratory illness and cough. Annual average of the 24-hour SO_2 concentrations was approximately 27.8 ppm. The correlation among pollutants was not reported. A logistic regression was used to examine the relationship of pollution to these symptoms. Sulfur dioxide had a statistically significant association with cough. The effects of other pollutants were unclear from this primarily methodological article. Nevertheless, the results suggest the following:

$$\text{logit (cough)} = 0.0130 * SO_2.$$

The standard error of the estimated regression coefficient was 0.0059 and the mean incidence rate was one percent. Taking the derivatives, and substituting, we obtain the following functions for children:

upper change in the probability of cough per 1,000 children per day
$= 0.187 * SO_2$ ($\mu g/m^3$),

central change in the probability of cough per 1,000 children per day
$= 0.129 * SO_2$,

lower change in the probability of cough per 1,000 children per day
$= 0.070 * SO_2$.

Schwartz et al. (1988) examined the effects of air pollution among a population beginning nursing school in Los Angeles in the early 1970s. Daily diaries were completed and provided information on incidence of symptoms including cough, phlegm, and chest discomfort. Pollutants under investigation included oxidants, sulfur dioxide, nitrogen dioxide and carbon monoxide. In models corrected for autocorrelation, a significant association was found between SO_2 and chest discomfort. Daily concentrations of SO_2 averaged approximately 0.09 ppm. Specifically, the results indicated

> ***... change in the probability of cough per 1,000 children per day = 0.129 $*$ SO_2 ...***

logit (chest discomfort) $= 1.88 * SO_2$ (ppm),

or converted to SO_2 in micrograms per cubic meter:

logit (chest discomfort) $= 0.00072 * SO_2$ ($\mu g/m^3$).

The standard error of the estimated regression coefficient was 0.00036. Taking the derivatives, and substituting the mean rate of chest discomfort of 0.04, the following functions are obtained for adults:

> ***... central change in the probability of cases of chest discomfort per day per 1,000 = 0.028 * SO_2.***

upper change in the probability of cases of chest discomfort per day per 1,000 $= 0.041 * SO_2$ ($\mu g/m^3$),

central change in the probability of cases of chest discomfort per day per 1,000 $= 0.028 * SO_2$,

lower change in the probability of cases of chest discomfort per day per 1,000 $= 0.0145 * SO_2$.

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4. LEAD

4.1 REVIEW OF EFFECTS OF BLOOD LEAD ON BLOOD PRESSURE

Exposure to ambient lead occurs primarily from leaded fuel in automobiles and from stationary sources including primary and secondary smelters and battery recycling plants. Once absorbed, lead is distributed throughout the body and is only slowly removed. Lead has been reported to cause many different health effects. Based on current knowledge, clinical effects that may occur at low blood lead concentrations include neurodevelopmental effects in children, and hypertension and related cardiovascular conditions in adults. A thorough review of health outcomes associated with lead exposure is provided by U.S. EPA (1990a,b) and ATSDR (1990). Of particular interest, low levels of lead in the blood have been associated with increased blood pressure and hypertension in adults. The relationship between both systolic and diastolic blood pressure and blood lead appears to hold across a wide range of blood lead values, with no apparent threshold. Most of the studies indicate an effect of blood lead only on males, although an effect on females cannot be completely ruled out.

Since the studies on blood pressure effects use lead concentration in blood as their indicator of exposure, it is necessary to relate changes in air lead to subsequent concentrations of blood lead. For this purpose an "aggregate" model was used, relating ambient concentrations of lead to blood lead, both directly through inhalation and indirectly through other media such as soil and dust. This model indicates a strong and consistent association between ambient concentrations of lead in the air and subsequently measured blood lead in adults. The aggregate model, as well as studies in experimental exposure chambers, were used to estimate the effects of changes in ambient lead levels on the subsequent blood lead levels in adults.

The association between lead and hypertension was first observed in animals. This effect has been shown across a range of doses and in several species (Victery 1988), and was recently examined in occupational and population-based epidemiological studies. The population-based studies will be briefly reviewed here.

Several investigators have used NHANES II data to investigate the relationship between blood lead level and hypertension (Harlan et al. 1985; Pirkle et al. 1985; Schwartz, 1985a,b, 1986a,b). NHANES II is a large, individual-level database that includes information on a variety of potentially confounding factors. Therefore, these studies avoided common study design problems (e.g., healthy worker effect, workplace exposures to other toxic agents, selection bias, and problems of control group selection).

Using these data, Harlan et al. (1985) demonstrated statistically significant linear associations ($p < 0.001$) between blood lead concentrations and both systolic and diastolic blood pressure among males aged 12 to 74 years. Further analyses reported by Pirkle et al. (1985) focused on white males, aged 40 to 59 years. This age group was used to reduce any potential confounding effects of age on blood pressure. In the subgroup studied, significant associations were found between blood lead and blood pressure even after controlling for all risk factors known to be correlated with blood pressure. Furthermore, no threshold for the effect was observed across a blood lead range of 7 to 34 μg/dL.

During the 4 years in which NHANES II data were collected, there were declines nationally for both blood lead and hypertension. In addition, sites were sampled without revisitation at different times. The variations observed in these sampling sites over time could be due to these national declines, resulting in the observed association. Schwartz (1985a,b, 1986a,b, 1988) reanalyzed the data of Pirkle et al. and showed that the association decreased but remained significant for both systolic and diastolic blood pressure when adjusted for site.

Landis and Flegal (1988) examined the robustness of the association of lead and diastolic blood pressure in males aged 12–74, adjusting for age, body mass index, and sampling site. The association remained statistically significant at the $p < 0.05$ level across 478 subgroups. They concluded that the NHANES II findings could not be dismissed as due to concurrent secular trends across the 4-year survey period. Furthermore, this analysis indicates that the blood lead–blood pressure relationship holds for all adult males, age 20 to 70.

Another major population-based study was conducted by Pocock et al. (1985, 1988) using data from the British Regional Heart Study (BRHS). In 7,371 men aged 40–49 there was a small but significant association of both systolic ($p = 0.03$) and diastolic ($p < 0.001$) blood pressure with blood lead level when adjusted for site in multiple regression analyses. The analyses included an adjustment for body mass, age, alcohol intake, social class, and observer. The magnitude of the effect of blood lead on diastolic blood pressure from this British data was very similar to those involving the analysis of NHANES II. Specifically, an estimated mean increase of about 1.5–3.0 mm Hg in diastolic blood pressure appears to occur for every doubling of blood lead concentration in adult males. Slightly smaller increases have been observed in adult females. The relationship between diastolic blood pressure and blood lead level appears to hold across a wide range of blood-lead values extending possibly down to at least as low as 7 μg/dL for men.

Additional support for the magnitude of the effect of blood lead on diastolic blood pressure is provided by a study of 342 San Francisco bus drivers (Sharp et al. 1988). Multiple regression analysis was used to examine this association while controlling for age, body mass index, sex, race, and caffeine intake. The results indicate a change of 1.83 to 2.45 mm Hg in diastolic blood pressure associated with a unit increase in ln(μg/dL). Additional analysis of this data (Sharp et al., 1990) showed that the effect of blood lead on blood pressure was greater for the Blacks than non-Blacks.

Data from the Canada Health Survey support the above findings (Neri et al. 1988). Over 2,000 male and female subjects aged 25 years to 64 years were studied for 10 months. The authors observed a small but statistically significant relationship ($p < 0.001$) between diastolic blood pressure and blood lead level.

Several investigators have attempted to predict the impact of the observed increase in blood pressure on other significant cardiovascular events. For example, large-scale epidemiological studies including the Pooling Project Research Group (1978) and the Framingham study (McGee and Gordon 1976) have shown that elevated blood pressure increases the risk of cardiovascular disease.

The Pooling Project combined the results of five longitudinal studies that examined the incidence of coronary heart disease (CHD) in middle-aged white men. The first onset of CHD, defined as fatal or nonfatal myocardial infarction and sudden CHD death, was measured over a 10-year period. Using logistic analysis, the research indicted that smoking, serum cholesterol, and diastolic blood pressure were major risk factors in the incidence of CHD (Pooling Project Research Group 1978). The Framingham study (McGee and Gordon 1976) was one of the studies included in the Pooling Project. Besides estimating the incidence of CHD, this study of white middle-aged men considered the incidence of deaths from all causes. Diastolic blood pressure was again identified as a significant predictor of all-cause mortality. These studies were used by Pirkle (1985) to estimate the quantitative effects of blood lead on diastolic blood pressure and subsequent CHD and mortality.

Using coefficients relating hypertension to other cardiovascular events derived from the Pooling Project and the Framingham studies, Pirkle et al. (1985) estimated that the 37% drop in blood lead levels from 1976-1980 resulted in a 4.7% decrease in incidence of fatal and nonfatal myocardial infarction over 10 years, a 6.7% decrease in the incidence of fatal and nonfatal strokes over 10 years, and a 5.5% decrease in the incidence of death from all causes over 11.5 years. In addition, as a result of the blood lead decrease observed in NHANES II, they estimated that the number of white males 40–59 with hypertension (as measured by diastolic blood pressure greater than 90) decreased by 17.5%.

Additional support for an association of lead with cardiovascular disease is provided by Schwartz (1989), where the relationship of abnormal electrocardiograms indicative of left ventricular hypertrophy (LVH) and lead level was examined. Using logistic regression, a statistically significant ($p < 0.01$) coefficient was found between blood lead levels and increased prevalence of LVH was found after adjusting for age, race, sex, and smoking.

4.2 THE CONTRIBUTION OF AIRBORNE LEAD TO HEALTH RISKS

For this analysis, it is necessary to establish a direct association between changes in lead emitted into the air and subsequent concentrations of lead in the blood of adults. There are two alternative methods for determining the relationship between air lead and blood lead. First, a disaggregate model can be developed in which the separate contribution of all potential sources of lead -- air, soil, dust, food, and water are determined. In order to arrive at an integrated lead exposure function, this method requires the estimation of the relationship between air emissions and subsequent concentrations of lead in other media, determination of exposure scenarios for all sources of lead (i.e., the amount of lead in air, food, water, soil, etc. and the amounts subsequently inhaled and ingested by both children and adults), and absorption factors for inhalation and ingestion.

A second method, an aggregate model, uses a single variable, air lead, to serve as an index for lead emitted into the air and ultimately affecting individuals though all possible media. This method relates air emissions to the subsequent change in blood lead through many different exposure routes. In a review of nine different studies covering 13 different populations of children, Brunekreef (1984) found high correlations between air lead and the lead found in dustfall ($r = 0.92$), soil ($r = 0.62$), and housedust ($r = 0.88$). Since we are interested in the ultimate impact that changes in air lead emissions will have on blood lead and subsequent health effects, the aggregate method, which requires fewer assumptions and extrapolations, is preferred and will be used to develop dose-response estimates for air lead.

There are many studies that relate ambient lead to blood lead in adults. Results are obtained from both observational studies in the field and studies in experimental exposure chambers. The latter are particularly useful in providing estimates of the blood lead/air lead slope since they may involve longitudinal examination of individuals exposed to relatively small changes in air lead under controlled conditions. For example, Griffin et al. (1975) exposed 24 adult volunteers to two different doses over a 16-week period in an experimental chamber. The pooled slope for all individuals, as determined by reanalysis of the data by EPA (1986), was 1.8.

Three other experimental studies (Rabinowitz et al. 1977; Kehoe 1961; Chamberlain 1983) also provide data that can be used to develop slope estimates. In their reanalysis of these data, EPA (1986) reported slopes of 1.6 to 3.6 for the

Rabinowitz et al. data, -0.34 to 2.60 for the Kehoe data, and 2.7 for the Chamberlain data. Although there are several distinct advantages to using these experimental studies, each has some deficiencies. Among these are the difficulty in fully determining either the lead exposure received prior to starting the controlled experiment or received outside of the chamber. Nevertheless, the weight of evidence regarding the magnitude of the slope, considering the studies together, is compelling.

These experimental studies are also supported by population studies. Azar et al. (1975) recorded air and blood lead for 30 individuals in each of five areas over a 2- to 4-week period. A linear regression of blood lead versus air lead for all of the subjects was highly significant and generated a slope of 1.75. Finally, Snee and Pfeifer (1983) reanalyzed the data from five previous observational studies and obtained a pooled slope of 1.4.

Considering both the experimental and observational information, we assume a blood lead/air lead slope of 2.0 for adults.

4.3 DOSE-RESPONSE FUNCTIONS FOR CARDIOVASCULAR EFFECTS

This section provides estimates of the change in health effects associated with changes in ambient lead concentration. The methodology used in this section is similar to that used previously by the U.S. EPA in its analyses of the effects of reducing lead in gasoline (Schwartz et al. 1985) and the effects of reducing lead in drinking water (Levin 1986). Dose-response functions are provided to estimate the effect of a change in air lead on the likelihood of hypertension (diastolic blood pressure ≥90 mm Hg), and the effects of more serious health outcomes including myocardial infarction (heart attacks) and mortality.

To estimate the change in hypertension related to air lead, we use dose-response information provided by Schwartz et al. (1985). Additional documentation for this estimation process is found in Pirkle et al. (1985) and Brennan et al. (1986). The estimates are based on a logistic regression of the probability of hypertension versus blood lead. The original estimates were conducted for the subset of the population of adult males age 40 to 59. However, sensitivity analysis conducted by Schwartz et al. (1985) and analysis by Landis and Flegal (1988) indicate that the blood lead–blood pressure relationship holds for all adult males, age 20 to 70. Therefore, the dose-response functions can be applied to all adult males.

To develop the estimate, the mean values of the NHANES II sample for covariates such as body mass, albumin, hemoglobin, vitamin C, dietary potassium, total carbohydrates, and recreational exercise were used. We determined how the risk of hypertension changes when we reduce ambient lead concentrations by 1.00 $\mu g/m^3$, resulting in a 2.00 $\mu g/dL$ change in blood lead. We then can predict the probability of hypertension as a function of log of blood lead by

We then can predict the probability of hypertension as a function of log of blood lead ...

$$\text{change in H} = (1 + \exp -(-2.744 + 0.793\ (\ln PbB2)))^{-1} - (1 + \exp -(-2.744 + 0.793\ (\ln PbB1)))^{-1}$$

where:

H = the probability of hypertension,
PbB1 = initial blood lead level ($\mu g/dL$),
PbB2 = new blood lead level ($\mu g/dL$).

Using the blood lead/air lead relationship, the change in the probability of hypertension can be expressed as :

$$\text{change in H} = (1 + \exp -(-2.744 + 0.793\ (\ln 2PbA2)))^{-1} - (1 + \exp -(-2.744 + 0.793\ (\ln 2PbA1)))^{-1}$$

where PbA1 and PbA2 = initial and new air lead levels ($\mu g/m^3$).

Although this model is nonlinear, sensitivity analysis indicates that the results are relatively insensitive to assumptions about either the initial blood lead level or the change in blood lead.

Since the risk of all-cause mortality and of heart attacks was provided as a function of diastolic blood pressure, the association between blood lead and diastolic blood pressure is needed. Based on Pirkle et al. (1985) and U.S. EPA (1986), we use the following relationship:

$$\text{change in DBP} = 2.74\ (\ln PbB2 - \ln PbB1)$$

where:

DBP = diastolic blood pressure.

Again, the above equation can be expressed in terms of a change in ambient air lead. We obtain change in DBP = 2.74(ln 2PbA2 - ln 2PbA1).

Using the sample means of the independent variables in the Pooling Project, we can express the relationship between the change in blood pressure and the change in the probability of a CHD event in the following 10 years as

$$\text{change in PR(CHD)} = (1 + \exp -(-4.996 + 0.030365\,(\text{DBP2})))^{-1} - (1 + \exp -(-4.996 + 0.030365\,(\text{DBP1})))^{-1},$$

where PR(CHD) = the 10-year probability of a coronary heart disease event, and DBP1 and DBP2 = initial and new diastolic blood pressure levels.

The Framingham study (Shurtleff 1974) can be used to estimate the change in mortality due to the change in diastolic blood pressure. Controlling for serum cholesterol levels and smoking, the association can be estimated by

$$\text{change in PR(MORT)} = (1 + \exp -(-5.3158 + 0.03516(\text{DBP2})))^{-1} - (1 + \exp -(-5.3158 + 0.03516(\text{DBP1})))^{-1},$$

where PR(MORT) = the 12 years probability of death from all causes.

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5. NITROGEN DIOXIDE

The epidemiologic evidence for an effect of nitrogen dioxide (NO_2) on respiratory symptoms is more uncertain than the effects of the criteria pollutants described above. Although many studies demonstrate an effect on children from indoor exposure to gas stoves, a primary source of indoor NO_2, effects from outdoor NO_2 on either children or adults are rarely observed. For this review, results from studies of gas stoves (usually associated with a 30-$\mu g/m^3$ increase in exposure to NO_2) are not used. Instead, we rely on the study by Schwartz and Zeger (1990) that relates outdoor NO_2 to respiratory symptoms.

This research examined the effects of air pollution among a population beginning nursing school in Los Angeles in the early 1970s. Daily diaries were completed and provided information on incidence of symptoms including cough, phlegm, and chest discomfort. Pollutants under investigation included oxidants, sulfur dioxide, nitrogen dioxide and carbon monoxide. In models corrected for autocorrelation, a significant association was found between NO_2 and phlegm. Daily concentrations of NO_2 averaged approximately 0.13 ppm. Specifically, the results indicated:

$$\text{logit (phlegm)} = 0.843 * NO_2 \text{ (ppm)}.$$

The standard error of the estimated regression coefficient was 0.343. Taking the derivatives, and substituting the mean incidence rate of phlegm of 0.0345, the following functions are obtained for adults:

upper change in the probability of cases of phlegm per day per person $= 0.0395 * NO_2\ (\mu g/m^3)$,

central change in the probability of cases of phlegm per day per person $= 0.028 * NO_2$,

lower change in the probability of cases of phlegm per day per person $= 0.0165 * NO_2$.

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PAPER 7[1]

HEALTH EFFECTS OF RADON

Exposure of people to radon has taken on increased interest during the last decade because of the understanding that buildings can serve to trap radon and its daughters, and thereby build up undesirable concentrations of these radioactive elements. Numerous studies of underground miners (often uranium miners) have shown an increased risk of lung cancer in comparison with nonexposed populations. Laboratory animals exposed to radon daughters also develop lung cancer. The abundant epidemiological and experimental data have established the carcinogenicity of radon progeny. Those observations are of considerable importance, because uranium, from which radon and its progeny arise, is ubiquitous in the earth's crust, including coal mines.

Risk estimates of the health effects of long-term exposures at relatively low levels require continued development, especially to address the potential health effects of radon and radon daughters in homes and occupational settings where the exposure levels are less than levels in underground uranium and other metal mines that have been the subject of epidemiological studies.

Two approaches can be used to characterize the lung-cancer risks associated with radon-daughter exposure: mathematical representations of the respiratory tract that model radiation doses to target cells and epidemiological investigation of exposed populations, mainly underground uranium miners. The mathematically-based dosimetric approach provides an estimate of lung-cancer risk related to radon-daughter exposure based specifically on modeling of the dose to target cells. The various dosimetric models all require assumptions, some of which are not subject to direct verification, as to breathing rates; the deposition of radon daughters in the respiratory tract; and the type, nature, and location of the target cells for cancer induction.

The most recent large committee effort drawn together to evaluate this issue was sponsored by the National Research Council. Its committee, known as the BEIR IV (Committee on the Biological Effects of Ionizing Radiation)

[1]Based largely on a working paper by C. Easterly.

chose not to use dosimetric models for calculating lung-cancer risk estimates in their report. The lung-cancer risk estimates for radon-daughter exposure derived by the BEIR IV committee are based solely on epidemiological evidence. However, the results of dose models were used to extrapolate lung-cancer risks derived from the epidemiological studies of underground miners to the general population receiving lesser exposure levels.

The committee preferred a direct epidemiological approach, because the studies of radon-daughter-exposed miners provided a direct assessment of human health effects. This decision coincides with our preferred source of data (i.e., direct estimates of risk based on human data) and thus we have chosen to adapt the BEIR IV results for the fuel cycle externalities study. Although each of the epidemiological studies that the committee assessed has limitations, the approach of a combined analysis of major data sets permitted a comprehensive assessment of the health risks associated with radon-daughter exposure and of other factors that influence the risk, such as age and time since exposure. In analyzing the data, the committee obtained data from four of the principal studies of radon-exposure and developed risk models for lung cancer based on analyses of these data. By means of statistical regression techniques appropriate for survival-time data, the committee found that the probability of dying of lung cancer at age a in the combined cohorts was best described by the following expression:

...the BEIR IV committee obtained data from four of the principal studies of radon-exposure and developed risk models for lung cancer ...

$$r(a) = r_o(a)[1 + 0.025\gamma(a)\{W_1 + 0.5W_2\}],$$

where $r_o(a)$ is the age-specific background lung-cancer mortality rate; $\gamma(a)$ is 1.2 when age a is less than 55 yr, 1.0 when a is 55-64 and 0.4 when a is 65 yr or more; W_1 is Working Level Month (WLM) [a measure of exposure to radon and radon daughters][2] incurred between 5 and 15 yr before this age; and W_2 is WLM incurred 15 yr or more before this age. The above relationship has been derived for underground miners. Measures of exposure

[2]A Working Level (WL) is any combination of short lived radon daughters in 1 liter of air that will result in the ultimate emission of 1.3×10^5 MeV of potential alpha energy. This number was chosen because it is approximately the alpha energy released from the decay of daughters in equilibrium with 100 picocuries of ^{222}Ra. A Working Level Month (WLM) is that exposure resulting from inhalation of air with a concentration of 1 working level of radon daughters for 170 working hours.

for evaluating effects associated with the coal fuel cycle should also be in Working Level Months, or the measures should be convertible to WLM.

In the event that radon and daughter exposures provide significant exposures to surface miners and/or members of the public, in order to more accurately evaluate risk, modifications would be required with respect to the equilibrium ratios of the daughters and the particles onto which the daughters attach and are thereby inspired. Further, several assumptions would be required relative to exposure times and differences in the cohorts. However, because of the effort required to obtain greater accuracy, modifications to the above relationship will not be pursued until evidence is obtained that suggests that exposure to surface miners and/or members of the public makes up approximately 10% of either group's total risk from the coal fuel cycle.

The relationship provided in the equation results in an age-specific lung cancer rate. In order to be useful in the present analysis, this must be converted to lost years of life or to an increase in numbers of cancers. Thus it will be necessary to know the workforce size, typical ages of work, (i.e., age 25-55 yr), and nominal exposure levels (in WLM).

The BEIR IV committee reviewed risk estimates made by other scientific groups, including the National Research Council (the BEIR III report). Comparisons between all epidemiology studies are not possible because of large differences in the populations assumed to be at risk, for example, duration of exposure and smoking prevalence. The BEIR IV results are near the middle of range of risks between the BEIR III (NRC 1980), the UNSCEAR (1977) and the NCRP (1984) reports. The major factor separating the BEIR IV from the BEIR III lifetime risks is that the BEIR IV work is based on a modified relative risk model that takes into account the reduced risk at age 65 or greater and the smaller effectiveness of exposures occurring 15 years or more in the past.

The risk of lung-cancer mortality associated with long-term exposure to radon daughters is a function of the duration of exposure and, because of competing risks of death from other causes, of age as well. A valuable measure of mortality is the number of years of life lost because of exposure, $L_0 - L_e$, where L_0 is expected years of life at birth for a nonexposed subject, and L_e is expected years for an exposed subject. Note that $L_0 - L_e$ is the average for a population of exposed persons. The number of years of life lost by those who actually die of lung cancer is usually much greater – typically, about 15 yr for males and about 18 yr for females.

The BEIR IV committee has calculated age- and exposure-specific risk measures including $L_0 - L_e$ for various magnitudes of annual exposure using the male and female 1980-1984 U.S. mortality rates. Mortality rates based on a specific period in the past can only approximate future rates in a population whose total mortality rates vary over time. Moreover, the age patterns and the relative importance of lung cancer as a cause of death are expected to change as smoking becomes less prevalent.

Lifetime risk of lung cancer and expected years of life lost are related to a specified exposure profile for persons who are followed from birth. Those measures do not, however, provide information about lung-cancer risk after survival to some specified age. Total life expectancy when a person is known to be alive at a given age is greater than that at birth, because the rigors of the intervening years have been survived. The following table (Table 1), prepared by the BEIR IV committee, compares risk measures computed conditionally on survival to specific ages for males in the general population. For this assessment, we chose to use general rates since this does not require additional assumptions relative to smoking habits. The column headed 0 gives baseline values without information on age. Table 1 shows the effects on the years of life lost if exposure were to cease at the conditional age known to be alive. The health benefits of eliminating exposure are seen to be substantial into the middle ages. Table 2 shows the effects on lifetime lung cancer risks (R_e) if exposures were to cease at the conditional age known to be alive. Similar tables have been derived for both sexes for smokers and non-smokers.

Application of the relative risk model to the health assessment for the coal fuel cycle requires several assumptions:

(1) Miners begin work at age 20 and continue working until age 60.

(2) The exposure rate to radon and daughters is between 0.1 and 0.2 WLM/y. [We know of no measurements for radon in surface mines; however, the 0.1 to 0.2 WLM/y should be an upper limit. O'Riordam et al. (1981) report levels of between 0.12 and 0.24 WLM/y in underground coal mines.]

(3) Smoking habits and mortality rates remain the same as during the 1980-84 period.

Given these assumptions, the time since exposure model (Table 1) predicts an average of between 0.03 and 0.07 years of life lost per worker. Alternatively, using Table 2, the lifetime risk is between 0.069 and 0.072. Given that the baseline risk is 0.067, the incremental risk is between 0.002 and 0.005. This means that 2 to 5 lung cancer cases are expected per thousand workers with a work history exposure averaging 0.1 to 0.2 WLM/y over the ages of 20 to 60 years. From the analysis of coal mine labor, it is expected that somewhat less than 200 persons would be required to maintain a reference coal plant. Thus, radon exposure might be expected to result, as an upper limit, in one additional lung cancer case over a 40 year period.

Table 1. Radon exposure: years of life lost (L_o - L_c) to lung cancer for males by age started and age exposure ends[a]

	Age (yr) Exposure Ends								
Age (yr) Started	10	20	30	40	50	60	70	80	110
					Exposure Rate = 0.10 WLM/yr				
0	0.01	0.02	0.03	0.04	0.04	0.05	0.05	0.05	0.05
10		0.01	0.02	0.03	0.04	0.04	0.05	0.05	0.05
20			0.01	0.02	0.03	0.03	0.04	0.04	0.04
30				0.01	0.02	0.03	0.03	0.03	0.03
40					0.01	0.02	0.02	0.02	0.02
50						0.01	0.01	0.01	0.01
60							0.00	0.00	0.00
					Exposure Rate = 0.20 WLM/yr				
0	0.02	0.03	0.05	0.07	0.09	0.10	0.11	0.11	0.11
10		0.02	0.03	0.05	0.07	0.09	0.09	0.09	0.09
20			0.02	0.04	0.06	0.07	0.07	0.07	0.07
30				0.02	0.04	0.05	0.06	0.06	0.06
40					0.02	0.03	0.04	0.04	0.04
50						0.01	0.02	0.02	0.02
60							0.00	0.01	0.01
					Exposure Rate = 0.50 WLM/yr				
0	0.04	0.08	0.13	0.17	0.22	0.26	0.27	0.27	0.27
10		0.04	0.09	0.13	0.18	0.21	0.22	0.23	0.23
20			0.04	0.09	0.14	0.17	0.18	0.18	0.18
30				0.05	0.10	0.13	0.14	0.14	0.14
40					0.05	0.08	0.09	0.09	0.09
50						0.03	0.04	0.05	0.05
60							0.01	0.01	0.01
					Exposure Rate = 1.00 WLM/yr				
0	0.08	0.17	0.25	0.35	0.44	0.51	0.53	0.53	0.53
10		0.08	0.17	0.26	0.36	0.42	0.44	0.45	0.45
20			0.09	0.18	0.28	0.34	0.36	0.37	0.37
30				0.09	0.19	0.26	0.28	0.28	0.28
40					0.10	0.16	0.18	0.19	0.19
50						0.06	0.09	0.09	0.09
60							0.02	0.03	0.03

Table 1. (cont'd)

	Age (yr) Exposure Ends								
Age (yr) Started	10	20	30	40	50	60	70	80	110
Exposure Rate = 4.00 WLM/yr									
0	0.33	0.66	0.99	1.35	1.71	1.93	2.00	2.02	2.02
10		0.33	0.67	1.03	1.40	1.63	1.71	1.72	1.72
20			0.34	0.71	1.09	1.32	1.40	1.42	1.41
30				0.38	0.76	1.00	1.08	1.10	1.09
40					0.39	0.64	0.72	0.74	0.74
50						0.26	0.34	0.36	0.35
60							0.09	0.10	0.10
Exposure Rate = 10.00 WLM/yr									
0	0.82	1.61	2.37	3.18	3.96	4.42	4.56	4.58	4.58
10		0.82	1.63	2.47	3.29	3.78	3.93	3.96	3.95
20			0.84	1.73	2.59	3.11	3.26	3.30	3.29
30				0.93	1.84	2.39	2.55	2.59	2.59
40					0.96	1.55	1.73	1.77	1.76
50						0.63	0.83	0.87	0.86
60							0.21	0.26	0.25

[a]Estimated with the BEIR IV time since exposure model. R_e includes R_o, the baseline risk for males in the 1981-84 U.S. population, 0.067; the expected lifetime of males is 69.7 yr.

Source: Adapted from Committee on the Biological Effects of Ionizing Radiations, Board on Radiation Effects Research, Commission on Life Sciences, National Research Council, 1988. Health Risks of Radon and Other Internally Deposited Alpha-Emitters. BEIR IV. National Academy Press, Washington, D.C.

Table 2. Radon exposure: lifetime risks (R_e) for lung cancer for males by age started and age exposure ends[a]

Age (yr) Started	Age (yr) Exposure Ends: 10	20	30	40	50	60	70	80	110
Exposure Rate = 0.10 WLM/yr									
0	0.068	0.068	0.069	0.069	0.070	0.071	0.071	0.071	0.071
10		0.068	0.068	0.069	0.070	0.070	0.070	0.070	0.070
20			0.068	0.068	0.069	0.069	0.070	0.070	0.070
30				0.068	0.068	0.069	0.069	0.069	0.069
40					0.068	0.068	0.069	0.069	0.069
50						0.068	0.068	0.068	0.068
60							0.068	0.068	0.068
Exposure Rate = 0.20 WLM/yr									
0	0.068	0.069	0.070	0.072	0.073	0.074	0.074	0.074	0.074
10		0.068	0.069	0.070	0.072	0.073	0.073	0.073	0.073
20			0.068	0.069	0.071	0.072	0.072	0.072	0.072
30				0.068	0.070	0.071	0.071	0.071	0.071
40					0.069	0.070	0.070	0.070	0.070
50						0.068	0.069	0.069	0.069
60							0.068	0.068	0.068
Exposure Rate = 0.50 WLM/yr									
0	0.070	0.072	0.075	0.078	0.081	0.083	0.084	0.085	0.085
10		0.070	0.072	0.075	0.078	0.081	0.082	0.082	0.082
20			0.070	0.073	0.076	0.078	0.079	0.080	0.080
30				0.070	0.073	0.075	0.077	0.077	0.077
40					0.070	0.073	0.074	0.074	0.074
50						0.070	0.071	0.072	0.072
60							0.069	0.069	0.069
Exposure Rate = 1.00 WLM/yr									
0	0.072	0.077	0.083	0.088	0.094	0.098	0.101	0.102	0.102
10		0.072	0.078	0.083	0.089	0.093	0.096	0.097	0.097
20			0.072	0.078	0.084	0.089	0.091	0.092	0.092
30				0.073	0.079	0.083	0.086	0.087	0.087
40					0.073	0.078	0.081	0.082	0.082
50						0.072	0.075	0.076	0.076
60							0.070	0.071	0.071

Table 2. (cont'd)

Age (yr) Started	Age (yr) Exposure Ends: 10	20	30	40	50	60	70	80	110
					Exposure Rate = 4.00 WLM/yr				
0	0.87	0.107	0.126	0.146	0.167	0.183	0.192	0.195	0.195
10		0.087	0.107	0.128	0.149	0.166	0.175	0.178	0.178
20			0.088	0.109	0.131	0.148	0.157	0.160	0.161
30				0.089	0.112	0.130	0.139	0.142	0.143
40					0.091	0.109	0.119	0.122	0.123
50						0.087	0.096	0.100	0.100
60							0.077	0.081	0.082
					Exposure Rate = 10.00 WLM/yr				
0	0.117	0.162	0.206	0.206	0.291	0.323	0.338	0.343	0.344
10		0.117	0.163	0.209	0.255	0.289	0.305	0.311	0.312
20			0.117	0.166	0.216	0.253	0.270	0.276	0.277
30				0.120	0.173	0.213	0.232	0.239	0.240
40					0.125	0.168	0.188	0.196	0.197
50						0.114	0.137	0.145	0.146
60							0.092	0.101	0.102

[a]Estimated with the BEIR IV time since exposure model. R_e includes R_o, the baseline risk for males in the 1981-84 U.S. population, 0.067; the expected lifetime of males is 69.7 yr.

Source: Adapted from Committee on the Biological Effects of Ionizing Radiations, Board on Radiation Effects Research, Commission on Life Sciences, National Research Council, 1988. Health Risks of Radon and Other Internally Deposited Alpha-Emitters. BEIR IV. National Academy Press, Washington, D.C.

Radon-Lung Cancer NUSAP[3]

Numerical Information: years of life lost/cancer risk

Units

Measurement Units: per WLM/yr for period of work history

Statistical Unit: mean (meta analysis)

Spread

Confidence Level: 95%

corresponding to ± 2 standard errors

Upper Bound: x 1.7
Lower Bound: ÷ 1.7

A 67% confidence level corresponds to the commonly used method of expressing uncertainty ± 1 standard error. The uncertainty at a 95% level of confidence, corresponding to ± 2 standard errors, would be represented by multiplication and division by 1.7.

Assessment

Informative Value Based on Spread: LOW

The analysis used provides improvements to previous analysis, but not radically so. It takes an epidemiologically-based risk approach rather than dosimetric.

Informative Value Based on Application: MEDIUM

Generalizability to Other Applications: HIGH

Robustness of Value over Time: MEDIUM

Analysis depends on population-based data (1980-84), and over a period of several decades this will change.

Pedigree (credibility of the entry's origin)

Theoretical Basis: GOOD

Mostly based on the preferred data source, epidemiological studies.

Data Inputs: EXCELLENT

Estimation Methods: EXCELLENT

Estimation Metric: GOOD

[3]Refer to Part VI of this report for a description of the NUSAP method for describing the quality of information and data.

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PAPER 8[1]

HEALTH IMPACT FROM LEAD: IQ DECREMENT

1. NEUROTOXICITY OF LEAD

Recent data have demonstrated health effects of lead in children at doses previously believed to be harmless. Childhood lead poisoning is among the more important of the man-made diseases. Its origin, unlike many other important diseases, is known. The neurotoxic properties of lead at high doses have been known for at least a century, and in 1943 (Byers and Lord 1943) there was the first suggestion that childhood exposure to lead below the dose sufficient to produce clinical encephalopathy was associated with deficits in psychological function. However, in spite of the large number of studies investigating the relationship of low-levels of lead exposure to health effects, the picture of the exposure-response relationship is not yet clear for any of the chronic endpoints.

Lead can enter the body upon breathing air with lead-containing dust or lead particles. Almost all of the lead in the lungs enters the blood and moves to other parts of the body. In adults, very little of the amount of lead swallowed in food, beverages, water and dust or soil enters the blood from the gastrointestinal tract and moves to other body parts. Much less lead enters the body through the skin than through the lungs or gastrointestinal tract. Regardless of how lead enters the body, most of it is stored in bone, rather than soft tissue.

The effects of lead, once it is in the body, are the same no matter how it enters. Exposure to lead is especially dangerous for unborn children, with high levels associated with premature birth and low birth weight. Young children are at risk because they swallow lead when they put toys or other objects soiled with lead-containing dirt in their mouths. A greater proportion of the lead ingested by children enters their bodies than enters the bodies of adults. For infants and young children, lead exposure has been shown in some studies to decrease intelligence (IQ) scores, slow growth, and cause hearing problems. These effects can last as children get older and can be permanent. Realistic risk assessment for lead-induced neurobehavioral deficit in environmentally exposed children must stem from consistent results from

[1]Based largely on a working paper by C. Easterly.

independent studies, as well as the documentation of dose-response relationships. At present, studies investigating such effects have not been definitive. However, taken as a whole, there is growing support for the absence of a threshold for adverse effects in exposed children.

2. EARLIER STUDIES

De la Burde and Choate (1972) found a mean Stanford-Binet IQ decrement of 5 points, fine motor dysfunction, and altered behavioral profiles in 70 preschool children exhibiting pica for paint and plaster and elevated blood lead levels (>30 μg/dL, mean = 58 μg/dL), as compared with results for matched control subjects not engaging in pica for paint and plaster. A follow-up study on these children at 7 to 8 years of age (de la Burde and Choate 1975) reported a mean Wechsler Intelligence Scale of Children (WISC) Full Scale IQ decrement of 3 points and impairment in learning and behavior, despite decreases in blood lead levels since the original study.

Rummo (1974) observed hyperactivity and a decrement of ~16 IQ points on the McCarthy General Cognitive Index (GCI) among children who had previously had encephalopathy and whose mean blood lead levels at the time of encephalopathy were 88 μg/dL. Asymptomatic children with long-term lead exposures and mean blood lead levels of 68 μg/dL had an average IQ decrement of 5 points on the McCarthy GCI, and their scores on several McCarthy Subscales were generally lower than those for controls, but were not statistically significantly different (at $P < 0.05$), while children with short-term exposure and blood lead levels of 60 μg/dL did not differ from controls. Blood lead levels in controls did not exceed 40 μg/dL (mean 23 μg/dL).

A study of 158 first- and second-grade children by Needleman et al. (1979) provides evidence for the association of full-scale IQ deficits of ~4 points and other neurobehavioral defects with tooth (dentine) lead values that exceed 20 to 30 ppm. Corresponding average blood lead values would probably be in the range of 30 to 50 μg/dL (EPA 1986). The study has been reanalyzed in additional reports (Needleman et al. 1982, 1985; Bellinger and Needleman 1983) and critically evaluated by the EPA (1986). In comparison with children having low dentine lead levels (<10 ppm), children having high dentine lead levels (>20 ppm) had significantly lower Full-Scale WISC-R scores (Wechsler Intelligence Scale of Children--Revised), with IQ deficits of ~4 points, and significantly poorer performance on tests of auditory and verbal processing, on a test of attentional performance as measured by reaction time under conditions of varying delay, and on a teachers' behavioral rating (Needleman et al. 1979). The distribution of verbal IQ scores was shifted downward in the high-lead group, such that none of the children in the high-lead group had verbal IQs >125, whereas 5% of the children in the low-lead group had verbal IQs >125. Furthermore, children in the high-lead group were three times more likely to have a verbal IQ <80 than were children in the low-lead group (Needleman et al. 1982). Using regression analysis, Bellinger and Needleman (1983) found that IQs of children in the high-lead group fell below those expected based on their mothers' IQs and that the amount by which a child's IQ fell below the expected IQ increased with increasing dentine lead levels in what appeared to be a nonlinear manner. That is, dentine lead level was not significantly correlated with IQ residuals in the low-lead children or in the high-lead children with 20 to 29.9 ppm dentine lead, but was significantly correlated with IQ residuals in high-lead children with 30 to 39.9 ppm dentine lead.

Schroeder et al. (1985) and Schroeder and Hawk (1987) evaluated 104 children of lower SES (socioeconomic status) on the Bayley Mental Development Index (MDI) or Stanford-Binet IQ Scale at ages 10 months to 6.5 years. Hierarchical backward stepwise regression analyses indicated that blood lead levels were a significant source of the variance in IQ and MDI scores after controlling for SES and other factors. Fifty of the children were examined again 5 years later, at which time blood lead levels were ≤ 30 μg/dL. The 5-year follow-up IQ scores were inversely correlated with contemporary and initial blood lead levels, but the effects of lead were not significant after covariates, especially SES, were included in the analysis.

Hawk et al. (1986) and Schroeder and Hawk (1987) replicated this study with 75 black children, 3 to 7 years old, of uniformly low SES. Backward stepwise multivariate regression analysis revealed a highly significant linear relationship between Stanford-Binet IQ scores and contemporary blood lead levels over the entire range of 6 to 47 μg/dL. The association was nearly as striking when past maximum or mean blood lead levels were used. Because SES was uniformly low, it was not a significant covariate. This feature of the study may limit the applicability of the findings to the general U.S. population of children, if SES and lead exposure interact in such a way that IQ is affected by blood lead at lower SES levels but not at high SES levels.

Fulton et al. (1987) studied 501 children, 6 to 9 years old, in Edinburgh, Scotland where a major source of lead exposure is drinking water. The children were selected from a larger sample of 855 (mean blood lead level = 10.4 μg/dL) by taking all subjects in the top quartile of the blood lead distribution from each of the 18 participating schools plus a random ~1 in 3 subsample of the remaining children. Mean blood lead level of the study population was 11.5 μg/dL, with a range of 3.3 to 34 μg/dL. Blood lead levels >25 μg/dL were found in ten children. Multiple regression analyses revealed a significant negative correlation between log blood lead and the British Ability Scales Combined Score (BACS) and attainment test scores for number skills and work reading after adjustment for confounding variables. Further analysis divided the children into ten groups of ~50 based on blood lead level and plotted the group mean lead values against the group mean difference from the school mean score, adjusted for covariates. The authors reported that this analysis revealed a dose-effect relationship extending from the mean blood lead level of the highest lead groups, 22.1 μg/dL, down through the mean blood lead level of the lowest-lead group, 5.6 μg/dL, without an obvious threshold.

Bellinger et al. (1987) reported significant deficits of 4.8 points on the Bayley Mental Development Index (MDI) at ages 6 to 24 months in children whose blood lead level at birth was 10 to 25 μg/dL, in comparison with children whose blood lead level at birth was <3 μg/dL. These findings are supported by the data of Dietrich et al. (1987), who reported inverse correlations between prenatal or neonatal blood lead levels (range 1 to ~25 μg/dL) and MDI at 3 or 6 months of age. Additional evidence is provided by the data of Ernhart et al. (1985, 1986) and Wolf et al. (1985), which indicated that neonatal performance on a Neurological Soft Signs scale was related to umbilical cord blood lead levels, and to the indirect effect of cord blood lead on Neurological Soft Signs (EPA 1986; Davis and Svendsgaard 1987). These effects were significantly correlated with cord blood lead levels that averaged 5.8 μg/dL and that ranged up to only 14.7 μg/dL. Additional analysis follow-up through 3 years showed a negative association between maternal, but not cord, blood lead with several developmental indices at 6 months (Ernhart et al. 1987). Winneke et al. (1985a, 1985b) found that errors in the Wiener Reaction Performance test were associated with maternal blood lead levels averaging 9.3 μg/dL and cord blood levels averaging 8.2 μg/dL; most of the blood lead levels were ≤15 μg/dL.

3. SUMMARY OF SELECTED RECENT STUDIES

Winneke et al. (1990) combined results from 8 different groups from 8 countries combined on the basis of a common study protocol with inherent quality assurance elements. In order to attempt to improve the dose-response information on neurobehavorial effects in children, the World Health Organization, Regional Office for Europe, in collaboration with the Commission of the European Communities, initiated this international study which was planned, executed and evaluated between 1984 and 1989. This collaboration made it possible to increase the sample size (n=1,879) and broadened the range of the exposure dimensions (below 5 to about 60 μg

Pb/dL blood). Blood-lead concentrations were used as markers of environmental lead exposure. The battery of tests used included the psychometric intelligence [WISC- Wechsler Intelligence Scale for Children], visual-motor intelligence [Bender Gestalt test and the Trail-Making test (TMT)], reaction performance [Delayed Reaction Time (DRT) and Vienna Reaction Device (VRD)], as well as general behavior as rated by parents and/or teachers by means of simple rating scales. Apart from blood-lead as the independent and psychological outcome as the dependent variable, the full model for analysis also included age and gender as exogenous variables and father's occupational status as well as maternal education as social confounders. Within the range of below 5 to about 60 μg Pb/dL blood, the study confirmed that there are detectable exposure-related neurobehavioral effects in school-aged children between age 6 and 11 years.

The strongest and most consistent effects were observed in visual-motor integration (GFT and VRC). WISC scores were also affected by lead exposure, although the effects were inconsistent across studies and the overall degree of association with blood lead was of only borderline significance. However, in 8 studies including 1,698 children, the WISC scores were consistently and negatively associated with blood lead, with an approximate 0.12 decrease in the mean WISC score per μg/dL. No significant associations with blood lead were found for the less standardized measures, namely, Trail-Making test and the general behavior test. The results of this European study are relevant to judging risk due to environmental Pb exposure, because the original data in the individual studies were collected using a common study protocol with inherent quality control elements. Further, the findings were extracted from a large sample size which was subjected to a uniform statistical analysis.

In contrast to the original data taken by Winneke et al. (1990), re-analysis study done by Needleman et al. (1990) identified 24 modern studies of childhood exposures to lead in relation to IQ. From this population, 12 studies that employed multiple regression analysis with IQ as the dependent variable and lead as the main effect and that controlled for non lead covariates were selected for a quantitative, integrated review or meta-analysis. The studies were grouped according to type of tissue analyzed for lead. There were 7 blood and 5 tooth lead studies. Within each group, Needleman et al. (1990) obtained joint P values by two different methods and average effect sizes as measured by the partial correlation coefficients. They also investigated the sensitivity of the results to any single study. The sample sizes ranged from 75 to 724. The sign of the regression coefficient for lead was negative in 11 to 12 studies. The negative partial r's for lead ranged from -0.27 to 0.003. The power to find an effect was limited, below 0.6 in 7 to 12 studies. The joint P values for the blood lead studies were <0.0001 for both methods of analysis (95% confidence interval for group partial *r*, -0.15 $\pm$ 0.05), while for the tooth lead studies they were 0.0005 and 0.004, respectively (95% confidence interval for group partial *r*, -0.08 $\pm$ 0.05). The hypothesis that lead impairs children's IQ at low dose is strongly supported by this quantitative review. The effect is robust to the impact of any single study. While the group partial r of 0.15 $\pm$ 0.05 is suggestive of a small effect whose confidence limits do not include zero, the authors did not evaluate a composite effect size. Thus, nothing can be said about level of effect on IQ from this analysis.

The primary objective of the following three studies was to determine if the early deficits of neurobehavioral development seen in children with low-lead levels persist into later life, and if so, how adverse and stable are these effects. Bellinger et al. (1990) studied the cognitive performance in children exposed to low levels of Pb *in utero* and followed their performance through 5 years of age. Umbilical cord blood lead levels at birth, capillary blood lead levels at 6, 12, 18, and 24 months old and venous blood lead levels at 57 months were determined. Cord blood and postnatal blood lead levels of below 3 μg/dL were classified as low, above 10 μg/dL were classified as high, and between 3 and 10 μg/dL were classified as medium. Mental Development Index (MDI) score from the Bayley Scales of Infant Development at 24 months and General Cognitive Index (GCI) score from the McCarthy Scales of Children's abilities at 57 months were used to measure the cognitive performance in children. In addition, information on a range of potential covariates of Pb exposure and cognitive function were obtained. These included demographics, reproductive history, exposures during and characteristics of the index pregnancy, labor and delivery, neonatal characteristics, and measures of child rearing environment and practices.

The results showed that up to 2 years of age, children with umbilical cord blood lead levels of 10 to 25 μg/dL achieved significantly lower scores on MDI than do children with lower prenatal exposure. Further, in the second year of life, children from lower social classes expressed deficit at lower levels of prenatal lead exposure than did children from the highest social class. This association between high prenatal exposure and lower performance persisted beyond age 2, although not as a main effect. Rather, the degree to which deficit persisted varied among subgroups of children with different sociodemographic characteristics and postnatal lead exposure profiles. By age 5 years, however, they appear to have recovered from, or at least compensated for, this early insult. Further, this recovery or compensation between ages 24 and 57 months of age for children with high prenatal lead exposure, was more positive for children with lower postnatal blood lead levels at 57 months of age and coming from higher socioeconomic status, higher Home Observation for Measurement of the Environment (HOME) scores, higher maternal IQ, and female gender. Hence, the risk that a deficit will persist through the preschool years is increased among children with high prenatal exposure and either high postnatal exposure or less optimal sociodemographic characteristics. Children already stressed by sociodemographic disadvantages are hence less able to weather the additional stress of high prenatal lead exposure.

Dietrich et al. (1990) assessed the neurobehavioral effects of fetal and postnatal lead exposure during the first two years of life. Blood was collected for Pb analysis prenatally from the mother and at quarterly intervals from the infant. Infants were given developmental assessment at 3, 6, 12, and 24 months of age by using the three part assessment of the Bayley Scale of Infant Development, namely, Mental Development Index (MDI), Psychomotor Development Index (PDI), and Infant Behavior Record (IBR). They found that the prenatal and neonatal blood lead levels were low, with only a handful reaching or exceeding 25 μg/dL. Most children reached their highest blood lead levels during the second year. They further found that 25% of study subjects had at least one serial blood lead determination of 25 μg/dL or greater during the second year. Ten potential covariates and confounders were included for making statistical adjustments in regression models. They included birth weight, gestational age, Obstetrical Complications scale, Postnatal Complications Scale, child sex, child race, composite index of tobacco and alcohol consumption, maternal age, socioeconomic status, and poverty. Multiple regression and structural equation analyses revealed statistically significant relationships between prenatal and neonatal blood lead level and 3- and 6-month Bayley Mental and/or Psychomotor Development Index. However by 2 years of age, no statistically significant effects of prenatal or postnatal lead exposure on neurobehavioral development could be detected. The authors, using Bayley MDI raw scores, show that a postnatal neurobehavioral growth catch-up can occur in infants exposed fatally to higher levels of lead. These results obtained by Dietrich et al. (1990) differ substantially from those obtained by Bellinger et al. (1990), who reported a continuous inverse relationship between cord blood lead level and Bayley MDI between 6 months and 2 years. However, the authors' finding of no statistical significance between indices of postnatal lead exposure and Bayley MDI or IBR factors was in accord with that of Bellinger et al. (1990).

Dietrich et al. (1991) performed a follow up study on the group of children studied in the earlier paper of 1990. Since in the earlier paper no significant associations were found between prenatal and early postnatal Pb exposure and indices of early sensorimotor development in late infancy, this follow up study was conducted on children at age 4 years to determine if this disappearance represented true recovery or recovery which was limited to infant sensorimotor functions. The Kaufman Assessment Battery for Children

(K-ABC) was administered to children at approximately age 4. This test is a neurobehavioral survey of general intelligence, information processing, and achievement for children between the ages of 2.5 to 12.5 years. The test rating was done on a standardized 5 point scale with a score of one indicating little or no validity, and a score of 5 indicating a completely untroubled protocol. Higher neonatal lead blood levels were associated with poorer performance on all K-ABC subscales. However, this inverse relationship was limited to children from the poorest families. Maternal blood-lead levels were unrelated to 4-year cognitive status. Few statistically significant associations between postnatal PbB levels and K-ABC scales could be found. However, a weak relationship was found between the postnatal PbB levels and performance on a K-ABC subscale that assesses visual-spatial and visual-motor integration skills. In many ways the overall findings of this study do not agree with those of Bellinger et al. (1990). Further, the main procedural difference is the use of the K-ABC test in this study versus the use of the McCarthy Scales of Children's Abilities as used by Bellinger and coworkers.

Since there still exists a great deal of controversy regarding the lead effects on cognitive function in children, Bhattacharya et al. (1990) suggest the use of lead induced neuromotor functions for assessment. This may be equally important as measures of cognitive function since portions of the higher center of the central nervous system which contribute toward cognitive development, also play a significant role in neuromotor development in children. They studied the postural balance of children and how it related to blood lead levels. Although the results are preliminary in nature, with a very small number of children used in the study, the data suggest a significant relationship between sway area response and peak or max PbB during the second year of life.

4. CHOICE OF ENDPOINT AND RESPONSE FUNCTION

Effects on cognitive capacity related to lead exposure have been investigated for several decades. Decrements in performance are one manifestation of neurological disfunction, which would be a more general category of effects. However, it is necessary to identify a symptom which can be used in our willingness-to-pay (WTP) paradigm (refer to Part IV of this report); thus we have chosen a specific effect which will be generally referred to as IQ decrement. Because of the differences in measurement tests and grading systems, intelligence tests measure different things and they also have different numerical ranges. We have reviewed the literature up to the current time and find that there is a solidification within the community that blood lead is negatively related to neurotoxicological effects, including cognitive capacity. The precise relationship for any of these effects is in question, however, most modern investigators, using large sample sizes or who incorporate meta-analysis techniques, have been unable to identify a threshold below which no effect could be observed. There is a building consensus, however, that a child's cognitive capacity, if significantly affected at birth, can improve over time, if the child experiences more favorable economic or exposure conditions during the early years after birth.

Over the years, several re-analyses/meta-analyses have been performed. The most recent, and the most comprehensive was done by Needleman et al. (1990). These authors calculated group partial correlation coefficients for studies using blood lead and for studies using tooth lead as indicators of exposure. Each of the correlation coefficients was small, but the 95% confidence limits were beyond zero, the no-effect point. Taken together, the analysis of the 12 studies provide stronger evidence of a negative effect than any study alone. When viewed in the context of the Winneke et al. (1990) data, which provide a borderline significant, but consistently negative association with blood lead, the overall picture does not change significantly. Clearly the effect for small levels of lead exposure will be small, even in the sense of a population effect, in spite of the data suggesting that there may be no threshold. Therefore, we choose to build our analysis on the seven blood lead studies reviewed and accepted by Needleman et al. (1990) and the results of Winneke et al. (1990). The median value of the effect size of these studies is identified in the following text box.

... IQ Decrement = 0.25 x Pb (in μgPb/dL blood) ...

Dose-Response Function
for LEAD

IQ Decrement: Based on median effect size from studies reviewed by Needleman et al. (1990) and Winneke et al. (1990)

$$\text{IQ Decrement} = 0.25 \times \text{Pb (in } \mu\text{gPb/dL blood)}$$

Given that no author within the past decade has suggested an effects threshold, this equation is used without consideration of a threshold. Some notion of the uncertainty can be gained by reviewing the uncertainty in the group partial correlation coefficient derived by Needleman et al. (1990). The

95% confidence limits were approximately 33% of the partial correlation coefficient. We take this same proportion as a guide for estimating the confidence limits for the effect size. Thus we estimate the 95% confidence limits to be 0.25 $\pm$0.1.

In order to use the information which we have from effluent modelling, it is necessary to be able to convert the in-air concentration of lead to in-blood concentration. The USEPA (1986) identifies the median blood lead/inhalation slope for children as 1.92 μg/dL blood per $\mu g/m^3$ air. The contribution from ingested dust contaminated with lead deposited from the air is 6.26 μg/dL per $\mu g/m^3$ air. This value may be high because the supporting data were obtained during a period of lead use in automobile fuels. However, we choose to maintain this value until definitive data suggest changing, possibly lowering, this dust pathway uptake factor. Thus, for children, the aggregated lead value of blood lead per concentration in air is 8.2 μg/dL per $\mu g/m^3$ of air.

5. ASSUMPTIONS USED IN ANALYSIS OF DATA

The concentration-response function chosen was a simple, linear function previously described. In order to use it, the following assumptions are required:

1. Exposures occur to the mother/fetus/infant, and 1-year's exposure is sufficient to elicit the response.

2. The infant is assumed not to improve in exposure conditions, nor is a change in socioeconomic status assumed.

3. The number of new cases is proportional to the number of births per year.

4. The IQ decrement, as calculated, is a statistically-based notion similar to the person-rem concept in radiation protection. Thus, the IQ decrement is statistically distributed over the population of exposed infants, however, the particulars of the distribution are unknown.

IQ Decrement NUSAP[2]

Numerical Information: 0.25 x blood lead concentration

Units

Measurement Units: person IQ/μg lead per dL blood

Statistical Unit: Median

Spread

Confidence Level: 95%

95% confidence interval, as suggested by Needleman (1990), based on proportion (33%) of partial correlation coefficient derived in a meta analysis.

Upper Bound: + 0.1
Lower Bound: - 0.1

Assessment

Informative Value Based on Spread: MEDIUM

Effect sizes vary from about -8 to +2. However, we reject the +2 because we do not accept the notion of lead being beneficial to human health.

Informative Value Based on Application: MEDIUM

Generalizability to Other Applications: HIGH

Robustness of Value over Time: MEDIUM

Relationship considered perishable because of the large number of on-going studies which may clarify the changes in IQ decrement vs. lead exposure as the child matures.

Pedigree (credibility of the entry's origin)

Theoretical Basis: POOR

Epidemiology results continuing to be controversial at present environmental concentrations.

Data Inputs: FAIR

Estimation Methods: FAIR

Estimation Metric: GOOD

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[2]Refer to Part VI for a description of the NUSAP method for describing the quality of data and information.

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PAPER 9[1]

OZONE HEALTH EFFECTS

1. INTRODUCTION

Ozone is a principal component of photochemical air pollution endogenous to numerous metropolitan areas. It is primarily formed by the oxidation of NO_X in the presence of sunlight and reactive organic compounds. Ozone is a highly active oxidizing agent capable of causing injury to the lung (Mustafa and Tierney 1978). Lung injury may take the form of irritant effects on the respiratory tract that impair pulmonary function and result in subjective symptoms of respiratory discomfort. These symptoms include, but are not limited to, cough and shortness of breath, and they can limit exercise performance. Some of the toxicological properties of ozone were first investigated over 100 years ago by Schonbein (1851), who found it to be a lethal gas when inhaled by mice in sufficient quantities. When Schonbein accidentally inhaled ozone gas, it produced cough and pain in the chest; this is still a common finding. The effects of ozone observed in humans have been primarily limited to alterations in respiratory function, and a range of respiratory physiological parameters have been measured as a function of ozone exposure in adults and children. These affects have been observed under widely varying (clinical experimental and environmental settings) conditions (Adams 1987).

The vast data base on the effects of ozone on humans and animals provides abundant evidence of the adverse acute effects of ozone. Laboratory-based human and animal studies have suggested effects on pulmonary host defenses and the immune system, for example, decreased particle clearance rate in rats. Pinkerton et al. (1989) and Amoruso et al. (1989) observed a modification of alveolar macrophages with respect to a decreased production of superoxide radicals in mice and rats. Orlando et al. (1988) observed possible adverse effects on the immune system with respect to a decrease in the T-cell mitogen-induced blastogenesis of circulating T-cells in humans. Burleson (1988) found a decrease in pulmonary natural killer cell activity in rats. Short-term effects on lung structure have also been observed in animal models such as shortened or less dense cilia in the trachea and bronchi of monkeys (Castleman et al. 1977); damaged cilia in rats (Schwartz et al.

[1]Based largely on a working paper by C. Easterly.

1976), and elevated collagen synthesis rates and histologically discernible fibrosis in rat lung tissue (Last et al. 1979).

In addition to acute effects, a wide range of subchronic and chronic effects have been identified in laboratory-based animal studies. Because chronic exposures are some cumulative function of a series of acute exposures, a linkage exists between acute and chronic exposures, but the mechanisms, at present, are not fully defined. Some of the effects seen in animals subjected to subchronic and chronic exposures to ozone are: an increase in respiratory bronchioles in monkeys and a thickening of their walls (Tyler et al. 1988; Moffatt et al. 1987); a thickening of the alveolar septa (Moffatt et al. 1987); increased numbers of inflammatory cells and fibroblasts in the interstitia of monkeys' (Tyler et al. 1988, Moffatt et al. 1987) and rats' (Pickrell et al. 1987) lungs; increases in the number of epithelial cells (Boorman et al. 1980); altered morphology and cell type shifts in rats (Huang et al. 1988; Grose et al. 1989); and an increase in amorphous extracellular matrix in rats (Huang et al. 1988; Grose et al. 1989). Finally, Zelikoff et al. (1991) have demonstrated that rabbits exposed to ozone exhibit compromised macrophage (MO) responses in lung tissue. Because the MO provides important functions in tumor surveillance, susceptibility to pulmonary cancer may be effected by ozone exposure.

The results of studies in animals and the range of chronic effects observed suggests that there is significant potential for chronic effects in humans. In addition, the types of morphological changes caused by ozone in animals are also observed in the lungs of cigarette smokers. These changes are generally interpreted as representing early stages of chronic lung disease in smokers. Several epidemiological studies tend to support a concern about the potential for chronic effects in humans (Detels et al. 1987; Knudson et al. 1983; and Kilburn et al. 1985). While there are acknowledged imperfections in these studies, they suggest an increased rate of lung function decline with ozone exposure that has also been observed in animal studies. Notwithstanding, there is no definitive evidence from epidemiological studies that ambient ozone exposures cause chronic effects in humans. Consequently, there will be considerable uncertainty in any estimates of externalities associated with the chronic effects of ozone on health.

NAPAP Report 22 (EPA 1990) identified a set of observations that are considered to provide clear and consistent evidence from human clinical, epidemiological and field studies regarding acute effects of ozone on human pulmonary function:

1. Inhalation of ozone causes concentration-dependent mean decrements in lung volumes and flow rates during forced expiratory maneuvers.

2. The mean decrements increase with increases in minute ventilation as a function of increasing exercise.

3. A wide range of reproducible individual responsiveness exists among healthy young adults, with the upper end of the distribution constituting the most responsive group of individuals.

4. There are small differences in average ozone responsiveness (based on lung function test changes) among different population groups. Healthy young adults and children appear to be slightly more responsive than older adults. Asthmatics have greater increases in airway resistance and smokers have smaller changes in spirometry after ozone exposure than healthy subjects.

5. Repetitive, daily exposures at levels initially producing a functional response may lead to an enhanced response on the second day, but responsiveness diminishes on subsequent consecutive days of exposure. This "attenuation" phenomenon is transient, disappearing about 1 week after termination of continued daily exposure. Responsiveness may have a seasonal pattern of variation; that is, there may be less responsiveness in the fall after repeated summertime high-ozone exposures. This seasonal variation may complicate or compromise the ability of epidemiological studies to detect effects.

6. Multihour or multiday exposures more likely lead to pulmonary response than 1-hour peak exposures. A pattern of simultaneous or sequential exposure to other pollutants such as acidic aerosols is, likewise, more likely to lead to pulmonary response.

7. In adults, ozone exposure causes respiratory symptoms such as cough and chest discomfort.

2. CHOICE OF MODEL

Risk estimates for a number of urban areas have been performed using existing or projected levels of ozone (e.g., Hayes et al. 1987; Whitfield 1988; Krupnick and Kopp 1988; Hayes et al. 1989; and Hayes et al. 1990). These estimates were developed for both pulmonary function and lower respiratory tract symptoms. In the willingness-to-pay paradigm that is used in our study, the most fully investigated health endpoint, pulmonary function, cannot be used because pulmonary decrements themselves have not been evaluated. Pulmonary decrements have not been linked to specific symptoms of ill health by the medical community and without a symptom, it is difficult to obtain a measure of the willingness to pay to avoid the pulmonary decrement. Because the focus of the present study is on willingness to pay, and because Krupnick and Kopp (1988) have evaluated eight symptomatic endpoints for use in that particular economic evaluation, we adopt their study for our use. Krupnick and Kopp (1988) draw on some of the same health studies as Hayes et al. (1987, 1989, 1990).

Krupnick and Kopp (1988) identified eight symptoms from five studies and reduced the quantitative information to mathematical models, usually probit functions (Schwartz, Hasselblad, and Pitcher 1989; McDonnell et al. 1983; Krupnick et al. 1987; Portney and Mullahy 1986; Holguin et al. 1985). One study (McDonnell et al. 1983), was reanalyzed by Krupnick and Kopp (1988) from the raw data in order to make the results more useful for an economic analysis. Krupnick and Kopp (1988) made several assumptions in order to fit the epidemiologic or clinical studies to the needs of an economic analysis. For the symptom endpoints, it was necessary to convert symptom incidence to symptom-day. Because the symptoms are associated with elevated breathing rates, it was necessary to identify the percentage of the time an average person spends engaged in heavy exercise, adjusted for indoor and outdoor activities. Baselines were also identified for several symptoms. Because Krupnick and Kopp (1988) investigated issues related to regulatory statutes, their assessment was designed around two key types of ozone measurements. These key measures are the maximum 1-hour ozone concentrations, and the average for a 2-week period of daily 1-hour maximum ozone concentrations. These two key measurements fit well into the present study because measurements can be readily identified for the baseline (present) conditions at the two sites. Further, atmospheric transport models can be used to estimate the increased concentrations of ozone due to emissions from coal-fired power plants.

Krupnick and Kopp (1988) identified eight symptoms from five studies and reduced the quantitative information to mathematical models, usually probit functions ...

These key measures are the maximum 1-hour ozone concentrations, and the average for a 2-week period of daily 1-hour maximum ozone concentrations.

The particular symptoms chosen for our analysis, based on the earlier development of Krupnick and Kopp (1988), are:

Epidemiologically-Based Endpoints

1. Total Respiratory Restricted Activity Days (TRRAD), used by Portney and Mullahy (1986). This measure is based on symptoms identified by adults over a 2-week recall period. Their health effects model was based on an average for a 2-week period of daily 1-hour maximum concentrations of ozone, as recorded within a 20-mile radius.

2. Any-Symptom Day (Krupnick, Harrington, and Ostro 1987). This study resulted in a variety of response functions for a variable that took the value of one if any of 19 symptoms or conditions were present on a given day and zero otherwise. Except for eye irritation and headache, these symptoms and conditions were all respiratory related. The response function is based on adults and daily 1-hour maximum ozone concentrations. In the accounting framework, the total number of Any-Symptom Days is reduced to the extent that there is no double counting of separately calculated symptoms.

3. Asthma-Attack Day (Holguin et al. 1985). Based on a 12-hour period of observations on identified asthmatics, and related to total oxidants, this study was modeled by Krupnick and Kopp (1988). They used two time periods in order to do statistical fits with daily 1-hour maximum ozone concentrations.

4. Eye-Irritation Day (Schwartz, Hasselblad, and Pitcher 1989).

5. Days of Coughing (Schwartz, Hasselblad and Pitcher 1989). This study investigated the relationship between total oxidants, coughing, eye irritation and chest tightness. Only the first two symptoms were found to be significantly associated with oxidant exposure to members of the total population.

Clinical Study-Based

6. Cough Incidence (McDonnell et al. 1983).

7. Shortness of Breath (McDonnell et al. 1983).

8. Pain upon Deep Inspiration (McDonnell et al. 1983). This study found the difference in symptom scores taken before and after 2-hour

ozone exposures in a clinical setting. Krupnick and Kopp (1988) obtained the raw data from this study and performed a re-analysis, and then developed a procedure for adapting results from 2-hour incidences to a symptom-day measure.

Several steps are required to apply the Krupnick and Kopp (1988) results to estimate the effects of ozone on health at a reference site:

1. The concentration-response functions from Krupnick and Kopp (1988) were coded into a simple Fortran program using the middle value coefficients plus the upper and lower 75% confidence limits.

2. For the months of May, June, July, August, and September, during which ozone production is significant at the Southeast site, daily 1-hour maxima were transcribed from the EPA's Aerometric Information Retrieval System (AIRS) data base, modified by a factor of 0.773 as described in Paper 3 of this document, and used as data input to which incremental ozone values were added. These increases in ozone concentrations were obtained from the modeling described in Paper 3 of this document using an estimate of median ozone conditions. The baseline and its increment were used as input to the health effects algorithms.

3. In the execution of the computer code, the AIRS data (the baseline) when combined with the additional amount attributed to the reference plant were checked for values below 0.08 ppm. This level was considered to be a practical threshold for the present study.

4. The populations used for this evaluation were divided into local (within 50 mile radius of the power plant) and regional (beyond 50 miles) populations.

Results of the computations for the 5-month ozone season were summed over the appropriate populations. This calculation provided an estimate of the numbers of cases.

The following dose-response functions provide relevant calculational information:

Dose Response Functions:
OZONE

Incidences of pain upon deep inspiration (PDI): Based on McDonnell et al. (1989)

$$\Delta C = \{[1/(1+\exp(-\gamma-\beta\omega X_1))] - [1/(1+\exp(-\gamma-\beta\omega X_0))]\} f\theta(\text{mpop})$$

where

ΔC = change in number of PDI incidences for 2-hour period t

X_0 = daily maximum hourly ozone concentration, baseline in reference environment

X_1 = daily maximum hourly ozone concentration including reference plant

γ = -0.799

β = 3.456, 6.946, 10.436

mpop = entire population

θ = percent of a 2-hour period the population is exercising

f = the incidence-day factor

ω = the scaling factor for 2-hour period t

Dose Response Functions (continued)
OZONE

Any symptom or condition (ARD): Based on Krupnick, Harrington and Ostro (1987)

$$\Delta ARD = \beta^* (X_1 - X_0) (apop)$$

where

ΔARD	=	change in the number of days of "any" symptoms/conditions
β^*	=	marginal change in the stationary probability of experiencing any symptom/condition
	=	$p_0(1-p_1)\beta[p_1+(1-p_0)]/(1-p_1+p_0)^2$, where p_0 is the conditional probability of illness on day t given wellness on day t-1, p_1 is the conditional probability of illness on day t given illness on day t-1, and β is the ozone coefficient from the logit model regression.
	=	0.13, 0.20, 0.27
X_0	=	daily maximum ozone concentration, baseline in reference environment
X_1	=	daily maximum ozone concentration including reference plant
apop	=	adult population

Dose Response Functions (continued)
OZONE

Total respiratory-related restricted activity days (TRRADs): Based on Portney and Mullahy (1986),

$$\Delta TRRAD = TRRAD\ [\exp\ [\beta(X_1 - X_0)]\ -1]\ (apop)$$

where

ΔTRRAD	=	change in number of respiratory-related restricted activity days for the 2-week period
TRRAD	=	baseline per capita TRRADs for a 2-week period
X_0	=	average daily 1-hour maximums of ozone concentrations for each 2-week period, baseline in reference environment
X_1	=	average daily 1-hour maximums of ozone concentrations for each 2-week period including reference plant
apop	=	adult population
β	=	2.63, 7.99, 13.34

Dose Response Functions (continued)
OZONE

<u>Asthma attacks:</u> Based on Holguin et al. (1985)

$$\Delta a = [m/(1+m) - p] (apop)$$

where

$$m = [p/(1-p)] \exp (\beta\omega X_1 - \beta\omega X_0)$$

and

Δa = change in number of asthma attacks for the 7AM-7PM or 7PM-7AM period

p = baseline number of attacks per asthmatic for the day

X_0 = maximum 1-hour ozone concentration for 7AM-7PM, baseline in reference environment

X_1 = maximum 1-hour ozone concentration for 7AM-7PM including reference plant

apop = asthmatic population

ω = scaling factors for half-day periods

β = 3.58, 6.20, 8.82

Dose Response Functions (continued)
OZONE

Incidences of coughing: Based on McDonnell et al. (1983),

$$\Delta C = \{[1/(1+\exp(-\gamma-\beta\omega X_1))] - [1/(1+\exp(-\gamma-\beta\omega X_0))]\} f\theta(mpop)$$

where

ΔC = change in number of coughing incidences in two-hour period

X_0 = daily maximum hourly ozone concentration, baseline in reference environment

X_1 = daily maximum hourly ozone concentration including reference plant

γ = -1.742

β = 10.961, 14.1, 17.239

mpop = entire population

θ = percent of a two-hour period the population is exercising

f = the incidence-day factor

ω = the scaling factor for two-hour period t

Dose Response Functions (continued)
OZONE

<u>Incidences of shortness of breath</u>: Based on McDonnell et al. (1989)

$$\Delta C = \{[1/(1+\exp(-\gamma-\beta\omega X_1))] - [1/(1+\exp(-\gamma-\beta\omega X_0))]\} f\theta(mpop)$$

where

ΔC = change in number of shortness of breath incidences for two-hour period

X_0 = daily maximum hourly ozone concentration, baseline in reference environment

X_1 = daily maximum hourly ozone concentration including reference plant

γ = -0.076

β = 4.938, 7.265, 9.562

mpop = entire population

θ = percent of a two-hour period the population is exercising

f = the incidence-day factor

ω = the scaling factor for two-hour period t

Table 1. Health effects estimated to occur from ozone exposure (in thousands) for the maximum extent of the ozone plume (about 215 km)

Southeast Reference site	Lower	Mid	Upper
1. Total restricted activity days	4.2	13	24
2. Any-symptom day	11	27	43
3. Asthma-attack day	0.85	1.5	2.2
4. Eye-irritation day	30	36	43
5. Cough day	9.5	15	20
6. Cough	60	84	109
7. Shortness of breath	34	49	62
8. Pain upon deep inspiration	22	47	73

Table 2. Health effects estimated to occur from ozone exposure within 50 miles (80 km) of the plant (in thousands)

Southeast Reference site	Lower	Mid	Upper
1. Total restricted activity days	3.6	11	20
2. Any-symptom day	9.1	23	37
3. Asthma-attack day	0.73	1.3	1.9
4. Eye-irritation day	26	31	36
5. Cough day	8.1	12	17
6. Cough	51	72	93
7. Shortness of breath	29	42	53
8. Pain upon deep inspiration	19	40	62

NUSAP: OZONE HEALTH EFFECTS[2]

Numerical Information: Symptoms (Any-symptom day, Asthma-attack day, Cough-day, Cough incidence, Shortness of breath, Respiratory restricted activity days, Eye irritation-day, and Pain upon deep inspiration)

Units

Measurement Units: symptom days/incidences

Statistical Unit: Lower Bound, Mean, Upper Bound

Spread

Confidence Level: 75%

Upper Bound: x 2 to 4
Lower Bound: ÷ 2 to 4

Eight endpoints are examined in this work; typically the lower to the high estimates are within a factor of 2 to 4. A formal sensitivity analysis has not been done; however, a 1% increment in effect is affected only slightly by ambient concentration.

Assessment

Informative Value Based on Spread: LOW

Informative Value Based on Application: LOW

Damages vary one to one with impacts.

Generalizability to Other Applications: MEDIUM

Robustness of Value over Time: MEDIUM

Studies useful for this assessment are expensive and will be relatively rare, thus present findings will not be challenged often. Assumptions within the secondary analyses necessary to utilize data in the economic analysis may be the subject of second guessing; however, most changes should only marginally affect overall results.

Pedigree (credibility of the entry's origin)

Theoretical Basis: FAIR

No theory needed. Data tell the story. However, good laboratory data support the findings of adverse effects.

[2] Refer to Part VI for a description of NUSAP.

Data Inputs: GOOD

Some data come directly from environmental epidemiology, other come from clinical work. In both cases, adaption was necessary to fit analysis needs. These adaptions probably represent the weakest link in that any assumption which moves outside the direct realm of the original data is an extrapolation.

Estimation Methods: GOOD

Estimation methods are basically fitting data to a curve and then applying conditions to meet analysis needs.

Estimation Metric: EXCELLENT

Symptoms are what is valued and this is what is measured.

Note: Health effects analyses are dependent on atmospheric modeling of ozone. Significant uncertainty exists in this source of information.

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PART IV

ECONOMIC VALUATION

PAPER NO. 10*

BENEFITS FROM REDUCING RISK OF DEATH

1. INTRODUCTION

Of the categories of benefits to individuals, reductions in the risk of premature mortality are of central concern to the public and environmental policy makers. These benefits can include those from reductions in "own- risk," for example, an individual's valuation of reducing his or her own mortality risks; reductions in risk to an individual's family, friends, or co-workers (i.e., of people known to the individual); and reductions in risks to unknown individuals. The last type would be an example of altruistic value.

The overall goal is to measure the welfare change from a change in the current and/or future probability of dying. The willingness to pay (WTP) reflects the amount of income taken from a person that would leave him or her indifferent to a decrease in risk, whenever it occurs. When this value is divided by the risk change, the resulting value is called the "value of a statistical life." Another relevant measure appearing in the literature is the value of life-years saved.

A distinction is made in the literature between the WTP for reductions in mortality risks and the WTP for reductions in morbidity incidents or morbidity risks. This distinction is useful if one can separate values for avoiding premature death from values for avoiding morbidity. If, for instance, WTP estimates for the former depend only on preferences for avoiding premature death, not on how death occurs (i.e., on whether it is accidental or through a disease, on the type of accident or disease, etc.), then WTP estimates for reductions in mortality risks can be added to those for reductions in morbidity. Take the case of valuing cancer risk reductions. Adding a WTP estimate for reductions in risk of accidental death to an estimate of reducing risks of the morbidity effects of cancer would yield an unbiased estimate of WTP for cancer risk reductions.

This separability approach can fail for two reasons. First, it may be that the cause or course of death affects WTP to avoid death or, more fundamentally,

*Based largely on a working paper by Alan Krupnick.

that individuals simply cannot unbundle risks of death from morbidity effects that can lead to death. Second, measurement problems can easily crop up. For instance, contingent valuation (CV) questions on reducing risks of mortality from cancer may not explicitly exclude the increased risks of morbidity that having cancer would imply.

A final definitional issue concerns the type of premature mortality risks one is valuing when environmental pollution is at issue. While most effort has gone into estimating the welfare effects of a change in current probability of death of healthy workers on the job, this is more relevant for characterizing the benefits of reducing accidental death risks than death from environmental causes. Exposure to pollutants raises risks of developing cancer, chronic heart, respiratory, and other diseases that raise mortality risks in the future. Such exposure also may raise current death risks for the very old and the sick. But, surely the pollution effect that is analogous to occupational health risks—pollution exposures high enough to raise current risks of death for the healthy, prime-age person—is insignificant in the United States.

2. THEORETICAL MODELS

Freeman and Cropper (1989) have a good discussion of the underlying model for estimating welfare changes from changes in the probability of death. In their model, each individual has a probability of dying (empirically, this can be found in life tables) at each age. A reduction in pollution exposure alters this survival probability distribution (P). This individual can also engage in averting behavior to reduce changes in this distribution. He experiences satisfaction (utility) each year of life and tries to maximize the present discounted value (PDV) of this utility, which is altered by changes in his mortality probability distribution. While it may be desirable to live longer, utility, not years, is what matters. The maximization of utility takes place subject to the constraint of lifetime earnings plus initial wealth.[1]

This model yields the solution that the WTP at any age (j) for a reduction in risk (r) at any age (k) equals the monetized loss in expected utility from (k) onward, plus the added earnings from living longer minus the reduction in consumption that can be afforded. This is all multiplied by r to get WTP for the risk change. This expression implies that the later the reduction in utility is expected, the lower is WTP. Depending on how consumption changes, WTP at age j may decrease with age or increase and then decrease. Also, the model

[1]Cropper and Sussman (1989) show that a bequest motive is not a particularly important aspect of this problem.

implies that more risk-averse individuals have a larger WTP for risk reductions, ceteris paribus.

This result shows the importance of age at exposure, latency, and future quality of life in determining current WTP. Concern about these factors has led to proposals and attempts to estimate WTP in terms of life- years lost and quality-adjusted life-years lost. However, most studies value average WTP for an average risk reduction and are not explicit about life-years saved or quality-adjusted life-years saved.

...most studies value average WTP for an average risk reduction and are not explicit about life-years saved or quality-adjusted life-years saved.

3. ESTIMATION APPROACHES

There are three approaches in use for estimating WTP: wage-risk, averting behavior, and contingent valuation methods (CVM). The first two are based on the aforementioned model with the survival probability distribution (P) endogenous and assume that individuals behave to equate marginal risks in all settings; the last assumes that P is exogenous and confronts individuals with scenarios that change P in specific, usually simple, ways.

Perhaps the most common approach has been to examine pay differentials across a sample of occupations posing different annual mortality risks. By holding constant other attributes of the occupations and the workers, it is possible to estimate how much extra compensation is required to induce people to accept slight increases in job risk. These wage premia are often used to measure individuals' WTP for reduced environmental risks.

For obvious reasons, the measures are far from perfect as measures applicable to environmental risks of death. The death being valued is instant and accidental not latent and lingering. The sample providing risk preferences is generally restricted to prime-age males not the older and young people that would be most at risk from exposure to pollutants. Occupational risks are generally thought of as being borne more voluntarily than the risks arising from air pollution and self-selection may place risk-lovers in the riskiest occupations,

There are three approaches in use for estimating WTP: wage-risk, averting behavior, and contingent valuation methods (CVM).

thereby making their preferences unrepresentative of those at risk from environmental pollution. Numerous studies of risk perception identify the voluntary/involuntary nature of risk as an important factor influencing the size of perceived risks. Another problem is that if workers lack information about their true risks the wage premia will reflect their perceived risks rather than the true risks. This is acceptable so long as the researchers incorporate perceived risks in their estimations. One study (Gegax et al. 1985) has done just this. It must be assumed that workers have perfect mobility; otherwise their wage will not reflect risks and other factors. For instance, if workers in Appalachian coal mines feel that they cannot relocate, then their wages may not fully reflect job risks. The way the data are collected makes it difficult to distinguish the wage premia effects of mortality risks from those of morbidity (including injury) risks. These studies value average life years lost for males averaging 40 years old. At a minimum, estimates of WTP that depend on age are needed, but most of these studies do not yield them. It must be assumed that all occupational risk is capitalized in labor markets. Some of these risks may conceivably be capitalized into property markets, leaving the former estimates too low.

Reduced mortality risks have also been translated into dollars using the contingent valuation method, for example, by asking individuals directly through survey means what they would be willing to pay for these risk reductions in hypothetical questions. The advantages of this approach are that scenarios can be designed to accord directly with the type of risk (involuntary and environmental, latency, etc.) and other characteristics of the problem; and the sample of people can be those at risk, or at least a more representative population.

Economists have resisted using this approach, however, because it relies on what people say, not what they do. A particular problem with respect to mortality valuation is the difficulty people have in understanding small risk changes.

Numerous studies of risk perception identify the voluntary/involuntary nature of risk as an important factor influencing the size of perceived risks.

The averting behavior method has also been used to value reductions in mortality risks. For instance, in some cases individuals living near hazardous waste sites take averting measures to reduce perceived carcinogenic and other risks to health. By observing the amounts they spend on such means—buying bottled water, for instance—it is possible to make tentative inferences about the minimum amounts they are willing to pay to reduce perceived risks. One problem with this approach is that such measures reduce perceived morbidity as well as mortality risks. Thus, it is not correct to attribute the entire averting cost to the mortality risk reduction.

It was common several years ago to value premature mortality using the so-called "foregone earnings" approach, in which deaths were valued according to the wage or salary income that would be lost as a result of premature death. This approach is no longer viewed as acceptable for modern benefit-cost analysis (BCA). Not only does it fail to assign any value to the prevention of mortality among the elderly and the retired (who have no current income), but it is also inconsistent with the basic fundamentals of economics from which BCA sprung. This is because it ignores individuals' WTP for reductions in their own — as well as others' — mortality risks. This WTP may bear little relationship to their income.

4. SURVEY OF THE LITERATURE

Table 1 summarizes 17 studies addressing the valuation of reductions in risk of premature mortality. Two of the studies are recent surveys of the literature; the others are more recent or particularly important or unique studies that deserve special treatment. These studies are classified according to whether they are surveys of the literature, value an average mortality risk reduction, value a life-year saved, are a derivative study, provide values by type of death, are based on jury awards, or are valuing other people's lives (altruism). Within each of these classifications, each study is described by eight characteristics: author/year, city/region sampled, estimation approach, type of death, sample size, baseline risk, scenario, valuation result. Most of the studies in the table are briefly reviewed in the following sections.

4.1 SURVEYS OF THE LITERATURE

Two recent surveys of the mortality valuation literature attempt to summarize the literature and arrive at reasonable estimates of the range of values. Fisher, Chestnut, and Violette (1989) review 21 studies, most wage-risk studies, concluding that the "most defensible range for the value of a statistical life (VSL) estimates is $1.6 to $8.5 million (1986 $s)" for risks in the 1 to 10 in 10,000 range. They put more faith in the lower end of this range. Miller (1990) reviews 47 "good quality" mortality risk valuation studies and adjusts their results to be more consistent with economic theory (using after-tax instead of before-tax wage rates, for instance), with the definition of risks they used, for the value of travel time (for the motor fatality studies), and for other problems.

He finds a mean VSL of $2.2 million (1988 dollars) with a reasonable range of 30%. He also finds no discernable trend in the VSLs as a function of the size of the risk change.

Fisher, Chestnut, and Violette (1989) review 21 studies, most wage-risk studies, concluding that the "most defensible range for the value of a statistical life (VSL) estimates is $1.6 to $8.5 million (1986 $s)" for risks in the 1 to 10 in 10,000 range.

These and other reviews find some empirical support for the theoretical prediction that people above prime age will evidence lower WTP. Jones-Lee, Hammerton, and Philips, for instance, find that WTP peaks at age 40, declining thereafter. In Mitchell and Carson (reviewed in the following paragraphs), being over 55 years old decreases WTP to reduce future risks of death. However, in studies that do not allow for age to affect WTP nonlinearly, age appears insignificant.

4.2 STUDIES OF AVERAGE MORTALITY RISK REDUCTIONS

Few studies have attempted to estimate WTP for reducing mortality risks in an environmental context. Mitchell and Carson (1986) examined WTP for various size reductions in cancer risks from reduced exposure to trihalomethanes (THM) in drinking water. In its questions, this study incorporates a latency period between exposure and possible realization of the cancer (assumed to be 20 years). It finds that for a 4 in 1 million change in latent cancer risks, the value of a statistical life is about $2.5 million (1986 dollars), but for risk reductions in the range of wage-risk studies (1 in 10,000), the VSL is only $167,000. The latency consideration should result in lower estimates of VSL relative to the literature on accidental death (where latency is zero). Yet, there is a supposition that VSL's for environmental types of death (non-accidental and involuntary) are likely to exceed those of worker or auto death. Thus, some doubt is cast on whether the Mitchell and Carson survey instrument was fully credible; in particular, whether respondents could understand the extremely small changes in health risks they were asked to value.

Table 1. Key features of valuation of mortality risk studies

Author/ Year	City/ Region	Estimation Approach	Type of Death	Sample Size	Baseline Risk	Scenario	Valuation (1989$)
SUMMARY OF THE LITERATURE							
Fisher, Chestnut and Violette (1989)[a]	NA	Review of 21 studies	Mostly accidental/on job	21 studies	10^{-4} to 10^{-3}	NA	$3.5 million best; $1.8-$9.6 million per life
Miller (1990)[b]	NA	Review; adjusts studies for after-tax wage, risk def., time value, etc.	Mostly accidental	47 studies	$5x10^{-7}$ to $5x10^{-4}$	NA	$2.3 million +/-30%
AVERAGE MORTALITY RISK REDUCTION							
Mitchell and Carson (1986)[c]	Herrin, IL	CVM for households paid in water bill, risk ladder of cigarette equivalents	Cancer from THM in drinking water, 20 yr. latency	237 households	Unspecif. variation	a) $-4x10^{-6}$ b) -10^{-4}	a) $2.8 mill. b) $0.19 mill.
Smith and Desvousges (1987)[d]	Acton, MA, Boston	CVM (in taxes and price) with many different levels of baseline risks and risk changes; sequence of larger risk reductions; attention to biases, focus groups	Unstated from hazardous waste exposure, 30 yr. latency	609 households	Varies widely	Two e.g.'s: a) 10^{-2} and $2x10^{-2}$ b) $6x10^{-4}$ and $7x10^{-5}$	a)Average bids are from $9 to $34 => $0.001-0.007 mill. VSL b) $2 mill.
Magat, Viscusi, Huber (1990)[e]	Greensboro, NC	Contingent ranking via computer game; trade-offs between mortality risk and COL in choice of residence	Auto accident	195 adults	Unspecif. variation	10^{-5}	$8.2 mill. (median = $2.3 mill.)
Gegax, Gerking, and Schulze (1987)	U.S.	Mail survey used in hedonic wage equation; rerun with actual risks by occupational group using BLS data	Accidental on job (perceived)	37 heads of household (out of 6,000 mailed)	4 to 10/10,000	NA	For unionized, blue collar and non-unionized workers, from $1.2-2.5 mill., 14-21% of VSL using actual risks

Table 1. Key features of valuation of mortality risk studies

Author/ Year	City/ Region	Estimation Approach	Type of Death	Sample Size	Baseline Risk	Scenario	Valuation (1989$)
Garen (1988)[f]	National wage database	Hedonic wage; selection bias in risky jobs tested	Accidental on job	2,863 adults (blue collar)	Varies with blue collar occupation	NA	$9.2 million compared to $4 mill. with selection bias
Atkinson and Halvorsen (1990)[g]	U.S.	Hedonic Box-Cox weighted least squares regression explaining annualized automobile costs with fatality risks by model, weighted by sales	Auto accident	112 1978 models	0.15 per 10,000	NA	$3.80 mill.
Leigh (1987)[h]	U.S.	Hedonic model with and without female workers	Accidental on job	541 F & M, 326 M only, 348 M only, & 230 F only	NA	NA	With CPS dataset: $5 mill. for men and women, $2.7 for men only
LIFE YEARS SAVED							
Harrison et al. (1990)[i]	NA	Divides literature VSLs by 40 years loss in occupational life expectancy and multiplies by 12.5 years (ave life exp. for those with environ. diseases)	Accidental from hedonic wage literature	NA	Same as hedonic wage studies	NA	VSL = 31% of "average" VSL
Viscusi and Moore (1988)[j]	National	Hedonic wage with risk weighted by discounted life-years remaining	Accidental on job	NA	$5x10^{-5}$	Average of $-5x10^{-5}$	Avg $0.19 mill. per expected life yr (@12% time pref.) => $2 mill. for 12.5 yrs lost
DERIVATIVE STUDIES							
Hall et al. (1990)	NA	Values from literature	Applied to air pollution	NA	Same as literature	NA	B=$3.7 mill.; LB=$1.7 mill., UB=$8.6 mill.
EPA (1990)							$3 million

Table 1. Key features of valuation of mortality risk studies

Author/ Year	City/ Region	Estimation Approach	Type of Death	Sample Size	Baseline Risk	Scenario	Valuation (1989$)
RER (1990)							B=$4 million, LB=$1.8 mill. UB=$9 mill.
VSLs BY TYPE OF DEATH							
Harrington (1984)[k]	National	Theoretical life-cycle model and life table, plus assumptions on parameters	For males a) cardio-vascular disease, b) cancer, c) auto accident	NA	Life tables relevant to each type of death; for 10-5 change	r=5%, B=0.2 million, max income=$24 K	CD/AA = 0.55, C/AA = 0.62
JURY AWARDS							
Bovbjerg et al. (1989)	Florida and Kansas City Metro area	Descriptive statistics of wrongful death awards	Wrongful death	95 cases	NA	NA	$1.3 mill. median $0.62 million
VALUING OTHER'S LIVES							
Needleman (1976)	Great Britain	Infers ratios of own risk valuation to close relative valuation from willingness to donate a kidney	From kidney disease	NA	NA	NA	Relatives valued at 45% of reducing own risk

Table 1. Key features of valuation of mortality risk studies

Author/ Year	City/ Region	Estimation Approach	Type of Death	Sample Size	Baseline Risk	Scenario	Valuation (1989$)

Notes:

a. Fisher et al.: Support for reduction in VSL above prime age, but not linear (see Harrison below).
b. Miller: VSL's invariant to size of risk change. Support for above age effect.
c. Mitchell: Question whether small risk changes were understood.
d. Smith: Risk circles made differences in risk changes impossible to distinguish. With numerator constant and denominator changing, VSL rises with smaller risk changes. Also finds that WTP for additional risk reductions rises (increasing returns). Deaths of younger people in auto accident give this a higher value.
e. Magat: Computer interactive procedure invalidated.
f. Garen: VSL increases dramatically when voluntary nature of job choice is corrected for. But less applicable if individuals can choose where they live.
g. Atkinson: Driver characteristics, such as age group, seat belt usage, sex, and alcohol use were insignificant. This raises questions about the entire study. Further, it is not clear that drivers have any perception about fatality risk differentials among models. Indeed for new models, there would be no data to draw upon.
h. Leigh: Including women has little affect on fatality coefficient but with lower baseline risk for women, VSL rises (but only using dataset with part-time workers included). Check use of Texas data.
i. Harrison: Average worker is 37 years old, dying at 77. Diseases are cardiovascular, cancer, and pulmonary. This approach is ad hoc, making the strong assumption that WTP varies linearly with life-years.
j. Viscusi and Moore: Better approach than Harrison but applicable only to prime age males exposed to immediate mortality risks.
k. Harrington: Ratios of values of life given because incomes are out-of-date. Different r would change results.

B = best estimate
LB = lower bound
UB = upper bound

Another study addressing environmental risks is Smith and Devousges (1987). They asked 609 persons in the Boston area and in Acton, Massachusetts (which has experienced incidents of well contamination), their WTP (in tax and price increases) to reduce household health risks from a hypothetical hazardous waste landfill 3 miles from their home. The complicated sampling frame involved many different levels of baseline risk and risk changes, and each individual was asked to value progressive reductions in risk. The study is strong because of its extensive focus group sessions and attention to possible CV biases and other issues.

The results of the study are disquieting in that the average bids are virtually identical irrespective of baseline risks and the size of risk changes. Thus, given that the average bid for obtaining any risk reduction is about the same (from $8 to $31), for smaller risk changes, the VSL is generally larger, ranging from a low in the $1,000–6,000 range (for risk changes of 1/100 to 1/200) to a high of $2 million (for risk changes of 1/60,000 and 2/300,000). Unfortunately, this result may be an artifact of the "risk circle" method used to communicate the size of risk changes in that alternative degrees of risk changes were virtually indistinguishable visually on the circles. An additional result, also called into question, is that the WTP for additional risk reductions rises (e.g., after the first 1/100 risk reduction takes place, the WTP is asked for an additional 1/100 risk reduction). Based on the "law of diminishing returns," one might have expected WTP to decline with additional risk reductions.

The remaining literature reviewed in this section addresses accidental death. Magat, Viscusi, and Huber (1990) obtained exceedingly large estimates of VSL based on reductions in risks of death in an auto accident. A sample of 195 adults in Greensboro, North Carolina, were led through a computerized game, being asked a series of questions to determine their indifference between living in one of two cities differing only in risks of death and cost of living. For a risk reduction of 1/100,000, the average VSL was $8.2 million, with a median VSL of $2.3 million. Seventy percent of respondents provided implied VSL's under $4 million. Whether these high responses are an artifact of the interactive computer procedure or the use of the cost-of-living indicator to elicit WTP, or occur for some other reason, is unclear.

Garen (1988) examined the question of selection bias in wage-risk studies using complex econometric techniques to correct for the possibility that people with risk preferences self-select into such jobs (and people with risk aversion select low risk jobs). This study yielded an average VSL of $9.2 million, compared to $4 million when standard regression techniques are applied to the same data base. This indicates that correcting for the voluntary, self-selecting nature of workplace risk raises the VSL. Whether this estimate is appropriate in the context of environmental risk depends on the degree to which such risks are involuntarily borne. To the extent that people can take action to limit their mortality risk (such as by avoiding living in a polluted area), these risks are analogous to those faced by workers, who can choose jobs; thus, Garen's estimate would not be appropriate. However, for cases of accidental, lethal releases of pollution, Garen's wage-risk self-selection-corrected estimate is probably reasonable.

4.3 STUDIES VALUING LIFE-YEARS SAVED

It is questionable practice to apply VSL estimates derived from prime-age males and for accidental risks of death to valuing the benefits of reduced mortality from cancer and other pollution-related diseases which occur mainly through exposure of older people. Some progress on obtaining VSL's that are more appropriate for environmental issues has come from Viscusi and Moore (1988), although they were interested in a different issue. This wage-risk study addresses the issue of life-years lost by estimating a model with the standard mortality risk measure weighted by the (discounted) remaining years of life of each member of the sample. The VSLs are found to be about $6 million (with an average risk reduction of 5/100,000), with an average value of an expected life-year of $170,000 (1986 dollars), along with an estimated rate of time preference of about 12%. Although this estimate of a value of a life-year is refined over studies that do not take years of life remaining into account, it still only applies to prime-age workers exposed to mortality risks that do not feature latency effects.

One recent, if ad hoc, attempt to apply the value of a life-year saved to estimate benefits in an environmental context is in Harrison et al. (1990). This study first estimates that fatalities in occupational accidents result in the loss of 40 years relative to average life expectancy.[2] Dividing this estimate into estimates of VSL from the wage-risk literature yields an estimate of the value of a year of life saved. This estimate can then be multiplied by average life-years lost from premature deaths caused by pulmonary disease, cardiovascular disease, and cancer, which is estimated to be about 12.5 years. This procedure results in a VSL based on life-years-lost (LYL) that is only 31% of the VSL derived from wage-risk studies.[3] It should be recognized that this type of correction makes very specific, strong assumptions about preferences for life years lost; for example, it assumes that WTP varies linearly with life-years lost. The appropriate approach is to infer values from the behavior of (or apply

[2]Assuming the average male worker is 37 years old, life expectancy for 37 year old males is 77 years.

[3]For instance, using a VSL of $3.7 million from wage-risk studies results in a LYL-based VSL of $1.16 million.

CV techniques to) a population of people who would be at risk from environmental diseases and related premature mortality. To our knowledge this has not yet been done.

4.4 DERIVATIVE STUDIES

Hall et al. (1990), in their review of the literature for a benefits analysis of air quality improvements in the South Coast Air Basin, settle on VSLs of $1.7 million (the lowest estimate supported by defensible research), $3.7 million [based on the Viscusi (1986) conclusion that this estimate was at the lower end of the environmental risk range and a representative estimate of wage-risk estimates], and $8.6 million (as the upper end of the environmental risk range and the wage-risk studies). The Garen (1989) estimate of $9.2 million is rejected on the basis of residential mobility.

EPA (1990) estimates the benefits of the Clean Air Act Amendments of 1990, incorporating a VSL of $3 million.

4.5 OTHER CLASSIFICATIONS

There is virtually no information on how WTP for risk reductions might vary by the type of death involved. The only empirical study (Harrington 1984) used a life-cycle model and life tables to estimate the ratio of WTP for reductions in the risks of dying by cancer or cardiovascular disease relative to dying in an accident. Because accidental risks are borne more evenly throughout one's life than risks of death by cancer or cardiovascular disease, reducing risks of the latter is valued less than the former.

One study of wrongful death judgements involving 95 cases (Bovbjerg et al. 1989) is included for completeness. The jury award approach involves compensation for identifiable deaths not for small risks of deaths to unknown individuals. Thus, the application of such values to the case at issue in this report is problematical. The mean award is $1.2 million.

Finally, Needleman (1976) indirectly values WTP to reduce risks of death in someone else's life through analysis of individual's willingness to donate a kidney to their relative. He finds that individuals value reductions in their relatives risk of death at 45% of their own, on average.

5. CONCLUSION

In spite of this voluminous and rich literature, its use to value mortality risks from environmental causes is not without problems, because nearly the entire literature addresses risks of accidental death taken voluntarily. The WTP to reduce accidental death caused by voluntary activities (in doing one's job or while driving a vehicle, for instance) may be viewed differently from death due to environmental pollution exposure (such as cancer from air pollution exposure) that is largely of an involuntary nature. Even if the nature of the death were unimportant to valuation, other factors would argue for hesitancy in applying this literature to environmental problems: the presence of differences in life-years lost (fewer with environmental risks than with workplace risks, for instance), in the age at which changes in risks are experienced (older people with environmental risks), and in latency.

In spite of this voluminous and rich literature, its use to value mortality risks from environmental causes is not without problems, because nearly the entire literature addresses risks of accidental death taken voluntarily.

Among the studies reviewed, Fisher, Chestnut, and Violette's (1989) review of 21 studies is probably least objectionable. Their range of estimates, based largely on wage-risk studies, is also comparable in order of magnitude to most of the other better studies. Thus, we generally adopt Fisher, Chestnut and Violette's (1989) estimates of the value of a statistical life to range from $1.8 to $9 million (1989 $) with a best estimate of $3.5 million.

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PAPER NO. 11*

THE BENEFITS OF REDUCED MORBIDITY

1. INTRODUCTION

Morbidity benefits refer to increases in utility arising from reductions in incidents of acute health impairments and from increases in the probability of developing chronic diseases. The impairments would run the gamut from a cough-day to a bed-disability-day, while the chronic diseases include classic pollution-related diseases, such as cancer, to *in utero* effects and learning disabilities. As with mortality benefits, there could be benefits to oneself and family and friends as well as benefits based on altruism.

A major difference between the mortality and morbidity valuation literatures is that while estimates of the former are always based on risk (one is never trying to obtain values for avoiding certain death), estimates of the latter generally are not. That is, most of the theory and empirical estimates are based on models where the effect to be avoided is certain. This assumption holds reasonably well for estimating common, acute effects, for example, the willingness to pay (WTP) for one less cough-day. It works less well, if at all, for chronic illness endpoints, where benefits seem to be appropriately expressed in terms of reduced risk of developing a disease or impairment.

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*Based largely on a working paper by Alan Krupnick, Carter Hood, and Ken Harrison.

2. THEORETICAL MODELS

Harrington and Portney (1987) have developed an accessible model for use in estimating WTP for certain reductions in acute health effects. In this model, as discussed in Freeman and Cropper (1989), WTP is the sum of the value of lost time (the wage multiplied by the net change in illness time as a result of the change in pollution);[1] the disutility from being ill (the money value of the effect of sickness on utility multiplied by the change in sick time, determined by the dose-response function); and the observed changes in expenditures on averting and mitigating activities. Thus, WTP is composed of the value to the individual of avoiding the sick time he actually experiences, which is the value of sick time multiplied by the quantity of sick time, plus the expenditures on medical care, drugs, etc., taken to mitigate the effects of being sick, plus any expenditures taken to reduce sick time or severity, such as using an air conditioner to avoid being exposed to pollution.

It is worth emphasizing that only epidemiological (as opposed to clinical) dose-response functions are appropriate to use in this formulation. These functions capture the effect of pollution on health after all averting and mitigating actions have been taken. Thus, averting and mitigation expenditures can be added to the value of reducing sickness without double-counting. In contrast, clinical (laboratory) studies provide estimates of health effects where avoidance and mitigation is impossible. Thus, adding averting and mitigating expenditures to estimates of sickness reductions from clinical studies would overestimate benefits.

Also, note that the aforementioned expression provides an estimate of individual WTP. Social WTP exceeds this for two reasons. First, the after-tax wage is the appropriate value of time for the individual, because this is the amount lost when 1 hour of work is traded for 1 hour of leisure. However, the individual's contribution to society's material output, his marginal product, is his before-tax wage, which exceeds his after-tax wage. Second, many individuals do not pay the full cost of medical care or sick leave.

There are few formal models for reducing the risk of having a chronic disease. Berger et al. (1987) develop a model where the probability of having a chronic disease depends on pollution and averting expenditures, but this model does not permit one to assess the morbidity effects associated with a possibly fatal disease. Magat, Viscusi, and Huber (1991) and Krupnick and Cropper (1992) assess WTP directly by asking individuals to trade income for reductions

[1]That is, the value of time is assumed to be equal to the wage and this is multiplied by the amount of time one is sick, determined by a dose-response function.

in the risk of developing a chronic disease. A version of the Harrington and Portney model would seem to apply to chronic illness as well, in that the value of avoiding pain and suffering, medical expenditures, avoidance expenditures and wage/leisure effects would also be relevant. In this spirit, other authors (Hartunian 1980; Oster et al. 1984; and Cropper and Krupnick 1992) have examined one or more of the components of the cost of illness, such as medical costs and reductions in wage rates or work time as a result of chronic illness.

3. ESTIMATION APPROACHES

There are three approaches for estimating WTP for acute morbidity reductions: cost of illness, contingent valuation, and averting behavior.

3.1 COST OF ILLNESS (COI)

The COI or distress includes medical expenditures, work and productivity losses, and leisure losses. Estimates of WTP based on the cost of illness should be regarded as underestimates of "true" WTP because they ignore the values related to pain and suffering and avoidance behavior. Several studies compare COI and WTP estimates. Rowe and Chestnut (1985) find for asthma a WTP/COI ratio of 2:0; Dickie et al. (1987), for ozone symptoms finds a ratio of 5:11; Schecter (1990) finds a ratio of WTP to an estimate obtained by an indirect method (hedonics) of 3:0.

Estimates of WTP based on the cost of illness should be regarded as underestimates of "true" WTP because they ignore the values related to pain and suffering and avoidance behavior.

The medical costs of an incident of illness or symptoms can be valued using either an expenditure or a frequency approach. The former starts with aggregate national data on medical expenditures and uses supplementary information to allocate these expenditures to diseases as desired. For instance, the Regulatory Impact Analysis for Particulates (Manuel 1982) allocated total medical expenditures on all diseases to acute and chronic respiratory diseases. These expenditures were further broken down geographically. In contrast, the latter approach estimates medical costs from knowledge of the price and frequency of use of medical services. The latter is often estimated from national surveys, as well.

Work, productivity, and leisure losses can be valued with data on the probability (or frequency) that a disease or symptom will result in such losses; and data are on the value of a unit loss. For example, a work-loss-day is usually valued at the daily-wage rate, under the assumption that workers are paid their marginal product. This assumption only holds under pure competition and may be difficult to estimate accurately in many markets. In addition, the value of work losses by homemakers, students, the elderly, volunteer workers, the unemployed, and others must be estimated. Problems in valuation are compounded when values are sought for productivity and leisure time losses. Productivity losses involve situations where workers are too sick to work at full capacity, but not sick enough to stay at home. Defining a measure of capacity utilization for labor is a complex issue. The valuation of leisure time also involves exceedingly difficult and currently unresolved questions about the value of inframarginal time. Marginal leisure time, where employees have some control over their number of hours worked, can be reasonably valued at the wage rate. Because workers can nearly always work as long as they want (for non-overtime pay), but not for a period shorter than some minimum, the value of marginal leisure time in this more realistic case is probably more than the wage rate.

3.2 CONTINGENT VALUATION

The contingent valuation approach is often applied by asking people how much they would be willing to pay for reductions in health effects. This approach may be the only approach for estimating values for pain and suffering. In addition, in principle, it can capture the WTP of an individual for health improvements in the general public. However, it may leave out certain costs to society not incurred by the ill individual, such as medical costs paid by health insurance. In CV studies it is always problematical to be sure that budget constraints are being invoked and that averting and mitigating behaviors are being taken into account by respondents.

3.3 AVERTING BEHAVIOR

The averting behavior approach requires estimating a "health production function," relating health to actions taken to improve it. If one buys bottled water to avoid ingesting carcinogens, the value of this purchase is at least equal to the WTP to avoid the risk of cancer. This approach is problematical because the averting behavior that one needs to estimate is generally not the observed behavior; rather it is the behavior that would return health to its original state. In addition, averting behaviors often have joint benefits. For instance, air conditioners convey benefits beyond reducing health effects. Allocating all the costs of running an air conditioner to health benefits would produce an overestimate of WTP. However, there is still considerable debate in the economics profession over whether an estimate of WTP using the averting behavior approach under or overestimates true WTP (Bartik).

The averting behavior approach requires estimating a "health production function," relating health to actions taken to improve it.

For estimation of benefits to reducing risks of chronic illness, there are some additional approaches being used. Magat, Viscusi, and Huber (1991) developed a contingent ranking survey approach installed on a personal computer. The respondent is led through a series of choices involving tradeoffs between risks of developing a chronic disease and another "commodity," either risk of dying in an auto accident or an increase in the cost of living. WTP for lowered risk of developing chronic disease can be determined directly using the responses to the tradeoffs involving the cost of living and indirectly by assigning a value to a statistical death using the tradeoffs involving risk of death in an auto accident.

When applying the cost of illness approach to chronic illness, a distinction is made between an estimate based on prevalence and one based on incidence. The prevalence-based approach involves estimating the yearly cost per person of having a disease. One can simply stop at this point for an annual disease cost or obtain an estimate of the lifetime cost of the disease by multiplying the annual cost by the number of years one has the disease (on average) and applying the appropriate discount factor.

The incidence approach is superior because it examines cost over the life cycle of the disease [either from actual data for individuals or from "disease and treatment profiles" developed by physicians and costed out (Abt, Forthcoming)]. If these costs occur unevenly over time, the present discounted value of the time-stream of these expenses is likely to be quite different than that obtained from the prevalence approach. Unevenness is likely to be a feature of many types of chronic illness. For instance, large medical expenses may occur (on average) in the diagnostic phase and near the end of the life cycle with very low expenditures otherwise.

4. SURVEY OF THE LITERATURE

Table 1 summarizes 35 studies addressing the valuation of reductions in various measures of morbidity. These measures include acute health effects, such as symptoms, restricted activity days, and asthma attacks (both original and derivative studies); chronic diseases associated with pollution, such as cancer, respiratory diseases, heart disease, and IQ loss from lead exposure; and injuries from auto and on-the-job accidents. One study is included because it estimates altruistic values. Unfortunately, the endpoint being valued is of little importance to fuel cycles -- poisoning prevention. There are no surveys of the literature that address as broad a set of morbidity measures as is listed here, although Cropper and Freeman (1990) examine some of these measures.

Unlike the mortality studies, some of the estimates presented in the table are not complete measures of WTP, capturing only life cycle medical costs or labor market effects, for example. Studies that present values easily obtainable for the reference environment, for example, those valuing only doctor visits, hospital visits, or work loss, are ignored in the table and this review. These values are easily obtainable for the area in question.

Just two studies using the hedonic property value method are included in the table. This approach, because it short-circuits dose-response functions and yields values that combine all environmental effects of pollution into the implicit valuation measure, is given less emphasis in this project. Smith and Huang's working paper that provides a review and meta-analysis of these studies (1991) is reviewed in the following paragraphs and included in Table 1.

Each study is described by nine characteristics: author and year, city and region sampled, estimation approach, effect being valued, payment vehicle (if CV study), sample size, baseline condition, scenario, and valuation result. In the following sections, the categories of studies are briefly reviewed. End notes complement the tabular entries.

4.1 ACUTE EFFECTS

Both the CV approach and the COIs approach have been used with some success to value acute respiratory symptoms and asthma attacks. Acute illness may be the easiest morbidity effect to value because of its frequent occurrence. For example, people are familiar with the good being valued and questions on its valuation need not be phrased to incorporate risks (as they should be where developing chronic disease or dying prematurely is at issue). Rather, one can straightforwardly ask for WTP for a reduction in a day of symptoms. Three CV studies (Loehman et al., Tolley et al., and Dickie et al.) have estimated values for respiratory symptom days, with average estimates ranging from $1 to $36 (1989 dollars) and more, on average, depending on the symptom, its severity, and whether it appears as a complex of symptoms.

Table 1. Key features of valuation of morbidity studies

Author/ Year	City/ Region	Estimation Approach	Effect Being Valued	Payment Vehicle	Sample Size	Baseline Condition	Scenario	Valuation (1989$'s)
ACUTE EFFECTS								
Rowe & Chestnut (1985)[a]	Glendora, California	Resource Cost - (Medical costs, drugs)	Asthma attack	NA	64 adults	74 attacks per year	Cost of preventing 1 attack	Mean value: $7.45/attack/person/year
		Contingent valuation: WTP	Asthma attack	Increased taxes	82 asthmatics (65 used)	Number of attacks experienced each year	Reducing number of attacks by one-half	Mean value: $508 for 50% reduction in attacks; $30/attack-day LB=$11; UB=$49
Krupnick (1988)[b]	United States	Resource cost (medical costs)	Asthma attack	NA	NA	Present number of attacks per year: 9.9	Cost of preventing 1 more attack	Given attack rates of 55.8, 9.9, and 7.3: $1.54-$43.08 per attack/person; best est.: $28.94
Loehman et al. (1979)[c]	Tampa Bay Area	Contingent valuation WTP	Respiratory symptoms: mild & severe; 1, 7, 90 days	Payment card	404 of 1800 in mail survey	Respondent's present number of symptoms per year	Cost of avoiding 1, 7, and 90 days of symptoms	For single day median value: ($/person/day) Minor cough: $5; Severe cough: $13
Tolley et al. (1986)[d]	Chicago & Denver	Contingent valuation: mitigation evoked; bid from start point of $100	"Light" symptoms	From family budget	199 household heads	Respondent's memory of last year's health coughs-day/yr=7.65	Cost of avoiding 1 or 30 days of "light" symptoms described in detail	1-day "light" symptom survey; median: $13 cough; combo of congestion, sinus, cough: $36.40
	Chicago & Denver	Contingent valuation: mitigation evoked; bid from start point of $100	Mild and severe angina	From family budget	199 household heads	4 baseline: 10 mild days 10 severe days 20 mild days 20 severe days	Avoiding 1 day of angina given baselines at left	Mean WTP to avoid: 1 mild day: $97 1 severe day: $167 1 mild day: $114 1 severe day: $241

Table 1. Key features of valuation of morbidity studies

Author/ Year	City/ Region	Estimation Approach	Effect Being Valued	Payment Vehicle	Sample Size	Baseline Condition	Scenario	Valuation (1989$'s)
Dickie et al. (1986a)[e]	Glendora & Burbank, California	Contingent valuation	Respiratory symptoms	Payment card	229 adults; non-smoke; mostly males	Present condition	Avoiding symptom	Mean value: ($/person/day) cough: $161 (normal group) cough: $236 (impaired group)
		Averting behavior: price of averting good/good's marg. prod. of avoiding symp.	Symptoms	NA	229 adults	Present condition	Avoiding symptom	Mean value: ($/person/day) cough: $3.0 (normal group) cough: $9.0 (impaired group)
Dickie et al. (1986b)[f]	Not reported	Averting behavior: price of averting good/good's marg. prod. of avoiding symp.	MRRAD	NA	Not reported	Present number of MRRADs	Avoiding 12.12 MRRADs by home AC; 4.2 MRRADs by automobile AC	Mean WTP to avoid one MRRAD: UB=$40; Best estimate=$23 - $34
Dickie et al. (1987)[g]	Glendora & Burbank, California	Contingent valuation: revise daily bids when presented as WTP for month	Symptoms	Not reported	221 residents in phone survey	Previous occasion respondent experienced symptoms	Avoiding one day of this symptom	Mean value: ($/person/day) cough: Initial Bid: $401; Revised Bid: $1.58
Krupnick (1988)[h]	NA	Use symptom day results	MRRADs	NA	NA	NA	One MRRAD	($/person/day): $18.65; (LB=$11.90; UB=$36.40)
Chestnut et al. (1988)[i]	Los Angeles, California	Medical costs; wages lost; defensive expenditures	IHD and angina	Not specified; phone	50 men with IHD	1 episode per week	IHD case/year Angina episode	Medical: $4,741 Wages: $10,043 WTP: $42 Annl defnsv exp: $2,255 for each of 21 subjects

Table 1. Key features of valuation of morbidity studies

Author/ Year	City/ Region	Estimation Approach	Effect Being Valued	Payment Vehicle	Sample Size	Baseline Condition	Scenario	Valuation (1989$'s)
ACUTE: DERIVATIVE								
Krupnick (1988)[j]	NA	Attempt to choose most defensible estimates from above symp. studies	Acute symptom effects	NA	NA	NA	One symptom day avoided	($/person/day) cough: $3.95 best est.; (LB=$1.38, UB=$12.44)
Krupnick (1988)	NA	Same	Asthma day	NA	NA	NA	One day	($/person/day) best value: $30; (LB=$11; UB=$49)
Krupnick & Kopp (1988)[k]	NA	Author's best judgement	Symptoms, MRRADs, asthma day	NA	NA	NA	One day	Best value: ($/person/day) any symptom day: $6; (LB=$3; UB=$12) MRRAD: $22; (LB=$13; UB=$36) asthma attack: $30; (LB=$11; UB=$49)
Hall et al. (1989)[l]	NA	Adjusts value in lit by concave function of # of symptom days avoided per person	Acute health effects	NA	NA	NA	Value of multiple days of symptom reduction	WTP for preventing N attacks equals $N^{0.5}$xWTP for preventing 1 attack (i.e. average WTP for preventing N attacks = $N^{-0.5}$xWTP for preventing 1 attack)
NERA (1990)	South Coast Air Basin	Use Krupnick and Kopp	MRADs, asthma attack, any symptom	NA	Not reported	NA	Cost of preventing 1 more MRAD, asthma attack, or symptom day	See Krupnick and Kopp RAD: daily wage
RER (1990)	NA	Use Krupnick and Kopp	Symptoms, MRADs, asthma	NA	NA	NA	NA	Authors cite Krupnick (1986), Brucato et al. (forthcoming), Rowe et al. (1986)

Table 1. Key features of valuation of morbidity studies

Author/ Year	City/ Region	Estimation Approach	Effect Being Valued	Payment Vehicle	Sample Size	Baseline Condition	Scenario	Valuation (1989$'s)
Rowe et al. (1985)	California	Literature	Symptoms, hosp admis, ER visits					RAD: daily wage
EPA (1988) (RIA)	31 Eastern States	Rely on Mathtech Model (1982) for TSP	Acute & chronic morbidity	NA	NA	NA	NA	Benefit from reduction in TSP (annual) $3.83/person/$\mu g/m^3$
EPA (1990)	Varies	Literature	Asthma Mortality Resp symp				Asthma symp==> VSL ==> symptom day==>	$0-$50 $3 mill. Krupnick & Kopp
CHRONIC DISEASE: MEDICAL AND LABOR MARKET COSTS								
Cooper & Rice (1976)	United States	Medical cost of illness Productivity losses from illness	Considers 16 disease categories	NA	Survey of healthcare providers in 1972	NA	Total direct medical costs, and productivity losses	<u>Medical costs/person/yr</u> Circ dis: $525 Resp dis: $280 <u>Productvty loss/pers/yr</u> Circ dis: $310 Resp dis: $335
Hartunian (1981)	United States	Medical cost of illness; incidence based; life cycle	Cancer, stroke, coronary heart disease, vehicle injuries	NA	NA	NA	Average total cost per incident	Medical life cycle cost: <u>Resp cancer:</u> $22,200 <u>Stroke:</u> $20,550 <u>Heart disease:</u> Angina: $3,120 Heart attack: $12,970 Ave hrt dis: $8,730 <u>Vehicle accidents:</u> $14,400 (all @ 6% disc. rt)

Table 1. Key features of valuation of morbidity studies

Author/ Year	City/ Region	Estimation Approach	Effect Being Valued	Payment Vehicle	Sample Size	Baseline Condition	Scenario	Valuation (1989$'s)
Krupnick & Cropper (1989)[m]	United States	Medical and labor market costs from individual data; hedonic wag est.; NMCES data; some life cycle costs	Chronic heart and lung disease	NA	Varies depending on symptom[n]	NA	Total medical costs (not social costs) per case	Mean medical cost/case/yr Heart attack: $2,570 Bronchitis: $198 Emphysema: $1,295 Hypertension: $442 Medical life cycle costs Heart disease: $19,226 Emphysema: $6,598 (5% discount rate) Annual lost earnings if working at age 55-65 Heart attack: $12,746 Emphysema: $14,474
NHLBI (1982)[o]	United States	Medical costs	Various chronic illnesses	NA	NA	NA	Total medical costs per case	Mean medical cost/case/yr: Bronchitis: $241 Emphysema: $209
Freeman et al. (1976)[p]	United States	Medical costs	Emphysema	NA	NA	NA	Total medical costs per case	Mean medical cost/case/yr: Emphysema: $479
Oster et al. (1984)[q]	United States	Life cycle medical costs	Emphysema	NA	NA	NA	Total lifetime costs	Life cycle cost for emphysema: $8,561
EPA (1985)	United States	Ad hoc	Lead exposure	NA	NA	NA	Cost per case above 24 μg/dL	Medical costs and compensatory educ: $6,461/case; hypertension cost: $284 per case/year
CDC (1991)	United States	Ad hoc	Lead exposure	NA	NA	NA	Cost per case above 24 μg/dL Cost per 1 μg/dL	Medical costs and compensatory educ: $4,631/case; IQ earnings:[r] $1,147/μg/ml Infant death using VSL=$3 mill.: $300/μg/ml

Table 1. Key features of valuation of morbidity studies

Author/ Year	City/ Region	Estimation Approach	Effect Being Valued	Payment Vehicle	Sample Size	Baseline Condition	Scenario	Valuation (1989$'s)
CHRONIC: WTP								
Viscusi, Magat, & Huber (1991)	Greensboro, North Carolina	Contingent ranking in choice of cities with CB risk tradeoffs	Chronic bronchitis	Computer game	111-254 shoppers from mall	55-75/100,000 chronic disease risk	Lower risk for higher COL or high risk of premature death	Value of a stat case: mean: $930,000; median: $460,000. VSC bronc to VSL ratio: mean: 0.62-0.68; median: 0.27-0.32
Krupnick & Cropper (1992)[1]	Washington, DC Metro Area	Contingent ranking in choice of cities with chronic disease risk tradeoffs	Chronic bronchitis and other chronic respiratory diseases	Computer game-same as above	70-77 adults with relatives with resp. disease	55-75/100,000 chronic disease risk	Lower risk for higher COL or high risk of premature death	Median value/stat case (using VMH model) VMH: $0.46 mil; This study: $1.06 mil; VSC bronc to VSL ratio: 0.31-0.39, in comparable terms
Viscusi, Magat & Forrest (1988)	Greensboro, North Carolina	Contingent valuation	Non-fatal poisonings	Not given	785 adults	15 poisonings/ 10,000 bottles of insect spray used	Reduction to 10/10,000 and to zero/10,000	Mean private value: $2,180-$3,870/statistical case; with altruism: $14,300-$24,400/statistical case
NON-FATAL INJURY								
Rossman, Miller & Pindus (1990)	United States	Descriptive statistics	Medical costs of spinal injuries; motor vehicle & occupational injuries by body part		NA	NA	Life cycle cost per case	e.g.: medical costs for upper extremity $19,000 (4% disc rate). Also, has legal fees, pain and suffering, legal costs, work loss HH production loss, travel delay, employer costs.

Table 1. Key features of valuation of morbidity studies

Author/ Year	City/ Region	Estimation Approach	Effect Being Valued	Payment Vehicle	Sample Size	Baseline Condition	Scenario	Valuation (1989S's)
INDIRECT VALUATION								
Loehman (1979)[u]	San Francisco	Contingent valuation	Specifics not provided	Not reported	Not reported	Not reported	WTP for reductions in air pollution	WTP for health 1/2 of total for reducing air pollution
Brookshire (1979)	Los Angeles	Contingent valuation	Chronic & acute health effects	Not reported	Not reported	Not reported	WTP for reductions in air pollution	Health effects 2/3 of total WTP for change in air pollution
Schecter & Kim (1991)	Haifa, Israel	Indirect welfare estimation Contingent valuation	50% air quality improvement 50% air quality improvement	City property taxes City property taxes	2000 hseholds Subset of same 2000 hseholds	Perceived pollution levels "Moderate" and "poor" air quality	50% improvement in AQ Change to "good" and "moderate" AQ	WTP ($/household/hr) Mod-->good (N-750): Mean: $25, Med: $19; Poor-->mod (N=192): Mean: $32, Med: $27
Schecter (1991)	Haifa, Israel	Contingent valuation Physician demand	50% air quality improvement Acute/visibility	City property taxes NA	3,500 hseholds (81% participation 954 households	Perceived pollution levels	50% improvement in AQ 50% change in air quality	WTP ($/household/hr) Open-ended bid: $25; Iterative bid, first: $17, revised: $45; Binary choice: $47 ($/household/yr) Preference model: $7; Health production: $60; Cost of illness: $123
Gerking & Stanley (1986)	St. Louis, Missouri	Compen vari derived from health-oriented choice model	Ozone reduction	Not given	824 employed indiv.	Existing ozone level (mean = 19 ppb?)	30% reduction in mean ozone	WTP/person/yr: $21.26 - $28.21
Dickie & Gerking (1991)	Glendura & Burbank, California	Medical care demand	Ozone reduction	NA	Glendura: 151 indiv. Burbank: 75 indiv.	# days with O_3 > 120 ppb Glendura: 117 Burbank: 87	WTP for no days w/O_3 > 120 ppb reduced medical cost of O_3 redu	Glen: $171-209/pers/yr Burb: $ 95-115/pers/yr Glen: $ 46-100/pers/yr Burg: $ 25- 58/pers/yr WTP to med-cost ratio: 2:1 - 4:1

Table 1. Key features of valuation of morbidity studies

Author/ Year	City/ Region	Estimation Approach	Effect Being Valued	Payment Vehicle	Sample Size	Baseline Condition	Scenario	Valuation (1989$'s)

a. Estimates WTP as function of number bad days reduced. Marginal valuation falls.
b. RFF did not directly study a group of asthmatics. Instead, the authors used existing data from secondary sources and certain assumptions to arrive at a range for marginal resource cost for a given person per asthma attack. For assumptions see Krupnick and Kopp, 1988.
c. The types of respiratory symptoms include shortness of breath, coughing and sneezing, and head congestion, including eye and throat irritation. WTP found to increase at decreasing rate with number of symptom days avoided, and increase with income and baseline illness days. Females bid more than males. Mean bids were exceedingly large because of outliers, e.g.,
d. Light symptoms include coughing, sinus congestion, throat congestion, watery eyes, headaches, drowsiness, and nausea, plus various combinations of symptoms. WTP to avoid a marginal day of illness was found to increase with baseline days ill. No socioeconomic variables were significant. Variations in WTP as a function of number of days avoided were not examined. A number of participants offered WTP of zero for unmitigated headaches.
e. Persons with chronic respiratory disease made up approximately 30% of the sample size. Types of respiratory symptoms included: being unable to breathe, experiencing pain on deep inhalation, being out of breath easily, wheezing, having a tight chest, coughing, having chronic throat irritation, having sinus pain, and experiencing severe headaches. The analysis the authors conducted was on normal, impaired and combined samples. Median values for avoiding symptoms were not reported by Dickie.
f. The impaired group, which accounted for 30% of respondents, consisted of respondents with asthma, bronchitis, emphysema, or other chronic respiratory diseases.
g. WTP regressions "not particularly strong" according to the study's authors. Bid revision procedure is problematic. For cough avoidance, the median initial bid was $1.13 and the median revised bid was $0. Respiratory symptoms included: being unable to breathe deeply, experiencing PDI, having shortness of breathe, wheezing, having a tight chest, coughing, experiencing throat irritation, having sinus pain, and having a headache. The results are presented only for one of the many respiratory diseases examined, namely sinus pain. Note: not everyone in the sample valued each of the symptoms.
h. Loehman lowest median severe symptom is LB; highest median symptom is best; Tolley 3-symptom combo trimmed mean for UB.
i. No statistical relationship between medical/wage costs and number of angina episodes. 20% non-responders to CV.
j. Krupnick et al. (1988) reviewed the following studies to estimate the health endpoints: Rowe and Chestnut (1985), Loehman et al. (1979), Tolley et al. (1986), and Dickie et al. (1986a, b; 1987).
k. The authors attempt to standardize the valuation of a symptom day for coughing, shortness of breath, chest tightness, eye irritation (combined into one category); a minor respiratory related restricted activity day; and an asthma attack.
l. Hall et al. develop a WTP function for the reduction of multiple symptom days ($WTP_N = WTP_1 \times N^{-0.5}$) using data from Loehman et al.; Rowe and Chestnut; and Dickie.
m. Values are reported for only some of the several chronic diseases included in the study. 5% discount rate, white males for life cycle costs.
n. The number of individuals included in the estimates of medical costs varies by the condition. The total sample to estimate medical costs consists of 4,789 individuals who suffered an episode of illness.
o. Comparable annual medical costs from Krupnick and Cropper are $61 for chronic bronchitis and $649 for emphysema.
p. This figure is based on the 1970 prevalence of emphysema.
q. This life cycle estimate is for all ages and both sexes combined, discounted at 3%.
r. Uses results from a wide variety of statistical studies linking IQ to wage rates and labor force participation. Infant death valuation based on VSL of $3 million (see text).
s. This is the implicit dollar value per statistical case of chronic bronchitis, analogous to a value of a statistical life. From contingent ranking of cities with auto death-chronic bronchitis tradeoffs, ratio of mean VSC chronic bronchitis to VSL is: .67-.8; median: .27-.4.
t. Krupnick and Cropper find that those with relatives who have chronic respiratory disease are WTP significantly more to reduce their risks of getting the disease. Significant variables: "women", "have children", and "non-smoker" pay more.
u. This study is a replication of Brookshire et al. (1979).
v. No specifics were provided on the health impacts of a 30% reduction in mean ozone.

All of these studies have significant drawbacks, mainly related to their age — the CV studies were performed before many of the most important advances in CV methodologies. At the same time, they offer quite consistent ranges of estimates for WTP to avoid a particular type of symptom.

One problem in the use of these studies to estimate population benefits is that most studies simply multiply the total number of symptom-day reductions by the relevant unit values to obtain benefits. This may be incorrect if one assumes (with some empirical justification) that marginal valuations decline with additional days illness reduced. Hall et al. (1989) pool the WTP estimates from asthmatics in the Rowe and Chestnut study and for respiratory symptom reductions from the Loehman study to estimate WTP as a function of days sick. This function is $WTP_N = WTP_1 * N^{-0.5}$, where WTP_1 is the WTP for avoiding one symptom day, WTP_N is the average WTP for each of N days avoided, and N is the total number of days avoided. Using this equation, a person who would pay $1 to avoid a cough-day would be willing to pay $4 to avoid 16 cough-days, or an average of $0.25 per cough-day.

Several caveats are in order, however. The distribution of symptom-days for each person cannot be estimated from the data but must be determined by dividing total estimated days reduction by population. Second, the studies finding declining marginal WTP are unclear about whether these days of reductions are to be experienced continuously or spaced over 1 year. WTP responses would likely be quite sensitive to this spacing. Thirdly, outside of the Los Angeles area, and for small enough changes in ambient air quality, N may be less than 1.0, which would mean that the Hall et al. procedure would raise WTP above that obtained when N is assumed to equal 1.0. Is this reasonable, since no one actually experiences half a symptom-day? Finally, the estimated decline in marginal WTP is sensitive to assumed functional form, but there is too little information in the literature to estimate such functions confidently.

Other acute effects that have been valued include angina and minor restricted activity days (RAD). Numerous authors have applied ad hoc approaches to valuing various types of restricted activity days. Rowe et al. (1985) followed standard practice in assigning the average daily wage (before taxes to reflect lost value to the economy) to a RAD. Minor restricted activity days (MRADs) in this study were arbitrarily valued at one-fourth of a RAD. Krupnick (1987) assumed that the restriction in activity occurs because of symptoms experienced by the ill person, therefore valuing minor respiratory restricted activity days (MRRADs) using severe symptom values ($36.40 in 1989$). Hall et al. (1989) valued a RAD for workers at the daily wage and a RAD for nonworkers at the mid-point of an MRAD. MRADs were valued using similar logic and to similar conclusions as that of Krupnick (1987).

The only study that attempted to value an MRRAD in a non-ad hoc manner is by Dickie et al. (1986b),[2] and even here, this measure had to be fabricated from other information. By relating the cost of installation of a home air conditioner to reductions in MRRADs, Dickie et al. find that the value of an MRRAD is $40—an estimate quite close to the "high" consensus estimate of the EPA's Ozone Benefits Work Group and used by Krupnick (1986) ($36.40). Given all of the problems with both estimates, this must be regarded as purely a coincidence.

Broader measures of RADs have been valued. Numerous studies have estimated values of an MRAD (median equals $22.50 in 1989 dollars) to arrive at low, medium, and high estimates of a respiratory restricted activity day (RRAD): $28, $45, and $73, respectively. Based on data showing that a RAD includes bed disability days, work-loss-days, and other RADs, RCG/Hagler, Bailly (1988) produced a best estimate of a RAD of $51 (1989 dollars).

4.2 STUDIES VALUING CHRONIC ILLNESS

There are three types of studies valuing chronic illness. The most prevalent are COI studies that look at the average annual medical costs associated with a given disease. Generally, these studies rely on aggregate statistics on costs and disaggregate to various specific diseases, although some studies attempt to characterize the medical interventions associated with a case of chronic disease, valuing each intervention and summing. A smaller literature attempts to estimate the medical costs associated with a given disease from its start to its cure or the person's death, where these costs are called life cycle costs.

A second set of literature contains studies estimating the effects of chronic illness on labor force participation and earnings. These costs can be added to medical costs for a partial estimate of WTP.

Finally, two studies use CV techniques to estimate WTP for reductions in the risk of getting chronic respiratory disease. These studies are analogous to contingent valuation studies addressing mortality risks.

[2]An MRRAD was defined as the absence of a respiratory bed disability day (RBDD) or a respiratory work loss day (RWLD) along with reduced effort in usual or planned activities.

4.3 MEDICAL AND LABOR MARKET COSTS—ANNUAL AND LIFE CYCLE

A recent study of medical and labor market costs is by Krupnick and Cropper. Krupnick and Cropper (1990) measured the medical and labor market costs of chronic heart and lung disease. These cost estimates are based on two national surveys of individual medical and labor market histories — the National Medical Care Expenditure Survey and the Social Security Survey of Disabled and Non-Disabled Adults. Medical costs include the costs of medication, doctors' visits, and hospitalization. Labor market costs include the lost earnings of people who stop working due to their disease and the reduced earnings of people who continue to work but cut back their hours or switch jobs.

The major finding in the study is that the costs of chronic illness vary greatly from one disease to another. Of the five diseases examined — hypertension, ischemic heart disease (IHD) (heart attack), other (nonspecific) heart disease, emphysema, and chronic bronchitis — medical expenses incurred from heart diseases dwarf those for some of the other diseases. For instance, expenses resulting from heart attacks (at $2,570 per person per annum in 1989 dollars) are almost 13 times higher than those from chronic bronchitis (at $198), but only twice as large as those from emphysema (at $1,295). Hypertension has no labor market effects and average annual medical expenses of only $442 per case. Of the five chronic heart and five chronic lung diseases studied, only six of the ten decrease the probability of working — emphysema, heart attack, and stroke being the most important, followed by chronic bronchitis, arteriosclerosis, and other heart disease. Emphysema, heart attack, and stroke cause the largest annual earnings losses; however, the time pattern of losses is very different for the three diseases. Losses due to a heart attack are largest in the 5 years following the attack (equal to 55% of earnings, on average) and then decline. Losses associated with emphysema do not begin until a person has had the disease at least 6 years and then, up to age 55, equal 73% of earnings. By contrast, asthma, allergies, and hypertension appear to have no effects on earnings.

Krupnick and Cropper also estimated life cycle medical costs of emphysema and heart disease in white males by age cohort and for all ages and compared some of these estimates with other estimates in the literature. Using a 5% discount rate, lifetime costs of emphysema are largest for the 65–74 cohort, at $12,078 and, curiously, smallest for the 55–64 cohort, at $6,159. Average lifetime costs over all cohorts are $7,636. These estimates are all in 1989 dollars.

The present discounted value of lifetime cost estimates for IHD shows that estimated costs decrease noticeably as age increases. The highest lifetime cost is for the 15–24 age group with an estimated cost of $51,888. The weighted average lifetime cost for all ages is $19,226.

Three additional studies provide cost estimates for respiratory diseases examined by Krupnick and Cropper. NHLBI (1982) estimate annual costs of chronic bronchitis and emphysema using a "top-down" approach while Freeman et al. (1976) use an engineering approach with aggregate data to estimate annual costs of emphysema. The third study, Oster et al. (1984), estimates life cycle costs for emphysema. Hartunian et al. (1981) estimates life cycle costs for heart disease.

The National Heart, Lung, and Blood Institute publishes estimates of the costs of specific chronic illnesses. These estimates are computed using a "top-down" approach (i.e., where the data manipulation begins with aggregate cost data, in this case, for medical services) rather than the micro-data approach used by Krupnick and Cropper. The annual medical expenses per case in 1977 were $241 and $209 for chronic bronchitis and emphysema, respectively (1989 dollars). These estimates contrast sharply with those of Krupnick and Cropper, who estimated costs per case of $198 and $1,295 for chronic bronchitis and emphysema, respectively. These differences are partly related to different estimates of prevalence. The Krupnick and Cropper estimate excludes individuals with zero medical costs, while the NHLBI estimates include them. Making various adjustments to make the former comparable to the latter yields an annual cost per case for chronic bronchitis and emphysema of $61 and $649, respectively for Krupnick and Cropper. These adjustments widen the gap between estimates for chronic bronchitis costs but lower the gap for emphysema.

Another estimate of the annual medical costs of emphysema is available from Freeman et al. (1976). Using data on health care utilization and average prices for 1970, they estimate costs to be $479 per case annually. These estimates are far lower than those of the NHLBI, and still lower than Krupnick and Cropper's conditional estimate ($1,295).

Oster et al. estimate life cycle costs of emphysema. They estimate that the life cycle cost for all ages and both sexes combined, discounted at 3%, is $8,561, in 1980 dollars. The comparable estimate from Krupnick and Cropper is $8,136 per person for white males.

Hartunian et al., estimate direct costs of coronary heart disease from the data obtained by the Framingham Heart Study (1974). The Krupnick and Cropper estimate of $17,183 for all IHD is close to Hartunian et al.'s for MI, $14,451, and CI, $17,395. However, there is a wide disparity between the estimates for angina pectoris uncomplicated (APU), with Hartunian's lower estimate for angina ($2,833) bringing down the weighted average cost for all

IHD to $10,395. Note also that although both studies show declining lifetime costs as age of onset increases, the Hartunian et al. decline is much "flatter" than Krupnick and Cropper's.

4.4 REDUCTION IN RISKS OF CHRONIC DISEASE

Both Magat, Viscusi, and Huber (MVH) (1991) and Krupnick and Cropper (1992) value reductions in the risks of chronic respiratory disease (the value of a statistical case — VSC) using a contingent behavior and contingent valuation survey. People were asked to choose between living in one of two cities identical in all respects, except the risk of developing chronic disease and either (in separate questions) the risk of dying in an auto accident or the cost of living (a money measure). With an interactive computer program driving their choices to a point of indifference between the cities, the VSC for the risk-risk trade was converted to dollars using the above-wage-risk literature, and the VSC for the risk-cost-of-living trade was obtained directly. The studies find VSCs less than the range of VSLs, with Krupnick and Cropper's best estimate of $1 million, obtained from a sample of people who, while themselves being healthy, each had relatives with chronic respiratory disease. Hence, they were familiar (to varying degrees) with the complex consequences of this disease. From a sample of people obtained in a shopping mall, MVH found mean VSC's similar to Krupnick and Cropper, although unfamiliarity with the disease in MVH's subjects related to significantly lower WTP from the trade off involving chronic disease risk and the cost-of-living.

4.5 LEAD BENEFITS

EPA (1985) estimated the benefits from reductions in blood lead levels, covering medical costs for chelation therapy and compensatory education in children, medical costs associated with hypertension, and foregone earnings estimates associated with myocardial infarction and stroke. Medical costs plus compensatory education costs were estimated to be $1,120 and $5,340 per child, respectively, in 1989 dollars, but only for a fraction of children (5% and 20%, respectively) with blood lead levels above 24 μg/dL. Compensatory education costs, which are estimated to occur over several years, are discounted to the present. For our study, this threshold would imply that only costs associated with children crossing the threshold because of the lead emissions of the incremental plant will count as incremental damages. An alternative assumption is that the probability of needing chelation therapy and of needing more extensive education is raised for those with baseline blood lead levels exceeding 24 μg/dL.

A more recent analysis (CDC 1991) focused strictly on the benefits of preventing lead exposure to children and fetuses, but included (in addition to medical costs and compensatory education), losses in lifetime earnings and reduced neonatal mortality. Medical costs and compensatory education were estimated for the average case to be $1,300 and $3,331, respectively, per child with blood lead exceeding 24 μg/dL, in 1989 dollars. The medical costs are updated from the aforementioned EPA study. A 5% discount rate is used to estimate compensatory education costs in years 2 and 3 of the 3-year program.

IQ effects on earnings and neonatal mortality effects show no threshold, with damages of $1,147 per μg/dL increase per child and neonatal mortality damages of $300 per μg/dL per child, in 1989 dollars. Earnings effects consider both changes in the probability of labor force participation and effects on the wage rate. From Griliches (1977), the wage rate is assumed to fall 0.5% for a one point decrease in IQ. Using ad hoc methods, years of schooling is found to fall by 0.131 years per IQ point, with an independent effect on the wage rate of 0.8%. A one point IQ drop is estimated to reduce the probability of graduating high school by 4.5% (Needleman 1990) with those failing to graduate having participation rates 10.5% lower than those who do (Cropper and Krupnick 1990). To compute earnings effects, real wages are assumed to increase at 1% per year, and lost productivity of nonworkers is taken to be half that of workers. Using Bureau of the Census earnings profiles for 1987, the earnings loss discounted to age three is as previously mentioned.

Increased risks of neonatal mortality are estimated from a study linking a pregnant woman's blood lead level to gestational age and another study linking gestational age to infant mortality. The damage associated with a 1 μg/dL increase in blood lead level is found using a $3 million estimate for a VSL.

RCG/Hagler, Bailly (1991) recently produced estimates of marginal health damages to children from exposure to lead produced by an incremental power plant and other lead sources. The analysis pairs estimates of lead emissions to the ISCLT atmospheric dispersion model (EPA 1986) to estimate worst case and annual incremental maximum airborne lead concentrations, estimates soil deposition for from 1 to 40 years; uses EPA's biokinetic uptake model to estimate the proportion of children with blood lead levels above various thresholds, and uses values for medical costs, compensatory education, and IQ loss to estimate damages.

Medical costs and education costs ($291 annually through age 7, in 1989 dollars) are estimated only for children with blood lead between 10 and 24 μg/dL, in contrast to CDC's much larger estimates, which are only for children with blood lead levels of 25 μg/dL or more and for years 2 and 3 of the child. RCG/Hagler, Bailly's analysis relies on expert judgement to link blood lead levels to IQ loss to dollar losses for children with blood lead between 10 and 24 μg/dL, while CDC uses a wide variety of studies to estimate each step in the chain from IQ loss to earnings loss. While RCG/Hagler, Bailly uses a biokinetic model to track blood lead changes rather than EPA's aggregate model, the former is sensitive to a host of assumptions. The aggregate model is more appropriate for this project. As RCG/Hagler, Bailly includes both compensatory education costs and earnings losses from IQ loss, there is a possibility of double-counting damages.

The worst case analysis yielded damage estimates for a change in lead emissions from a 500 MW coal-fired power plant placed in an area of 682,000 people of $\$2.18 \times 10^{-8}$/kWh.

4.6 ALTRUISTIC MORBIDITY VALUES

Altruistic values for reducing non-fatal risks of poisonings caused by insecticide handling were obtained in a CV study of 785 individuals in Greensboro, North Carolina by Viscusi, Magat, and Forrest (1988). Average values of a statistical case avoided ranged between $2,180 and $3,870 for a scenario involving risks reduced 5/10,000 by a new, safer product compared to a hypothetical product presented. WTP was elicited for an advertising campaign in North Carolina and in the United States that would result in the same risk reductions (2,000 statistical cases in North Carolina) through improved handling of insecticides. When aggregated across the population of North Carolina, the value of a statistical case avoided ($14,300–$24,400) was over five times that obtained through private risk valuations. A similar result held for the U.S. campaign, although respondents were not willing to pay for a campaign that would only benefit residents of another State, for example, Georgia.

Caution in using this study is urged due to (1) the relatively unsophisticated nature of the study's survey design, (2) the possibility that private valuations are spilling over into the altruism responses, and (3) its failure to probe the deeper issue of whether respondents would change their "altruistic" bids upon learning how high their value per case was relative to a value obtained through private risk reductions. The authors urge caution for any use of this study outside of the commodities it examined.

4.7 STUDIES VALUING INJURIES

There are two approaches taken in the literature for estimating the WTP for avoiding non-fatal injuries, where the purview of these studies is injuries on the job resulting in at least one lost work day (WLD). One approach, exemplified by Pindus, Miller and Douglass (1991), may be termed a bottom-up approach, as it seeks to identify the damage associated with an injury on a component by component basis, e.g., medical costs, work loss days, household productivity loss. The second approach is an hedonic wage approach, where variation in injury rates across types of jobs and industry classes and other variables is used to explain variation in wage rates and labor force participation. This is the approach used by most researchers to obtain values of a statistical life; indeed, many of these studies contain a variable for injury rate as well as a variable for accidental death rate.

For the bottom-up approach, Pindus, Miller and Douglass (1991) estimate average costs for specific injuries that are described by the location on the body where they occur and the nature of the injury (e.g. burn). The break costs down into several categories. In addition to (0) quality of life loss, these categories include (1) medical costs, (2) ambulance costs, (3) vocational rehabilitation, (4) wages and fringe benefits forgone, (5) household productivity losses, and finally (6) administrative costs, which we do not consider because these costs would be fully internalized if they exist at all. Quality of life losses are the least certain of these costs. Pain and suffering associated with permanent disabilities are estimated using expert opinion to represent the loss as some fraction ("functional capacity") of the value of a statistical life (VSL). Average productivity losses are then subtracted to avoid double-counting. Productivity losses include the value of lost work days and housework days due to the injury. They assumed that the value of lost household production was 7.51% of the wage rate, which is the employer contribution to Social Security. Medical costs are largely based on the Detailed Claims Information (DCI) database of the National Council on Compensation Insurance (NCII). The majority of the average injury cost was the quality of life losses. They estimated that 71% of the total average cost of a railroad injury was attributed non-monetary quality of life losses.

We found nine studies using the hedonic wage approach to estimate the value of avoiding a statistical injury (VSI). These models include personal and job characteristics and most include corrections for workers compensation benefits (which are tax-exempt) by assuming full earnings replacement from workers compensation and using after-tax wage rates. These researchers generally use ordinary least squares (OLS) and non-linear least squares estimation techniques, assuming that job risks are exogenous. If individuals with higher earnings capacity choose less risky jobs, this assumption will result in a downward bias to the VSI.

The studies as a group indicate that workers do receive higher wages as a result of working in occupations or in industries with higher injury rates. The one study that disaggregated injuries by major and minor injuries (Martinello and Meng 1992) found that wage compensation was insignificant (actually negative and significant) for minor injuries but was quite positive, significant, and large for major injuries, implying the VSIs ranging from $116,000 to $138,000 in 1989 U.S. dollars. For the studies that do not differentiate injury severity, values range from $10,000 to $70,000, although the bulk of the studies are in the $20,000 range.

There are two studies that stand out from the rest: Martinello and Meng (1992) and Moore and Viscusi (1988).[3] The Martinello and Meng study, although simple in its estimation techniques, provides an analysis with by far the most recent database (1986 vs the 1970s, except for the some of the Viscusi papers, which are based on 1982 data, but for the chemical industry only). The model also contains terms for both fatal and non-fatal injuries. The sample is blue-collar workers in logging, mining and manufacturing, a reasonable sample for occupational injury damage estimation in the fuel cycles, providing estimates of both fatal and non-fatal risks in excess of those for a random sample of occupations and industries.[4] Unfortunately, the model does not include worker compensation adjustment, uses pre-tax wage rates, and is estimated using OLS. Given these problems, one would expect that the Martinello and Meng estimates of VSI for aggregate injury rates (not by severity) would be below those of the other studies, which they are. Without controlling for severity,, Martinello and Meng estimate a VSI ranging between $8,000 and $10,000. However, their VSL estimates of from $1.6 to $4.9 million (1989 U.S. dollars) are not out of line with the VSL literature.

Moore and Viscusi (1988)...estimate the discount rate being applied to life cycle decisions...best estimate of VSI is $26,000...

The other, more theoretically sophisticated, study is by Moore and Viscusi (1988). They apply non-linear least squares techniques to estimate the discount rate being applied to life cycle decisions, use the after-tax wage, correct for workers compensation programs, and contain dummies to reflect self-assessed hazard. The drawbacks of this study are that it uses data from 1977 and the sample in relatively small, only 317 non-farm household heads who worked more than 20 hours per week. Their best

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[3]Those studies relying on data from the chemical industry are inappropriate for our purposes because this industry is riskier than average or than the industries of importance to fuel cycles. The remaining studies are less sophisticated and older studies and so will not be discussed either.

[4]Occupational injury rates average 6.3 days/100 days for this sample versus 4.68/100 for a more representative sample.

estimate of VSI is $26,000 (1989 dollars), with a range $17,000 to $34,000 for average baseline risks of about 4.7 lost work days per 100 days. Their VSLs are among the highest in the literature, ranging from $6.8 to $7.8 million.

4.8 HEDONIC PROPERTY VALUE STUDIES

Smith and Huang (1991) performed a meta-analysis on 167 studies of the relationship between property values and air pollution. They conclude that the literature on this topic is out of date in terms of air quality and housing conditions and in reflecting current conceptual and empirical techniques. They find that the mean change in property values with a 1 $\mu g/m^3$ change in second highest daily reading of total suspended particulates (TSP) [the slope of the hedonic price function] ranges between $150 and $250. The geometric mean particulate reading is 121 $\mu g/m^3$ (the daily standard was 150 $\mu g/m^3$). The study provides a number of regression results explaining variation in this slope as a function of baseline air quality (those living in dirtier areas value improvements less), and income (positive effects on the slope). Such results could be used for benefits transfer if double-counting issues could be resolved.

5. SYMPTOMS

5.1 CONTINGENT VALUATION AND AVERTING BEHAVIOR APPROACHES

There are four recent contingent valuation studies that address the value of avoiding a day of symptoms: Loehman et al. (1979), Tolley et al. (1986), Dickie et al. (1986a), and Dickie et al. (1987). The first two and the last study used only the contingent valuation approach, while Dickie et al. (1986a), in addition, used the averting behavior method.

5.2 LOEHMAN ET AL. (1979)

Among other tasks, the Loehman study set out to collect bids through a mail questionnaire from a random sample of 1,800 people (404 responded) for avoiding three types of respiratory symptoms (shortness of breath; coughing and sneezing; and head congestion, including eye and throat irritation) at two severity levels (mild/severe) for 1, 7, and 90 days. Median and mean bids were reported, but the authors believed that their median bids were a better measure of central tendency because of a number of high bids, designated by the authors as protest bids. Bids were found to increase at a decreasing rate with days ill, and increase with income and baseline days of illness. Females turned in larger bids than males.

The reported bids for the 90-day scenario are quite small. In fact, they are much smaller than the medical costs associated with a disease severe enough to cause this frequency of symptoms. For instance, for 90 day/year of shortness of breath (which the authors equate to 90 asthma attacks), the WTP bid is only $263.20 (1989 dollars) or $2.92 per attack-day.

Because ozone reductions are unlikely to result in multiple-day symptom reductions for the average person and because respondents may associate multiple symptoms with certain diseases rather than days of symptoms, the results for multiple-day symptoms are ignored in the benefit analysis.

The single day values are:	Median dollars/day (1979 dollars)	Median dollars/day (1989 dollars)	Mean dollars/day (1989 dollars)
Minor shortness of breath	$ 4.90	$ 8.00	$ 84.00
Severe shortness of breath	10.92	19.00	137.00
Minor coughing/sneezing	2.31	4.00	45.00
Severe coughing/sneezing	6.96	12.00	79.00
Minor head congestion	3.80	6.00	56.00
Severe head congestion	8.17	14.00	92.00

It is difficult to address the reliability of these results because so little is known about the protocols followed by the Loehman team. Nevertheless, several points can be made. First, the survey was a mail survey with only a 22% response rate. Such a low rate may cause severe non-response bias (which would generally tend to inflate the estimate of the average bid for the population as a whole, if non-responders value the symptoms less than responders). Second, the definitions of the symptoms were apparently quite vague (simply "mild" and "severe"). Thus, it is not clear if respondents understood what they were valuing. There were many quite high bids, suggesting a problem with interpretation of the symptom definitions or with other aspects of the survey.

5.3 TOLLEY ET AL. (1986)

During 1984–85, four different CV surveys were administered to a total of 199 randomly chosen heads of households in Chicago and Denver. Two of these dealt with the WTP to avoid "light" symptoms, either for an additional symptom day annually or for an additional 30 days annually. The two other surveys dealt with the WTP to avoid angina. Below, the results from the 1-day light-symptom survey are reviewed.

Forty respondents provided acceptable responses to this survey. An unknown (but small) number of observations were dropped from the sample. These observations were thought to be protest bids.

Respondents were first asked about their general health status and condition over the previous month. They were asked how often they experienced each of seven symptoms (cough, stuffed sinuses, throat congestion, itchy eyes, drowsiness, headache, and nausea) over the past year, the consequences of this distress (e.g., RADs, WLDs), and the medical treatment they received.

The respondents then were asked to think about a situation where last year's health with respect to these seven symptoms would be identical to next year's health except that they would suffer one additional day of a particular symptom or combination of symptoms. Respondents were then read the following description of hypothetical symptoms and asked to rank each in turn as to relative bother.

The symptom descriptions were:

(1) Cough: You will cough about twice an hour in spells that last 10 to 20 seconds. You will feel the cough in your chest, but it is not severe enough to make you red in the face.

(2) Sinus: You will have congestion and pain in your sinuses and forehead all day. You will be bothered by a feeling of stuffiness in your head, accompanied by sinus drainage in your throat. You will need to blow your nose every few minutes. You will have to breathe through your mouth most of the time.

(3) Throat: You will make repeated efforts to clear your throat. The throat clearing is annoying to you and those around you. Your throat will be scratchy. Your voice will be hoarse, and you will have some difficulty speaking.

(4) Eye: Watering and smarting of your eyes forces you to interrupt what you are doing about every 15 min. You rub your eyes and close them. Stinging of your eyes brings tears three times during the day — bad enough to cause you to use a handkerchief or kleenex around your eyes.

(5) Headache: Two rather painful, splitting headaches will strike some time during the day. Each period of headache will last 2 h.

(6) Drowsiness: You will have extreme difficulty staying awake during 6 of the hours when you are normally awake. Sometimes your eyelids will flutter. You will doze off for an instant now and then. The drowsiness will interfere with your social activities and other leisure. You will find the drowsiness dangerous if it comes over you while you are driving or working with tools, appliances, or other machinery.

(7) Nausea: Throughout the day, you will have a lingering urge to vomit, but you will not be able to do so. Stomach distress will be strong. There will be no actual pain.

Respondents were reminded that any payments to avoid these symptoms would come out of their family budget. They were also reminded about actions taken when symptoms are experienced, such as buying and taking medicine, visiting a doctor, and so on. Finally, the interviewer said that the "cure might be worth much more to us than that price of medicine or doctor visit, if we really had to pay for it," reminding them of the difference between the price of mitigation and WTP.

Bids for each symptom were elicited starting from an initial bid and doubling or halving the bid until the respondent no longer wanted to change it. Specifically, the respondent was first asked if experiencing one less day of your least bothersome symptom would be worth $100. Once a bid was determined, it was doubled and the respondent was asked if he would be willing to pay this amount for one less day of the most bothersome symptom. Then the respondent filled in bids for the other symptoms and was given the opportunity to adjust the bids. Finally, the respondent was asked to bid on symptom combinations and on avoidance of a day of symptoms for their household and the entire United States. Next, the respondent was asked to give reasons for the answers and describe how the symptoms differed from what had actually been experienced. Then the respondent was asked about the WTP to avoid all of last year's symptoms.

Median bids for the symptoms of interest (1–5) range from $13–$24, with a median 3-symptom combination bid of $36.40. The 25th percentile bids range from $5.10 to $6 and mean bids range from $30.10 to $47.90. It is clear from these statistics and those in the "5 HIGH," "5 LOW," and "standard deviation" columns, that responses varied enormously from respondent to respondent, but were fairly similar across symptoms. In addition, note that the bids for the symptom combinations are less than the sum of the bids for each symptom separately.

The marginal bids for hypothetical symptoms far exceed the average bid for actual symptoms. This is consistent with the finding of increasing marginal disutility of illness (ε = 0.43 for cough) in a regression analysis of the 1-day survey results, although the 30-day survey did not yield this result. In the "1-day" regression analysis, none of the socioeconomic variables were significantly related to the bids. A respondent's distaste for illness (comfort) was the number one reason given for their bids.

The Tolley study is admirable for its careful and realistic descriptions of hypothetical symptoms. Respondents should have understood the nature and severity of the symptoms they were bidding to avoid (although, as discussed in the following paragraphs, these notions may not have been the ones intended by the researchers). Also, the set of symptoms in the survey closely match those experienced by participants in clinical studies of the regulatory effects of air pollution. Finally, many of the protocols appear to have been quite well thought out to reduce several potential survey biases.

Nevertheless, the study results are likely to overestimate the true value of avoiding a symptom-day for two reasons. First, the protocol is likely to have introduced a starting point bias by initiating the bidding with a $100 price attached to the least bothersome symptom. The bids probably would have been much different if the starting point was lower.

Second, the combination of interviewer remarks on the issue of WTP versus the price of mitigation and the symptom descriptions may have led the respondents to bid as if mitigation were impossible. For instance, the headache description involves two painful headaches of a 2-h duration. The respondent is asked to provide the value of avoiding two 2-h headaches, not two headaches that could have been partially cured by aspirin. The respondents should have been asked to value the entire symptom experience, including the distress before and after mitigation, the cost of mitigation, and the side effects of mitigation. The symptom description could have encouraged the respondent to think about what he would do when confronted with a given symptom and to provide values accordingly.

In addition, the credibility of the results, even interpreted as an upper bound, must be questioned because of possible widespread misinterpretation of the "good" being purchased. It is hard to explain how some respondents provided bids of zero to avoid unmitigated headaches of the severity described. Further, even perfectly mitigated headaches of this severity must be worth something to avoid, for example, the cost of aspirin. These remarks apply with equal force to bids to avoid the other symptoms.

5.4 M. DICKIE, S. GERKING, W. SCHULZE, A. COULSON, D. TASHKIN (1986A)

This study used contingent valuation and revealed preference techniques to estimate alternative values for the WTP to avoid respiratory symptoms. The data collection effort was part of a study of air pollution and respiratory health headed by R. Detels at UCLA. Health, and personal and family characteristics were obtained in a background survey and daily morbidity and avoidance information were collected by telephone over 8 days from July to December 1985 for 229 nonsmoking, employed adults 25–59 years old (primarily males) living in Glendora and Burbank, California. As impaired individuals (those with asthma, bronchitis, emphysema, or other chronic respiratory disease) made up 30% of the sample, all analyses were conducted on normal, impaired, and combined samples. The data base was supplemented by daily 1-h maximum concentrations for O_3, SO_2, and NO_x matched to the daily symptom data, although no analyses were reported involving NO_x. Meteorological variables and some important "lifestyle" variables (diet, exercise, etc.) were not included.

The contingent valuation component involved simply asking respondents for their maximum WTP to avoid the symptom they experienced that day or the day before. No payment vehicle was referenced. A payment card using values from zero to more than $1,000 was used featuring gradually increasing increments between bids. The occurrence of multiple symptoms and the issue of symptom severity were not addressed in the CV question and the respondent was given no preparatory remarks on how to interpret the WTP question. However, respondents were asked if zero bids were protest bids.

As has been typical of CV symptom studies, responses were skewed to higher bids. Mean bids dropped dramatically after each of the tails of the bid distribution were trimmed 2.5%. No median bids were presented. Interestingly, bids to avoid a symptom-day from the "impaired" sample were much larger than from the normal sample, even for nonrespiratory symptoms and for respiratory symptoms experienced by the two groups for about the same average duration and frequency over the 8 days. Table 2 presents the results for mean bids for the normal and impaired samples. A more detailed presentation of CV results is provided with an improved, supplementary study reviewed in the following paragraphs.

The revealed preference component has a number of sub-components: estimation of the WTP for a reduction in ozone concentrations and estimation of the WTP to avoid various types of symptom-days. Both subcomponents rely on a model that generalizes the "standard" non-joint averting behavior model to many symptoms and averting activities and allows averting goods to confer utility directly. With the assumption that one averting good is a necessary input to the avoidance of respiratory symptoms and confers no utility, this model implies that changes in consumer surplus caused by an ozone-induced shift in the Marshallian demand curve for that good can serve as an estimate of the WTP for this improvement [see Bockstael and McConnell (1983) for a model applicable in a more restricted setting].

Table 2. CVM estimates of willingness to pay for avoiding symptoms for 1 day (1989 dollars)

Symptom	Normal Mean Bid	Impaired Mean Bid
Could not breathe deeply	$ 37	$312
Pain on deep inhalation	48	224
Out of breath easily	295	431
Wheezing/whistling breath	14	385
Chest tight	235	228
Cough	161	236
Throat irritation	52	245
Sinus pain	112	275
Headache	145	177

Source: Dickie et al. (1986a, p. 15).

The WTP to avoid a reduction in ozone from 0.12 to 0.10 ppm was estimated from a physician visit demand equation derived from sample data, under the assumption that a physician visit is a necessary avoidance input. Ozone was found to have a positive and statistically significant effect on the daily probability of visiting a doctor for both the impaired and normal samples. The change in consumer surplus associated with a change in the daily maximum ozone concentration was computed assuming that the price at which physician demand falls to zero equals the highest price paid for a physician visit by any respondent in the sample. Because the physician visit demand equations predicted a greater frequency of doctor visits for the normal group (based on the greater observed frequency of this group), this exercise yielded the implausible result that the WTP of the impaired sample was below that of the normal sample. Accordingly, the intercepts of the demand equations were adjusted to more properly reflect the "typical" frequency of doctor visits of the two groups, which was obtained from a background question rather than from the daily questionnaire.

The model also yielded an expression that decomposed the aforementioned measure of WTP into the value of time lost and the value of utility lost. Time losses were estimated for work loss alone. First, logit regression was used to explain whether a respondent worked less than their normal daily hours. Ozone proved to be a significant, negative explanatory variable for the normal, but not the impaired, sample. Daily work losses as a result of reducing ozone from 0.12 to 0.10 ppm were estimated for the normal sample by computing the change in the estimated probability of work loss due to ozone, multiplying by average work loss, and multiplying the product by the average daily wage (before taxes). Subtracting the estimate of time losses from the WTP estimate based on physician visits yielded an estimate of the value of the utility loss from an ozone change. For normals, the latter estimate swamped the estimate of the value of work loss.

The model also yielded an expression for the second subcomponent: the WTP to avoid a symptom-day. This expression equals the price of a good used to avoid a symptom divided by the marginal product of that good in avoiding the symptom (the change in the symptom probability per unit change in the quantity of the good). The derivation of this expression required the assumption that each symptom type is avoided by a unique combination of averting goods. Because short-term avoidance behavior may be determined simultaneously with symptom experience, the two-stage least squares approach (modified to handle limited dependent variables) was used to address the problem of simultaneous equation bias.

Several long-term avoidance strategies, such as the presence of home and car air conditioners (denoted by the variable name ACCAR), air purifiers (APHOME), and gas (vs. electric) stoves (GASCOOK) occasionally were found to exert a negative and significant effect on the probability of experiencing symptoms.[5] These effects were interpreted as the marginal product terms; market data yielded the out-of-pocket annual costs of these long-run avoidance goods. Tables 3 and 4 provide the results of this analysis.

The authors are to be commended on the breadth and originality of their study. In addition, the study addresses an important weakness of earlier valuation studies by attempting to estimate values based on revealed preferences that account for avoidance behavior rather than those based on hypothetical responses.

Nevertheless, the results of this study need considerable refinement before they can be used with confidence in a morbidity benefit analysis. The

[5]Ozone was found to be an insignificant determinant of the probability of experiencing a symptom.

Table 3. Averting behavior and willingness to pay (WTP): normal subsample

Symptom	Averting good	Change in probability of symptom	Expected symptom-days avoided	WTP per symptom-day avoided (1989 dollars)
Could not breathe deeply	---[a]	---[a]	---[a]	---[a]
Pain on deep inhalation	GASCOOK[b]	0.0079	2.88	$33.56
Out of breath easily	---[a]	---[a]	---[a]	---[a]
Wheezing/ whistling Breath	---[a]	---[a]	---[a]	---[a]
Chest tight	ACCAR[d]	0.0116	4.25	41.21
Cough	ACCAR[d]	0.0287	10.47	16.34
	GASCOOK[d]	0.0866	31.63	3.07
Throat irritation	ACCAR[d]	0.0291	10.63	16.48
Sinus pain	ACCAR[d]	0.0300	10.94	16.01
Headache	ACCAR[b]	0.0211	7.69	22.78

[a]No coefficients of averting goods were correctly signed and statistically significant at 10% using a one-tail test in symptom production function.
[b]Denotes coefficient significant at .01 (one-tail) in symptom production function.
[c]Denotes coefficient significant at .05 (one-tail) in symptom production function.
[d]Denotes coefficient significant at .10 (one-tail) in symptom production function.
Source: Dickie et al. (1986a, p. 6).

Table 4. Averting behavior and willingness to pay: impaired subsample

Symptom	Averting good	Change in probability of symptom	Expected symptom-days avoided	WTP per symptom-day avoided (1989 dollars)
Could not breathe deeply	GASCOOK[b]	0.0908	33.14	$ 2.92
Pain on deep inhalation	ACCAR[a]	0.0258	9.41	18.61
Out of breath easily	GASCOOK[a]	0.0954	34.82	2.78
Wheezing/ whistling breath	GASCOOK[b]	0.0781	28.51	3.39
	ACHOME[a]	0.0677	24.70	19.36
Chest tight	ACHOME[a]	0.0476	17.38	27.51
	ACCAR[a]	0.0709	25.88	6.76
	GASCOOK[c]	0.2376	86.71	1.12
Cough	ACCAR[a]	0.0536	19.56	8.95
Throat irritation	ACCAR[b]	0.0685	24.99	7.01
Sinus pain	ACHOME[b]	0.0505	18.45	25.92
Headache	ACHOME[a]	0.0629	22.96	20.82
	APHOME[a]	0.0634	23.41	6.00

[a]Denotes coefficient significant at 0.01 (one-tail) in symptom production function.
[b]Denotes coefficient significant at 0.05 (one-tail) in symptom production function.
[c]Denotes coefficient significant at 0.10 (one-tail) in symptom production function.
Source: Dickie et al. (1986a, p.7).

limitations arise in the theory, data, statistical, and implementation phases of the study.

Theory

The existence of different degrees of severity, the influence of baseline health status on WTP, and the fact that various symptoms usually appear together were not addressed in the theoretical model. These issues are important because Dickie et al. (1986a) found that, irrespective of symptom frequency or duration, bids from impaired respondents exceeded those of normal respondents. This result may be explained by greater perceived (or actual) severity levels among impaired individuals or by a positive relationship between the marginal disutility of a symptom and baseline impairment levels.

The solution to the model implies that when the number of "pure" averting activities is less than the number of symptoms, the model yields an infeasible solution. Given the way in which pure is defined, this condition is nearly certain to hold, rendering the aforementioned solution invalid. The solution to the model also implies that averting activities can completely mitigate the symptoms. This is seen in the expression: $WTP_j = q_i/MP_{ij}$, where q_i is the full price of averting good i and MP_{ij} is the marginal product of the good in reducing the probability of symptom j. If mitigation is partial, then there will be residual pain and suffering that the individual will be willing to pay to avoid.

In using the model to value symptom avoidance, the assumption was made that each symptom is avoided by a unique combination of goods. Inspection of one's own behavior casts serious doubt on this assumption. Finally, the theory does not provide an approach for choosing among unit values for a particular symptom corresponding to the use of different averting goods.

Data

One may question the linkage between purchase of some of the long-term averting goods and their properties in reducing the probability of respiratory symptoms. Taking a trip out of the area (the variable RECTRIP) is one example of a weak link. Another is the ownership of an air conditioner. Such a good may be purchased for many reasons, but primarily for relief from heat. Using the full cost of an air conditioner to value reduced symptoms would grossly overestimate WTP. In addition, those who do not purchase air conditioners are more likely to be income constrained rather than have a lower value for symptom avoidance. The choice over a gas or electric stove is even more constrained; it is nearly inconceivable that health effects would enter into such a choice. Further, this choice variable is tied to NO_2 rather than ozone.

The use of these goods, rather than their ownership, is probably more closely tied to health and air quality. It is unfortunate that data on use were unavailable and the effects of use of other goods on symptom probability were insignificant in the symptom production functions.

The model requires that there be at least one "pure" avoidance good, for example, one that confers no utility directly. Doctor visits are characterized as such a good. While this good probably confers little direct utility, it is not typically an averting good, but rather a mitigating good. In this case, is it appropriate to use this good at all? Even if its use as an averting good is accepted, such visits are not a necessary input to symptom avoidance. Symptom incidences can be avoided without calling, let alone visiting, a doctor. What is more, from the questionnaire, it is not clear that such visits were tied to the experience of a respiratory problem.

Turning to omitted variables, the lack of meteorological variables, particularly temperature and humidity, is a serious omission, as temperature and ozone are likely to be highly correlated. Indeed, such a correlation may account for the positive and significant relationship found between ozone concentrations and time spent outdoors. Also, values for ozone concentrations in December and NO_X in November and December were missing, with mean values used for November and the entire period, respectively. As the analysis uses daily data, missing values may play an important role. Apparently, NO_2 was dropped from the analysis; but no mention was made of this in the text.

Statistics

The use of a one-tail test of significance is inappropriate because reverse causation is a distinct possibility, a priori. For instance, it is possible that those experiencing many symptoms would be more likely to own an air conditioner, and that a regression of "presence of an air conditioner" on symptoms would therefore yield a positive sign, rather than the negative sign required for the marginal product term to be meaningful.

No attempt was made to correct for the possibility that estimation errors for each individual were correlated over their 8 days of observations (or over day t and t-1). A Markov process approach, such as suggested by Korn and Whittemore (1979), could be used to correct for this autocorrelation.

Implementation

Two procedures related to the calculation of the WTP for symptom avoidance and its components are questionable. First, the use of only out-of-pocket expenditures may underestimate the full price of the avoidance good and thus underestimate WTP. Second, the estimate of time loss is based only on work loss and such losses are estimated indirectly. Respondents are not asked if they lost work because of symptoms. In addition, leisure time loss, sleep loss, and productivity loss on the job are ignored in this calculation. If they were considered, the utility loss component would be a much smaller percentage of total WTP.

All of the structural equations assume that the daily ozone concentrations have contemporaneous effects on the probability of experiencing symptoms. For some symptoms, such as eye irritation, this assumption is quite reasonable. However, the probability of experiencing other symptoms, such as throat irritation or sinus pain, on a given day may be affected by a dose integrated over a number of days or by high concentrations a few days before. In principle, these hypotheses are testable with the existing data base.

For several key variables, greater (or equal) reliance is placed on responses obtained from the background questionnaire than from the daily questionnaire, even though the responses on the former survey are subject to much more serious recall problems. For instance, using data obtained from the daily questionnaires, the average frequency of doctor visits of the normal group was found to exceed that of the impaired group. This surprising finding resulted in the estimate of the WTP for ozone reductions in the normal group exceeding that of the impaired group. In contrast, the annual frequency of doctor visits obtained from the background questionnaire showed that the impaired group recalled visiting a doctor much more frequently the previous year than the normal group. When the responses from the background questionnaire were substituted for the daily responses, the anomalous result concerning the WTP was reversed. These new estimates were then offered as the more reliable estimates of WTP. A similar problem is encountered when background information on the usual number of hours spent outdoors is used along with the responses on the time actually spent outdoors (obtained from the daily questionnaire) to estimate outdoor time losses.

The work loss variable is converted into a dummy variable for purposes of estimation rather than kept as a continuous variable. Thus, the probability of any work loss is being estimated rather than the amount of work loss.

Concerning the CV analysis, the approach is quite limited and, therefore, the results should be treated with great caution.

5.5 DICKIE ET AL. (1987)

In this study, an extensive questionnaire was used to probe symptom severity and frequency as well as avoidance and mitigation strategies for 221 residents of Glendora and Burbank, California, during the July–October period of 1986. Bids for avoiding a day of each of the last month's three most bothersome symptoms were taken using the following questions:

> Think about the last time in the past month when you had [SYMPTOM]. Suppose it had been possible to pay a sum of money to have eliminated [SYMPTOM] immediately that one time. What sum of money would you have been willing to pay?
>
> We are not talking about getting rid of [SYMPTOM] forever; we are talking about the amount of money you would have been willing to pay to have eliminated the symptom that one time.
>
> FURTHER EXPLANATION:
>
> Think of this as the price of a special treatment that would prevent the symptom one time. How high a price would you have been willing to pay to eliminate the symptom this one time?

The initial bids were then multiplied by the number of days the symptom was usually experienced per month, summed, and presented to respondents as an initial aggregate monthly bid. Respondents were then permitted to revise this bid. The revised daily bid for each symptom was then calculated by multiplying the initial daily bid for each symptom by the ratio of the revised aggregate monthly bid to the initial aggregate monthly bid. Variations in bids were then explained with a regression analysis.

Table 5 presents the mean and median bids before and after revision. Dramatic reductions in mean initial bids and substantial reductions in median initial bids for most symptoms were observed when respondents were given the opportunity to revise bids. By any standard, the revised bids are far below those obtained by other studies. In addition, and just as important in light of the questionable bid revision procedure (see the following paragraphs), the median initial bids are lower than median bids obtained by other studies. According to the authors, the results of the regression analysis were "not particularly strong" and varied substantially from symptom to symptom.

This study is a vast improvement over the earlier Dickie et al., CV study and serves to highlight the sensitivity of respondents to the form of the WTP questions and the importance of making explicit the consequences of a respondent's bids. It is also a marked improvement over the Loehman and Tolley studies because of the treatment of the symptom severity issue. In the Loehman study, respondents were asked to bid on the avoidance of "mild" and "severe" symptoms, while in the Tolley study, symptom severity was defined in

Table 5. Initial and revised CV bids
(Bids converted from 1986 dollars to 1989 dollars)

Symptom		Initial	N	Revised[a]	N
Not breathe deeply	Mean	$1,290.00	32	$3.46	17
	Median	1.13	32	0	17
Pain on deep inhalation	Mean	1,079.49	16	6.75	8
	Median	3.96	16	2.35	8
Short breath	Mean	8.92	16	0.68	8
	Median	0	16	0	8
Wheezing	Mean	65.62	11	3.10	6
	Median	2.26	11	0	6
Chest tight	Mean	920.63	25	5.70	11
	Median	5.66	25	1.19	11
Cough	Mean	401.76	29	1.57	25
	Median	1.13	29	0	25
Throat irritation	Mean	16.97	26	2.75	17
	Median	3.39	26	0	17
Sinus pain/ discomfort	Mean	270.97	44	2.13	20
	Median	3.96	44	1.61	20
Headache	Mean	201.83	61	4.92	16
	Median	1.13	61	2.66	16

Source: Dickie et al. (1987)
[a]After consistency check exclusions.

detail by the researchers. Both approaches are problematic because they offer no guidance on the severity of symptoms actually experienced by the respondents. Therefore, a match between the severity of the symptom avoided and the value attached to avoiding that symptom may be lacking. In contrast, the Dickie et al., study asks respondents to value avoiding the severity of symptoms actually experienced. Thus, the linkage between observed severity and value is direct.

Nevertheless, there are several problems with this study that argue for caution in interpreting the following results:

1) Income constraint: Dickie neither implicitly nor explicitly invoked an income constraint (i.e., implied monthly aggregate expenditures) until the respondents were provided with the opportunity to revise their monthly bids.

2) Bid revision procedure: The large reductions in bids were, in part, dependent on the questionable bid revision procedure, where marginal bids were assumed to be constant (equal to average bids) over symptom frequency. It is unclear that respondents would accept the resulting value as their daily bid. More importantly, if the Loehman finding of sharply declining marginal bids with days of symptoms is accepted, then the revision procedure would dramatically underestimate daily bids.

3) Averting behavior: Averting (actually, mitigating) behavior was examined in the study but was not invoked when respondents were asked their WTP. Doubtlessly, some people did not link their earlier responses to mitigation questions to their WTP answers. However, such linkage would further reduce the already low average bids.

4) Outliers: Examination of numbers of people with inconsistent bids reveals that a large number of people revised their monthly bids from something to zero. There is no count of the number of people with this inconsistency. However, according to the authors' consistency checks, if observations were dropped for respondents who had severe symptoms before mitigation but whose revised bids were less than \$1 or who mitigated but bid less than a \$1, half the observations and over half of the revised bids of zero would be eliminated. Clearly, many people were confused about the meaning of the WTP question as reflected in both their original and revised bids.

5) Survey approach: Dickie used a telephone survey. For the elicitation of WTP for improved health, a personal interview approach is far superior (Mitchell and Carson, forthcoming).

6) Risk vs. certainty: Dickie asked respondents what their WTP would have been to avoid the last symptom-day experienced. A better approach is to ask for the WTP to reduce the probability of a future symptom day. Not only does the latter approach permit attitudes towards risk to be included in the response, but the response would be in terms of the expected level of severity (the appropriate measure) not the level of severity most recently experienced.

In summary, the problems with this study argue against using the revised bids in a benefits analysis and serve to bias upwards the initial bids. The median initial bids are probably the most credible set of statistics from the study. The reasons for the relatively low median initial bids are unclear, although theory and some of the regression results identify consideration of avoidance behavior as a likely factor.

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PAPER NO. 12*

THE BENEFITS OF VISIBILITY

1. INTRODUCTION

The benefits of visibility improvement (or the damages with additional degradation) refer to increases (or decreases) in utility obtained in three different dimensions. The first of these is associated with the nature of the visibility change. Visual range may be improved so that features of an area become more distinct or the sky becomes clearer. Alternatively, normal features of an area may be marred, say by the site of a power plant or its plume (called plume blight). The second dimension is the location of the change: in an urban area, in a rural setting, or in a recreational area or area of particular beauty, such as the Grand Canyon. The third dimension is the type of value: use or non-use. Thus, a person who visits the Grand Canyon (or may visit it in the future) may hold use values for improving his view of the Canyon or its surroundings and may also hold non-use values for improved visibility (whether for altruistic or other reasons) irrespective of present or planned visits. In all, therefore, there are 12 possible combinations of the elements in these three dimension, each of which is logically distinct from the others and which demands attention in the literature to derive willingness to pay (WTP).

Indeed, some of the literature suggests that an eastern-western U.S. dimension is also relevant (Oates and Cropper 1990). We see this dimension as mainly a proxy for visual range, with western visibility baselines so much greater than those in the east. To the extent that baseline visibility affects WTP for marginal changes in visibility, and there is every indication that it does, then this dimension is also needed, making for 24 possible combinations.

Unlike the other papers of Part IV, this paper contains a short review of the science of visibility measurement and the relationship between visibility and pollution. This is followed by a review of the literature on visibility valuation.

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*Based largely on a paper by Alan Krupnick and Diane DeWitt.

2. APPROACHES TO MEASURING VISIBILITY CHANGE

Both SO_2 and NO_2 emissions from power plants are associated with changes in visual range through their transformation products — sulfates and nitrates. NO_2 also has a direct effect through absorbing light and giving the atmosphere a reddish-brown tint. Other pollutants, such as large particulates and even water vapor, are associated with plume blight from power plants.

Visibility monitoring is not an exact science because indexes used for characterizing visibility rely on various assumptions and visibility is not a well defined parameter. Three distinct indexes are used to monitor visibility: aerosol indexes, optical indexes, and scene indexes. Broadly speaking, aerosol indexes involve the chemical and physical makeup of the particles in the atmosphere (i.e., the particle composition, size distributions, and mass concentration); optical indexes describe the function of light in the atmosphere (i.e., scattering coefficient, absorption coefficient, extinction coefficient); and scenic indexes characterize the scene viewed by the observer (i.e., observed visual range, contrast, color, detail).

Three distinct indexes are used to monitor visibility: aerosol indexes, optical indexes, and scene indexes.

There is a relationship between visual range and the light extinction coefficient, by which scenarios in studies using one type of index can be expressed in terms of those using the other index. This relationship is given by the Koshmeider equation:

Visual Range = 23.6/Light Extinction Coefficient

It should be noted that the Koschmeider coefficient (the aforementioned 23.6) varies depending on the threshold contrast used, where this term is the difference between the brightness of the sky and the brightness of the object, reduced to the degree to where the observer can just see the object. The accepted range of threshold contrast lies between 2–10%. The constant above is computed for a threshold contrast of 5%.

The range of accepted threshold contrasts implies that visual range changes of less than 5% will not be perceptible...

The range of accepted threshold contrasts implies that visual range changes of less than 5% will not be perceptible, while some people will not notice changes in visual range below 15%. As the WTP for unperceptible changes is likely to be zero [although some researchers, such as Chestnut and Rowe (1985), seem to disagree], the policy significance of this choice is great.

3. REVIEW OF THE LITERATURE

Several studies have summarized the visibility valuation literature. Freeman (1979) has a good summary of the early literature. Chestnut and Rowe (1988) summarize and contrast four visibility studies that also valued health benefits from reduced particulates, finding that for three of the studies the former were on average 15% of the latter, where health benefits include mortality effects. Most recently, the National Acid Precipitation Assessment Program (NAPAP) [section by Rowe] summarizes the visibility literature.

...contingent valuation (CV) approach is the most promising for eliciting WTP.

From these reviews, there is agreement that the contingent valuation (CV) approach is the most promising for eliciting WTP. But two major problems need to be resolved. One of the most difficult problems with application of the CV method to visibility is in the ability of researchers to present changes in visibility conditions to the respondents in a realistic way. Most of the CV studies examined rely on photographs to illustrate the effect changes in visibility have on the scene being observed. If visual range is measured when color photographs are taken, one can establish a data base that shows the difference between the measured values and the appearance of the scene. However, gathering these data can be expensive.

Most importantly, it is difficult to link the concentration of pollutant with the impact that is being valued. The most recent innovation has been to rely on the use of image processing techniques in conjunction with optical indices to simulate the effect being valued. Using atmospheric optical models, a computer can add a determined level of apparent haze to a photograph that was taken on a clear day. Therefore, in the future, visibility studies may use image processing on the photographs; they may then have more validity than the studies that rely exclusively on subjective monitoring.

A major innovation in CV survey techniques may be seen in the Decision Focus study. Here, rather than provide respondents with scenarios asserting annual average changes in visual range, respondents were asked to respond to scenarios where visibility would change only during specified seasons of the year and only for a limited number of days. Such scenarios were consistent with projected changes in visibility as part of an action to reduce SO_2 emissions at a power plant in the southwest. Compared to responses that provide scenarios in which visibility changes are not distinguished by time of year and frequency, WTP estimates were much smaller (although one should not put too much stock in these empirical results, as the Decision Focus study was only a pilot study involving 200 people).

A major innovation in CV survey...respondents were asked to respond to scenarios where visibility would change only during specified seasons of the year and only for a limited number of days.

Another major issue with contingent valuation studies (or any other approach) is jointness — the possibility that respondents are using visibility as a proxy or as an indicator for a complex of effects, including health effects, damages to plants and animals, and in the case of the southwest, damage to rock formations. This phenomenon would lead to overestimating benefits associated with emissions reductions if health and other non-visibility damage estimates were added to the damage estimates from visibility studies. According to Oates and Cropper (1990), the best way to disentangle WTP is to ask respondents what they assume the health effects to be and to control for these effects, say by valuing them and subtracting them from total WTP. Simply asking people to ignore the health component and give a value only for visibility improvements is seen as less satisfactory.

A third issue, which applies to all of the valuation literature but which is particularly apparent in the visibility literature is the "embedding" problem, for example, where WTP is sensitive to whether a good is valued by itself or as part of other goods. Debates are ongoing on guidelines for addressing this problem. A reasonable resolution is to develop CV scenarios that reflect the actual choice situation created by the policy or activity, and elicit values for this bundle. One manifestation of this problem is estimating WTP for improvements in visibility at one site and simply adding it to such estimates for other sites. One might expect that the WTP for improvements over many sites should be less than the sum of WTP over

...the "embedding" problem...WTP is sensitive to whether a good is valued by itself or as part of other goods.

each site. Without accounting for embedding, WTP may be overestimated. Findings drawn from Schulze et al. (1983) and Tolley et al. (1986) show that when asked to value visibility improvements in the Grand Canyon alone, WTP was $95 per year per person, when this improvement was joined with improvements in visibility in Chicago and on the east coast, the WTP for Grand Canyon improvements fell to $22.

The recent very detailed literature review of visibility benefits studies in NAPAP is the best place to start the literature review, although the studies identified are only those that are applicable to changes in visibility associated with acid rain control. Included in the studies are those for the eastern U.S. and selected urban areas of California. Table 1 shows a summary of all the urban visibility studies NAPAP reviewed. Below we will briefly summarize the problems and issues NAPAP addressed with each study it critiqued.

The Six Cities study undertaken by Tolley et al. (1986) for the Environmental Protection Agency (EPA) estimated values for changes in visibility in the eastern U.S. Tolley et al. are the only researchers to date that have attempted to measure visibility values for multiple cities in the eastern U.S. Most completed CV visibility studies have been for western recreational areas.

NAPAP found several major problems with the study that add to the uncertainty in the results. The three most prominent problems included the use of one starting point ($13) in the CV survey, the problem with linking visibility changes with perceived health effects, and the absence of a parallel between the photographs chosen and the WTP questions.

Tolley et al. argue that changing the starting bid does not change the responses, but this finding runs counter to results of other CV studies (i.e., Mitchell and Carson 1989). Additionally, although the respondents were told that the questions concerned aesthetic effects of pollutants, NAPAP concluded that many previous reviewers were not convinced that relying on this method ensured that the responses only included aesthetic valuations. Finally, the photographs used in the survey did not parallel the questions asked. For example, the photographs showing local visual range were all taken in Chicago. However, the respondents from cities outside Chicago are obviously not as familiar with the scenes. In addition, the changes in visual range the respondents were told to consider did not match the photographs. In different cities the photos showed higher or lower visual range levels than what was to be hypothesized in the WTP questions. For these reasons the Tolley et al. (1986) effort for EPA is seriously flawed. Nevertheless, we are limited in comparing the Tolley results with other visibility studies for eastern U.S. cities, since Tolley has been nearly the only researcher to work in this geographic region. (EPA is currently funding a new eastern visibility benefits study.)

Table 1. Summary of NAPAP urban contingent valuation visibility value results

Study	City	Total Mean Annual WTP/Person (1990$)	Mean Annual WTP/Person/ VR Mile (1990$)	Starting Average Visual Range (VR) (miles)	Change in Average Visual Range (VR) (miles)
Tolley et al. (1986)	Chicago (n=221)	$124	$24.89	9.0	-5.0
		120	13.32	9.0	9.0
		149	7.09	9.0	21.0
	Atlanta (n=90)	101	20.23	12.0	-5.0
		97	9.69	12.0	10.0
		145	7.25	12.0	20.0
	Boston (n=114)	75	14.96	18.0	-5.0
		72	7.18	18.0	10.0
		89	4.43	18.0	20.0
	Mobile (n=90)	81	16.11	10.0	-5.0
		87	8.70	10.0	10.0
		102	5.10	10.0	20.0
	Washington, DC (n=86)	120		15.0	-5.0
		123	12.33	15.0	10.0
		156	7.82	15.0	20.0
	Cincinnati (n=94)	24	4.89	9.0	-5.0
		24	2.44	9.0	10.0
		27	1.35	9.0	20.0
	Miami (n=47)	42	8.32	13.0	-5.0
		37	3.70	13.0	10.0
		44	2.20	13.0	20.0
Rae (1984)	Cincinnati (n=314)	74	14.73	11.4	5.0
Brookshire et al. (1979)	Los Angeles				
	(n=56)	45	4.50	2.0	10.0
	(n=56)	114	4.39	2.0	26.0
	(n=186)	63	3.96	12.0	16.0
Loehman et al. (1981)	San Francisco				
	(N=56)	74	32.19	18.6	-2.3
	(N=228)	44	19.08	16.3	2.3

Note: *-All numbers are from the actual figures reported in each of the studies. This column is not estimated.

If any study could be compared to the Tolley et al. (1986) eastern cities study, it is the Rae (1984) benefits study of visibility for Cincinnati. Of the three major flaws in the Tolley et al. study discussed above, only one is present in the Rae (1984) study — it is likely that respondents did not separate the aesthetic and health effects associated with air pollution.

NAPAP also reviewed three visibility studies (two CV studies and one hedonic property study) conducted in California where the change in visual range levels are comparable to those in eastern cities. NAPAP concludes that the biggest contribution the two California CV studies make is in the attempt to separate out the perceived health effects associated with change in visual air quality. The Brookshire et al. (1979) study in Los Angeles asked respondents for WTP for visibility aesthetics only, for visibility plus acute health effects and for visibility, acute health, and chronic health. However, depending on the order of the questions, the aesthetic visibility valuation ranged from 10-70% of the total WTP of all bids. Similarly, Loehman et al. (1981) conducted a CV study for the San Francisco area by presenting a range of visibility conditions in terms of numbers of non-polluted, moderate and poor days. Respondents were asked their WTP for a combination of health/visibility conditions.

Although NAPAP credits the two California CV studies for making strides in attempting to separate health and aesthetic benefits of different visibility levels, the studies are very dated and the CV designs are not as good as more recent studies. NAPAP also reviewed a hedonic property value study for the Los Angeles and San Francisco areas conducted by Trijonis et al. (1984). The study's biggest drawback is the strong correlation between health and aesthetic effects; therefore, separate values cannot be obtained.

In an attempt to more accurately compare results of the studies, NAPAP assumed a functional form to put the mean WTP results from the studies into a common unit. We ignore this approach because it ignores the confidence intervals around the mean estimates and estimates some regressions based on three observations.

The key results from Table 1 are that there appears to be diminishing marginal utility from increased visual range, which is consistent with economic theory and that marginal WTP to prevent deterioration of visual range exceeds marginal WTP for an improvement. NAPAP also concludes that there is a large difference in the eastern and western results and that this is due to CV design issues and location differences rather than the separation of visibility from health values. As visibility conditions in Los Angeles and San Francisco are so different and the values are so different, it seems hard to conclude anything from this comparison.

The results for Cincinnati are perhaps the closest to our Southeast Reference environment. Here, the WTP to avoid a 5 mile reduction in visibility is about $5/person/mile, with an improvement of 10 miles valued at $2.50 per mile. Rae, for Cincinnati, finds a much higher WTP — $15 per person per mile for a 5 mile improvement. WTP for Cincinnati is much lower than that for the other eastern cities surveyed, however.

Because NAPAP's focus in reviewing the visibility literature was linked to acid rain control, the treatment of western recreational studies is very sparse. Essentially NAPAP discusses these studies in general, but does not critique them in the manner above. Table 2 outlines the use values for recreational sites in the studies NAPAP reviewed. Again, they estimated a consensus function, listed at the bottom of the table, to determine WTP per visitor per day. The results of on-site use values differ remarkably because of differences in sites examined and the use of different elicitation methods (i.e., iterative bidding, contingent ranking, etc.). Of most interest to the Southeast Reference environment is the Rae study of Great Smoky Mountain National Park. The other studies apply generally to the southwestern site. Use values at the Smokies were found to increase by only $3 to $6 per visitor per day for an enormous increase in visibility (from 6-12 miles to 60 miles). For the western sites, use values are generally in the $1-$8 range per person per day for improvements, with ranges of $6 to $12 for substantial degradation in visibility (at the Navajo site).

General problems common to all of the studies include the application of use values to number of user days (the number of visitor days can be linked to visibility levels), and the possibility that different starting bids may signal different incremental increases.

Below, additional literature is reviewed. Of note are studies of non-use values associated with reductions in southwestern visibility (Decision Focus), and the regional effects of acid rain (EPRI, using Tolley et al.).

Tables 3 and 4 summarize the key aspects of nine studies valuing visibility. We chose major studies, often-quoted studies, and recent studies; our list is not inclusive, but is reasonably representative, except that we include only one property value study. The key general features of these studies are author/year; city/region; measure of value (use or non-use); approach (CV, property value); payment vehicle, if CV; and sample size and composition. Key specific features include: baseline visibility level; scenario; the commodity being valued (e.g., visibility as measured by a photograph); presence of a linkage to pollution; WTP result (in 1989 $s); and comments.

Table 2. NAPAP user values for visibility protection at recreation sites

Study	Site	Sample Size	Initial Visibility (in miles)	Revised Visibility (in miles)	1990$ WTP Per Visitor Party Day*	Method Used (a)
Rowe et al. (1980)	Navajo	26	75	50	$7.71	IB
	Navajo	26	75	25	$12.31	IB
	Navajo	26	50	25	$5.73	IB
Schulze et al. (1983)	GCNP	166	75	95	$2.73	CL
	GCNP	166	75	125	$4.49	CL
	GCNP	166	75	175	$6.33	CL
	GCNP	166	75	240	$8.36	CL
MacFarland et al. (1983)	GCNP	1000	70	100	$1.55	CL
	GCNP	1000	70	130	$2.59	CL
	GCNP	1000	70	165	$3.39	CL
	GCNP	1000	70	215	$4.45	CL
	MVNP	800	70	100	$1.37	CL
	MVNP	800	70	130	$1.86	CL
	MVNP	800	70	165	$2.82	CL
	MVNP	800	70	215	$4.12	CL
Rae (1983)	MVNP	196	74-95(b)	160	$4.88	CR
	MVNP	193	95	160	$13.13	CL
	GSMNP	202	6-12(b)	60	$6.26	CR
	GSMNP	202	12	60	$4.15	CR
	GSMNP	201	12	60	$3.25	CL

Table 2. (continued)

Schulze et al. do not report visual range estimates for their photographs, but Chestnut and Rowe calculated approximate visual ranges using historic data on visual range levels at the Grand Canyon.

Note:

(a) All studies used entrance fees as payment vehicles.
(b) Two scenarios were merged.
IB = Iterative bidding
CL = Check list of value ranges to select from
CR = Contingent ranking
GCNP = Grand Canyon National Park
MVNP = Mesa Verde National Park
GSMNP = Great Smoky Mountain National Park
Navajo = Navajo Reservoir, N.M.

*The change in visual range and the mean of the CV responses was used to estimate the following equation.

$$\text{WTP/Visitor party/Day} = \underset{(6.30)}{\$6.02 * \ln(V2/V1)} - \underset{(-2.54)}{\$3.57 * E * \ln(V2/V1)}$$

where: V1 = initial visual range
V2 = new visual range
E = 1 if eastern study, 0 otherwise

With respect to the 12 combinations of attributes that define the commodity "visibility," the studies primarily address use and non-use values for changes in visual range in special areas. Some studies also examine plume blight and the physical intrusion of a power plant. Thus, visibility effects for the Southwest Reference site are adequately represented in the literature. One particularly glaring omission is the recent Rowe and Chestnut study to support the Best Available Retrofit Technologies (BART) action associated with the Navajo Power Plant in the Southwest. This study is addressed in the Decision Focus study (1990).

Only one study addresses WTP for improvements in visual range in eastern cities. Visibility improvements in Denver (related to CO and the "brown cloud", which is not caused by power plant emissions) are represented in two studies. There are no studies of visibility changes exclusively in rural areas, nor of plumes or physical intrusion in urban areas. The Electric Power Research Institute (EPRI) study includes visibility damages throughout the east, however. The Southeast Reference environment is not represented by the literature.

The visibility benefit studies we reviewed primarily use scenic indexes (through use of photographs), where visual range (the distance an object can be viewed against a background sky with uniform haze) is the key parameter varied in the valuation scenario. The hedonic property value study uses a more objective measure of visibility, the extinction coefficient, the amount of light scattered and absorbed between the target and observer. None of the studies reviewed used an aerosol index to measure visibility in part due to the lack of accepted models, and the problem of large uncertainty of the results. Several studies analyzed used a threshold contrast of 2%. Most of the Koschmeider constants in the visibility benefits studies examined ranged from 18.7 to 24.3.

Other conclusions from this literature are: (1) the payment vehicle used in a CV study can influence the results; the use of entrance fees vs. electric utility bills shows that the latter tends to result in lower WTP; (2) the size of the photographs used in the study may have some impact on WTP (i.e., the larger the photos the more distinct the atmospheric pollution is, hence the higher the WTP); and (3) very few studies mention how long the respondent would be paying for visibility improvements (e.g., over the duration of the power plant vs. annually for several years).

Table 3. Key general features of air visibility studies

Author(s) & Year	City/ Region	Measure	Method	Payment Vehicle	Sample Size/Drawn From
Schulze et al. (1983)	Grand Canyon & Southwest Parklands	Use & preservation value (use + non-use)	CV - iterative bidding game	Increased park fees/electric bill addition	600 +/Chicago, Denver, Los Angeles, Albuquerque (a)
Rowe, D'Arge, Brookshire (1980)	Four corners Southwest U.S.	Use value	CV - iterative bidding game	Electric bill addition & payroll deduction	Less than 100/market unknown (a)
Tolley, Frankel & Kelley (1988)	Major Eastern U.S. cities	Use & non-use value	CV - iterative bidding game	Abstract	800/from Atlanta, Boston, Cincinnati, Miami, Mobile, Washington, D. C. (a)
Brookshire, Ives & Schulze (1976)	Lake Powell region in Colorado	Use value	CV - iterative bidding game	Park fees	104/from locals, motel visitors, camp visitors, remote campers (a)
Trijonis et al. (1985)	Los Angeles & San Francisco areas	Use value	Property value (c)	Not applicable	Housing data for 1978-79: 4765 houses in Los Angeles, 3106 houses in San Francisco
Decision Focus Inc. (1990)	Grand Canyon	Use & non-use value	CV - iterative bidding game	Electric bill addition	202/St. Louis & San Diego; (a), (d)
EPRI (1990)	Eastern U.S. (associated with acid-rain)	Use & non-use value	Did not conduct a CV experiment. Instead the authors use results from the Tolley et al. study to determine WTP for two SO_2 reduction scenarios using the STATMOD model.		
Chestnut & Rowe (1988)	Denver metro area	Use value	Relied on previous CV studies to estimate function used to bound values for visual range estimates for Denver.		
McClelland et al. (1990)	Denver metro area	Use value	Use seven survey versions to test potential CV problems.		1400 (200 for each of 7 surveys)/residents of Denver metro area (b)
Bonneville Power Admin. (1987)	Pacific Northwest	Non-use value	Survey 26 visibility studies to determine appropriate values to impart on visibility impairment from a generic coal fired plant in the Pacific Northwest (e)		
Chestnut & Rowe (1990)	National parks in California, Southwest, & Southeast	Use & non-use value	CV	Payment card approach ($0 to more than $750) non benchmarks	1647/residents of Arizona, California, Virginia, Missouri, and New York (b)

Table 3. Key general features of air visibility studies

Author(s) & Year	City/ Region	Measure	Method	Payment Vehicle	Sample Size/Drawn From
Chestnut & Rowe (1986)	Four air basins in California	Use value	Rely on three previous urban visibility studies to estimate benefits for two different air pollution control strategies.		

Comments
(a) This study relies on personal interviews with respondents.
(b) This study uses a mail survey to solicit WTP from respondents.
(c) This study is the "best" property value study on visibility benefits. Most importantly, the authors use visual range measurements from airport data as opposed to ambient air pollution as previous hedonic property studies have done.
(d) Study commissioned to examine the visibility benefits associated with a BART action at Navajo Power Plant.
(e) All of these 26 visibility studies were not included in this table due to the fact that they were dated.

Table 4. Values of visibility studies

Author	Baseline Visibility	Scenario	Visibility Measure	Link to Pollutants	Visibility Valuation ($/Person/Mile/Yr except where noted) Adjusted to 1990$	Comments
Schulze et al. (1983)	No visual range reported in study, but Chestnut & Rowe (1990) report baseline visual range of 124 miles (a)	(1) Prevent visible plume in Grand Canyon (2) Change in visual range of -28 miles (see footnote (1) to explain) (3) Prevent regional change in visual range of -28 miles (4) Specified conditions in Grand Canyon (b)	8 by 10 photographs	Summer 1979 SO_2 emissions, 692-1034 tons per day, new plants add 1398-1587 tons SO_2/day in 1990	Use value ($/day) (1) $1.83-$2.77 (3) $1.82-$2.84 (4) $3.10-$5.07 (see footnote (b)) Preservation value ($/Person/Year) (1) $19.66-$29.42. For (2), (3) units are ($/person/mile/yr) (2) $.92-$1.27 (3) $.71-$1.11 Based on function (1) $15.11/person (2) $.87/person/mile (3) $1.54/person/mile (c)	Study links SO_2 levels in 1990 to 1979 "below average" visibility; range of values covers the mean bids of four cities
Rowe et al. (1980)	Hypothetical visual range of 75 miles	Change in visual range from baseline visibility: (1) -25 miles (2) -50 miles (3) -50 miles plus visible power plant	Photographs (size not specified)	No	Use value: <u>Residents:</u> ($/person/month) (1) $3.97 (2) $5.46 (3) $5.71 <u>Non-residents:</u> ($/person/day) (1) $2.50 (2) $3.39 (3) $3.81	
Tolley et al. (1988)	Baseline visual range of 14 miles	Change in visual range: (local) (1) -5 miles (2) 10 miles (3) 20 miles (4) 10 miles Eastern U.S. (5) 10 miles all U.S.	3 1/2 by 5 photographs	No	Individual with mean characteristics, would pay $18.59/yr/mile change in visual range. (This value includes the average baseline)	(c), (d), (e)

Table 4. Values of visibility studies

Author	Baseline Visibility	Scenario	Visibility Measure	Link to Pollutants	Visibility Valuation ($/Person/Mile/Yr except where noted) Adjusted to 1990$	Comments
Brookshire et al. (1976)	Hypothetical visual impact of construction site of a power plant	Change from the baseline: (1) visible power plant; (2) visible plant w/smoke	Photographs (size not specified)	No	Use values: ($/day) (1) $2.11-$5.13 (2) $4.25-$8.21	Range of values is the range of means of groups surveyed; (f)
Trijonis et al. (1985)	Visual range: <u>Los Angeles</u> 9.49 miles <u>San Francisco</u> 16.8 miles (g)	Hypothetical visual range change: <u>Los Angeles</u> .86 miles <u>San Francisco</u> 1.53 miles	Hedonic price of visibility	No	<u>Los Angeles:</u> $1385/person/visual range mile <u>San Francisco:</u> $963/person/visual range mile (h)	Results combine aesthetic and health valuation
Decision Focus, Inc. (1990)	Summer - (1) 127 miles (2) 69 miles (3) 30 miles Winter - (1) 221 miles (2) 61 miles (3) 14 miles (j)	Summer - (1) 202 miles (2) 151 miles (3) 106 miles Winter - (1) 270 miles (2) 127 miles (3) 106 miles	8 by 12 photographs	Yes, future BART Action at Navajo limiting SO_2 emissions by 70-90%	Value: (1) summer & winter (2) summer only (3) winter only (4) 20 winter weather days (5) 10 winter weather days; use & non-use: mean values in ($/person/year) St. Louis - (1) $32.97 (2) $20.68 (3) $8.71 (4) $2.06 (5) $2.05 San Diego - (1) $22.98 (2) $11.16 (3) $4.16 (4) $2.66 (5) $2.48	Connection of valuing over the lifetime of Navajo (20 yrs) (i)
EPRI (1990)	No visual range reported	Over the 60 year SO_2 reduction plan, the two strategies lead to different visual range levels each year	N/A	Yes, reduce 1980 SO_2 emissions by 10 million tons: (1) by 2000 or (2) by 2025	For 5-20% change in visual range, under the two scenarios, estimate is $1.10/person/yr over a 60 year span of SO_2 reduction	Estimate for summer average visibility; (k) (l)
Chestnut & Rowe (1988)	Average visual range of 22.4 miles over 10 monitors in the Denver-Metro area (j)	Average visual range increase of 4.7 miles to meet federal PM levels for 1984-86	N/A	Yes, average of 1984-86 particulate matter levels	Value of $4.16/person/year/visual range mile (m)	Rely on previous dated CV studies

Table 4. Values of visibility studies

Author	Baseline Visibility	Scenario	Visibility Measure	Link to Pollutants	Visibility Valuation ($/Person/Mile/Yr except where noted) Adjusted to 1990$	Comments
Bonneville Power Admin. (1987)	No visual range reported	Effect of coal-fired plant within 109 miles of plant	N/A	Emissions from generic coal-fired power plant in the Pacific NW emits 28.5 tons of SO_2, 1.4 tons TSP, 13.1 tons NO_x, 1.3 tons CO per year	Employ value of $19/mile/yr/person (q)	Only one of the 26 studies BPA reviewed was in the area of interest, (r)
Chestnut & Rowe (1990)	Visual range Yosemite: (1) 56 miles Grand Canyon (1) 96 miles Shenandoah (1) 16 miles	Change from baseline visual range: Yosemite: (1) 22 miles (2) 37 miles (3) -28 miles Grand Canyon: (1) 28 miles (2) 59 miles (3) -25 miles Shenandoah: (1) 15 miles (2) 31 miles (3) -10 miles	3 by 5 inch photographs (s)	No	Mean values: Existence values: ($/person/yr/mile) Yosemite: (1) $ .56 (2) $ .41 (3) $ .50 Grand Canyon: (1) $ .40 (2) $ .25 (3) $ .51 Shenandoah: (1) $ .72 (2) $ .49 (3) $1.36 Use + option values: ($/person/yr/mile) Yosemite: (1) $.24 (2) $.17 (3) $.22 Grand Canyon: (1) $.18 (2) $.11 (3) $.23 Shenandoah: (1) $.32 (2) $.22 (3) $.61 (t)	(u)

Table 4. Values of visibility studies

Author	Baseline Visibility	Scenario	Visibility Measure	Link to Pollutants	Visibility Valuation ($/Person/Mile/Yr except where noted) Adjusted to 1990$	Comments
Chestnut & Rowe (1986)	Visual range: San Diego: (1) 5.35 miles (2) 3.85 miles San Francisco: (1) 9.79 miles (2) 8.82 miles San Joaquin: (1) 10.67 miles South Coast: (1) 4.42 miles (2) 5.58 miles	Change in visual range: San Diego: (1) 1.07 miles (2) .78 miles San Francisco: (1) .8 miles (2) .72 miles San Joaquin: (1) .10 miles South Coast: (1) .55 miles (2) .50 miles	N/A	Emissions of TSP, SO_4, and nitrate obtained from CARB for 1979; emissions for 1987 based on state and local implementation plans (v)	Urban use value: ($/person/yr/mile) San Diego: (1) $37.45 (2) $27.30 San Francisco: (1) $28.00 (2) $25.20 San Joaquin: (1) $ 3.50 South Coast: (1) $19.25 (2) $17.50	(w)

(a) Although the photos depict different visibility conditions, no quantifiable measure, like visual range, is equated with the photos. However, Rowe et al. (1990) report photos representing "average" (current) visibility and "below average" visibility represent approximate visual ranges of 200 km (124 miles) and 155 km (96 miles), respectively although Rowe et al. (1990) do not specify how they arrive at these visual range levels. These visual range estimates are used to determine WTP per mile for only scenarios (2) and (3), since visual range estimates in Rowe et al. are not given for the remaining Schulze et al. photos.

(b) The authors report use values for visibility changes in the Grand Canyon from "poor" to "below average", "poor" to "average", "poor" to "above average", and "poor" to "excellent". Only the use values for changes from "poor" to "excellent" conditions are presented in the table due to space constraints.

(c) Visibility benefit function based on average family income, race, sex, and distance from particular site, etc. This value function is used to derive national benefit estimates from U.S. population.

(d) Estimated visibility function = 251.362 x (1 -exp(-.069 x change in visual range)).

(e) As a follow up to the original study, Tolley found that using electric bills as a payment vehicle reduced the bids by $24 per year. Seasonal variation and the use of larger photographs had no significant impact on bids, however, the value of improving visibility for a greater number of days revealed diminishing marginal value of improvements in visibility.

(f) Estimate of physical siting of power plant included in study. The study does not make use of visual range estimates since the study was commissioned to examine solely the aesthetic damages from the construction of a power plant.

(g) Visual range estimates are determined by converting the mean light extinction coefficient for each market area via the Koschmeider formula, assuming a 5% contrast threshold detection for the observer.

(h) The results depend in large part on what functional form is used. The values reported are for the "best" functional form according to the researchers. These are the log-linear form for the Los Angeles area and the semi-log form for the San Francisco area.

(i) Respondents are asked to value: (1) a shift in visibility conditions from the baseline summer and winter conditions to the improved summer and winter conditions (a shift in conditions of all six scenarios), (2) a shift in only the summer baseline conditions, (3) a shift in just the winter baseline conditions, and (4) finally, a shift in conditions on 20 and 10 days in the winter (movement form 14 miles to 106 miles on 20 or 10 days of the winter).

Table 4. Values of visibility studies

Author	Baseline Visibility	Scenario	Visibility Measure	Link to Pollutants	Visibility Valuation ($/Person/Mile/Yr except where noted) Adjusted to 1990$	Comments

(j) The authors calculate visual range estimates by converting the light extinction measures via the Koschmeider formula, assuming a 2% contrast threshold detection for the observer.
(k) The authors assume 50 million households make up the Eastern U.S. Aggregate Eastern U.S. estimates were changed to household estimates using the 50 million population figure.
(l) The authors use the Tolley et al. (1984) estimates to derive WTP estimates for this study. This is problematic because the visual range differential in Tolley et al.'s study is roughly 10 times the visual range difference between the two SO_2 reduction strategies EPRI valued.
(m) The dollar per visual range mile was calculated as follows. We took the equation the authors adopt: Annual value per HH = 206 x ln (V2/V1) where V2 and V1 are the two visual range estimates and plugged in the different visual range estimates for each of the ten monitors in the baseline and scenario to calculate ten annual values per HH. These ten values were averaged and divided by the difference in visual range over the ten monitors.
(n) One problem the authors did not address is that Denver has a higher than average visual range background, and layered visibility conditions which make it a different city than the common uniform haze seen in most urban surroundings.
(o) Although photographs depict different visibility conditions, no quantifiable measure, like visual range, is equated with the photographs. Therefore, changes in visual range cannot be mapped with respondents' valuation of a given visibility scenario.
(p) The researchers ask respondents to value air quality in terms of visibility and healthiness of the air. In doing so, the respondents bias upwards the value of improving visibility by linking it as a joint product with health effects.
(q) In other words, the visibility impact of a coal-fired power plant operating in the Pacific Northwest depends on where one is located in relation to the power plant. The authors use a value of $19/mile/yr/person -- for a given person, the benefit of not operating the coal-fired plant is $19/year for each mile that person is away from the plant.
(r) Studies relied on ignore the change in ambient visibility <u>and</u> the impact of structure on power plant and plume, which are important to the study BPA conducted.
(s) The photographs shown represent "typical" summer visibility for the parks included. All photos used in the study were obtained from the National Park Service inventory.
(t) The allocation of WTP to existence values includes bequest values, since bequest values and existence values essentially co-exist.
(u) The CV survey asks the respondent to consider the photographs for Yosemite as representative of California parks, the photographs of the Grand Canyon as representative of parks in the Southwest, and the photographs for Shenandoah as representative of parks in the Southeast. The WTP responses are for valuation over these entire regions, rather than for these parks in particular. To calculate WTP for each specific parks, the authors suggest taking the WTP numbers and multiplying by 41% for Yosemite, 45% for the Grand Canyon, and 44% for Shenandoah. These figures were obtained by asking respondents what percentage of their total WTP would be allocated to the park in question.
(v) The actual and projected emissions data are not reported in the study. Visual range estimates for the five scenarios are computed from the emissions data using Trijonis et al. (1980) regression equation relating total extinction to the SO_4, nitrate, and TSP levels and a simple equation showing the relation between total extinction and visual range using the Koschmeider equation, assuming a 2% contrast threshold.
(w) The authors use a constant value per mile change in urban areas, which understates the estimates since WTP can be expected to be higher per mile when pollution levels are higher.

4. OVERVIEW OF EPRI STUDY

To obtain visibility valuation estimates, one would want to have estimates for four relationships: (1) emissions and ambient concentrations, (2) ambient concentrations and optics, (3) optics and perception, and (4) perception and valuation. No study has been able to link all four relationships independently. However, EPRI (1991) is the only study to date that has successfully attempted to piece all four relationships together relying in part on previous studies.

EPRI attempted to measure the summer average visibility benefits for the Eastern United States associated with a decrease in SO_2 by 10 million tons per year under a retrofit strategy (immediate reduction) and a replacement strategy (a phased in SO_2 reduction by the year 2025). EPRI relied on a model called STATMOD, the Statistical Long-Range Transport Model, to determine ambient sulfate concentrations based on projections of future SO_2 emissions under the two scenarios listed above. STATMOD has several important properties to measuring air visibility: (1) it treats aerosol optics, (2) the parameter values are obtained by minimizing the difference between model calculated values and observations of ambient SO_2, aerosol sulfate and precipitation sulfate, and (3) the model is designed to take into account the relationship between source areas and receptor locations. The most important assumption driving the model is that long-term concentrations do not change significantly with chemistry and the weather. Thus, the model is best used for analyzing changes in SO_2 over several years, since the year to year changes in weather are not considered. For a further discussion of the STATMOD model see EPRI (1990).

The data used to determine the model parameters were gridded NAPAP 1985 emission, wind speed and direction from the National Weather Service, and ambient SO_2, sulfate, and precipitation sulfate concentration data for 81 eastern U.S. locations from July to September 1988. Estimates of future ambient sulfate concentrations are based on projections of utility emissions for the High Clean Coal Technology Scenario described in the EPRI report in greater detail.

The relationship between ambient concentration and optics was obtained by reviewing the studies for the NAPAP literature. This literature was important in providing values for light extinction due to non-sulfate material, and light extinction due to sulfate and precipitation sulfate concentration, all of which are used to calculate the light extinction coefficient. Total light extinction was estimated every fifth year from 1990 to 2050 for 33 urban and rural locations in the eastern United States.

To determine the relationship between the change in light extinction and perception, EPRI adopted a threshold perception value of 15% change in light extinction based on a study by Pitchford et al. (1989). The Pitchford analysis was more relevant to the regional haze perception of interest to EPRI, as compared with plume studies in which the perception threshold would be much lower.

Finally, to link human perception to economic valuation of visibility changes, EPRI used the contingent valuation results of the Tolley et al. (1984) analysis. EPRI converted the light extinction coefficients described in the previous paragraph to visual range using the Koschmeider equation, assuming a 4% perceptible contrast threshold. However, the visual range difference between the two SO_2 reduction scenarios is roughly one-fifth the visual range differential Tolley attempted to value in his study. The visual range differential Tolley valued was on the order of 100-233%, whereas the EPRI visual range differential corresponded to a 5-20% change in visual range. EPRI used simple linear extrapolation to convert the Tolley results of approximately $100 per household as a nominal value for a 100% improvement in visual range to an appropriate dollar value for the difference in visual range applicable to the EPRI study. Only differences exceeding the perception threshold were included. An annual dollar estimate for the eastern U.S. was obtained from the dollar per household value under the following three assumptions: (1) 50 million households in the east, (2) summer 50% of the year, (3) exclusion of 20% of summer days for rain or fog. The best estimate of the differences in visual range of 5–20% under the two scenarios is $144 million per year.

The fact that the EPRI report makes the linkage between the four visibility relationships indicates that its contribution to the visibility literature is significant.

...EPRI report makes the linkage between the four visibility relationships...contribution to the visibility literature is significant.

4.1 DECISION FOCUS STUDY (1990)

The Grand Canyon visibility study undertaken by Decision Focus, Inc. was contracted by EPA to examine what households would be willing to pay for visibility improvements resulting from a Best Available Retrofit Technology (BART) action at the Navajo power plant in Arizona. Because the study was contracted to consider the results of this policy action, most earlier visibility benefit studies of the Grand Canyon cannot be compared with the Decision Focus results since other studies did not examine the same sort of visibility change. In particular, other Grand Canyon/Southwest Parkland studies value large annual average visibility levels rather than considering changes on a certain number of days in the winter months, which is what EPA believes would be the benefit from a BART action at Navajo. In addition, other studies consider the impact of visibility changes over Southwest parks in addition to the Grand Canyon.

The early portion of the study took the form of focus groups, and telephone surveys. These results were used to design a CV study to value five major program changes in the Grand Canyon — an aggressive regional visibility improvement program in the first two programs as well as the effects of a Navajo BART action program in the last three programs. In particular, these five major program changes in the Grand Canyon are: (1) a program that would achieve large improvements during the winter and summer in both clear and cloudy weather conditions, (2) a program that would achieve visually apparent improvements only during the summer, (3) a program that would achieve visually apparent improvements only during the winter but in both clear and cloudy weather conditions, (4) a program that would achieve visually apparent improvements only during 20 days of layered haze that occur during some weather days and (5) a program that would achieve visually apparent improvements only during 10 days of layered haze or some winter weather event days. Only the results of the first program can be compared with previous Grand Canyon studies, since it considers changes in visibility regardless of season and weather conditions. This comparison is taken up below.

The Decision Focus study used photographs maintained by the National Park Service, as opposed to computer modified photographs to illustrate different visual range levels. The latter option was eliminated because it ignores the effect of visibility changes on cloudy, misty or foggy days because computer technology does not permit an accurate depiction of haze with cloudy conditions. The photos used by Decision Focus have an advantage over photos used in previous studies. In particular, the researchers tried to use several pairs of photos that kept the season, time of day, and lighting conditions the same. In addition, the photos were chosen to reflect large perceivable changes in visibility in comparison to the photos showing current conditions. Focus group results showed that respondents viewed with suspicion spending money for programs that reflected slight or imperceptible changes in visual range. Finally, the 8 by 12 inch photographs are much larger than photographs used in previous studies.

The researchers used two photoboards with six 8 by 12 inch photographs, representing summer and winter. Each photoboard had three rows representing different air visibility and two columns with current and improved visibility conditions. All six summer pictures depicted clear morning skies while four winter pictures were clear sky morning conditions and two were winter weather conditions. (Apparently, there is significant cloud cover in approximately one-third of the winter.) The researchers determined the levels of visibility represented as light extinction and the number of days represented by each level. Researchers picked visibility levels and frequencies for clear sky conditions in the summer and winter, with three levels for summer and two levels for winter. The pilot study was carried out in St. Louis and San Diego.

The results of this study may be an upper bound to visibility damages from a new power plant as the Decision Focus study addressed the benefits of fairly significant SO_2 emissions reductions from retrofit of an existing plant (ranging from 50% to 90% removal).

The results of this study may be an upper bound to visibility damages from a new power plant as the Decision Focus study addressed the benefits of fairly significant SO_2 emissions reductions from retrofit of an existing plant (ranging from 50% to 90% removal). These reductions would exceed the residual emissions from a new plant meeting applicable Federal standards.[1] In addition, it should be noted that participation of this plant in the marketable permit system for SO_2 would suggest, at least in the first instance, that externalities would be zero as increases in SO_2 from this plant would be offset by decreases somewhere else. To the extent that the marketable permit system is imperfect in accounting for spatially differentiated marginal damages, this conclusion would not hold.

In any event, the study estimated the WTP of both users and non-users who live in St. Louis and San Diego for visibility improvements in the Grand Canyon region larger than that associated with the retrofit and then extrapolated these estimates downwards to match changes in visibility projected for the retrofit. The original estimates averaged about $2.50 per year per person surveyed, although the authors argue that the 5% trimmed mean estimates, which are about $0.50, are preferred on statistical grounds. After extrapolation downwards and application to 100 million U.S. households, total visibility benefits for 50% SO_2 removal are $0.8 million/year, or about $0.47 mills/kWh.

[1]In applying these results to our project, it should be kept in mind that the Navajo Generating Station has 2,250 MW of coal-fired generating capacity, which is 4.5 times that of our incremental plant. As the plant emits 0.82 lb SO_2/MMBtu with a heat rate of 10,300 Btu/kWh, a 50% reduction in SO_2 implies an emissions reduction of 0.00423 lbs SO_2 per kWh.

4.2 CARSON STUDY

An ongoing study by Carson et al. (1989) has been commissioned by EPRI to examine visibility valuation using Cincinnati as a case study. The results discussed below are based on the completion of the first pilot study.

The Carson et al. study is significant in its attempt to address the problem of separation of health and visibility values. The researchers believe that relying exclusively on the assumption that respondents in a CV study can ignore health effects is problematic. They assert that if shown pictures of improved visibility, respondents should be valuing programs anchored to a specific level of health and visibility. Visibility benefits can then be separated from health benefits through hedonic pricing.

The survey instrument used 5 by 6 photographs representing visibility ranges of 1-6 miles, 7-14 miles, and 15+ miles that accompanied a program card which indicated the number of days in the year that would fall into each category. Three potential health effects - no effects, mild effects, and moderate effects - were valued simultaneously with the corresponding changes in visual range. A second version of the same survey was administered to a different set of respondents. The new survey was identical in all respects to the first one, except the programs only included visibility changes. A total of 151 personal interviews were conducted: 76 using the first version, and 75 using the second version.

The results are interesting. First, respondents are willing to accept the notion that health improvements are associated with large changes in visibility, but they are unwilling to separate health and visibility benefits for small visibility improvements. Secondly, people are not identical in their values for visibility and health. In other words, some prefer visibility over health improvements, some prefer health to visibility improvements, and others are able to make tradeoffs between both. As a result of this pilot study, the researchers' conclude that CV surveys should incorporate simultaneous health and visibility changes.

...CV surveys should incorporate simultaneous health and visibility changes.

NAPAP briefly reviewed the preliminary work by Carson et al. (1989) in its review of the visibility literature. The most important criticism NAPAP found of the Carson et al. study is that the respondents were not briefed on what constitutes health effects. Thus, if the intended health effects differed from an individual's perception, the measurement of health impacts will be flawed.

4.3 TRIGONIS ET AL. (1985) STUDY

Trijonis et al. (1985) attempted to measure the visibility benefits associated with different air quality in the areas of Los Angeles and San Francisco. The authors use a hedonic property value study with 4,765 homes in the Los Angeles area and 3,106 homes in the San Francisco region. Home sale prices for the years 1978-79 was used. Visibility data were obtained from weather recordings of visual range at weather stations and airports. The authors convert the visual range indices to extinction indices using the Koschmeider formula relationship in order to emphasize the distinction between valuing a unit change in visibility at lower visibilities than at high visibilities. They measure two different light extinction indicies - the annual median and the annual median subtracting sea haze. The latter extinction index is important because aerosol water contributes an important contribution of fine particulate mass, which affects visibility. The authors find that the latter measure of light extinction is significantly different from the other measure, but that it introduces a large amount of multicollinearity into the data set.

The authors estimate separate hedonic price functions for the Los Angeles and San Francisco areas using a variety of functional forms. They assume that individuals within Los Angeles and San Francisco are identical with the exception of income (visibility is neither a substitute nor a complement with other characteristics). In all cases the light extinction indicies have a significant negative correlation with home sale price, but the coefficients vary considerably depending on the functional form employed.

The hedonic equations are differentiated to determine a WTP for a unit change in extinction. Given the implicit prices of extinction for each individual, a demand curve was estimated by regressing price against quantity. Integrating under the demand curve for any change in extinction rate, yields the aggregate benefits.

The biggest problem with this study, as with all property value studies, is that it is difficult to separate the values for health, and other air pollution effects from the purely aesthetic effect of air pollution, namely visibility. However, one strength of the Trijonis study is its use of light extinction data to measure visibility benefits, as opposed to ambient pollutant concentrations, which are even less reliable.

In addition, another benefit of the Trijonis study is its use of direct market data. Although these benefits are notable, hedonic property studies have limited use in the valuation of visibility outside of urban areas.

Finally, there are two other major criticisms of the hedonic property method: (1) there must be noticeable variation in air quality across the survey market and (2) residents within the defined market must factor visual air quality changes into their purchase of houses (EPRI 1988). For the Trijonis study the first factor holds true. Air pollution is noticeably worse in counties in and around Los Angeles than the counties around San Francisco. In addition, the results of the study seem to suggest that air quality as measured by light extinction is a significant determinant of home sale price. Therefore, these major criticisms of hedonic property studies apparently do not seem to be the case with the Trijonis study. However, since the visibility literature we reviewed is heavily skewed toward contingent valuation studies, it is not possible to draw far-reaching conclusions on the basis of the review of one hedonic property study.

4.4 BROOKSHIRE, IVES AND SCHULZE (1976) STUDY

This study attempts to measure benefits locally, for the residents of Lake Powell recreational area, to avoid the building of a power plant that would impair visibility. The study estimates benefits from the visual impact of the power plants (without smoke emitted) as well as the smoke plumes and loss of visibility.

The starting bid for the game was $1 and bids were increased in increments of $1. The payment vehicle was a user fee. Using the survey data collected, it was shown that the equivalent and compensating consumer surplus measures were not very different, in part because the typical recreator in Lake Powell has substitute recreation opportunities and has an aesthetic valuation as only a small percentage of his income. No strategic bias was seen because there are not a large number of zero and extremely high bids.

4.5 ROWE ET AL. (1980) STUDY

This study attempts to measure the visibility benefits associated with different visual ranges (75, 50, and 25 miles) in the Four Corners Region of the U.S. This study is unique in its ability to link the photographs to actual quantifiable measures of visibility. In addition, the study attempts to test the biases associated with iterative bidding (starting point bias, strategic bias) and the biases introduced by a survey design. In the latter attempt, Rowe et al. (1980) took the Brookshire (1976) study one step further by giving the mean bid of other respondents (to test strategic bias), and varying the starting bids (to test starting bid bias). The effect of increasing the starting bid was approximately $0.60/month on a $1 increase within the $1 to $10 range. The effect of prior information (other respondents mean bids) caused respondents to bid \$1.70/month less than without the prior information. Strategic bias was not perceived to be a problem if zero and large bids are analyzed and possibly rejected. The authors found that the survey instrument structure was not robust to changes in the structure of the survey (i.e., different payment vehicle). These distortions can bias the results up to 40% in this survey.

...the survey instrument structure was not robust to changes in the structure of the survey...can bias the results up to 40%...

4.6 SCHULZE ET AL. (1983) STUDY

This study attempts to measure the visibility benefits of the Grand Canyon and southwestern United States, using entrance fees and electric bill additions as payment vehicles for a WTP survey. No initiation point for the bidding process was suggested which according to the authors eliminates starting point bias. This study is not ideal for examining the effect of different survey parameters have on the results.

The survey does support the Dubos hypothesis which states that once degradation has taken place in a pristine environment, additional damage matters little. The survey supports the notion that incremental improvements in air quality does not lead to small benefits to viewers. Total user value alone is two orders of magnitude smaller than total preservation (user plus existence value) value.

4.7 TOLLEY ET AL. (1988) STUDY

Tolley uses a large sample to estimate a visibility value function based on WTP bids from respondents in six Eastern U.S. cities valuing three changes in local visual range (local is defined as all land within a 75-mile radius of the test city), and two changes in visibility in the eastern and western United States. The bid function the authors adopt has three important properties: (1) the bid for no change from the present situation is 0, (2) increases in visual range will lead to increases in a bid, and (3) any increases in a bid will take place at a diminishing rate. The equation was estimated using non-linear least squares regression. Of the 19 independent variables included in the bid equation all but three are significant at the 95% level. The most important conclusion from the regression equation is the effect of the four city dummies for Atlanta, Washington, Cincinnati, and Miami. (Only four of the six cities were included in the equation since the intercept and another variable used up two degrees of freedom. Therefore, Boston and Mobile remained as the base cities from which to compare.) The city effects for Atlantic and Washington, D.C. are positive, but they are negative for Cincinnati and Miami.

Tolley uses the parameter estimates of the original bid function to calculate a visibility parameter equation for an individual having mean (i.e., average) characteristics. This equation shows that an individual with mean characteristics is willing to pay an estimated $16.83 per year for a 1-mile improvement in visual range.

In the Six City Survey, Tolley used photographs from Chicago to show local visual range at 4, 13, and 30 miles, corresponding to conditions of "poor", "median," and "good" visibility. Respondents were shown these photographs, told the visual range in their own city and asked to value the three changes in local visual range. In addition, respondents were also shown pictures of a scene at the Shenandoah National Park and the Grand Canyon National Park under the same visibility conditions and asked to value a 10 mile improvement throughout the eastern United States and the entire United States, respectively.

It is not clear how the photographs relate to the changes in visual range the respondents were told to bid on.

It is not clear how the photographs relate to the changes in visual range the respondents were told to bid on. For example, the photos of the Grand Canyon and the Shenandoah National Park do not appear to relate to valuing changes in 10 miles for the eastern and entire United States. The authors never disclose the visual range of the pictures of the Grand Canyon and Shenandoah, so it is hard to make the connection from the photos to the valuation questions. Also, it appears arbitrary that the authors select Shenandoah National Park to represent the area of interest to value changes in eastern visibility as a whole. Clearly, it is doubtful that all respondents would choose Shenandoah as a place representative of the entire eastern U.S. In addition, Tolley shows the photos of changes in local visual range in Chicago to respondents from other cities. The problem with this is that scenes from Chicago are unfamiliar to people from other areas and thus it is likely to cause a downward bias in the bids of people outside the city. Also, it is not clear whether the authors discuss the reasons for changes in visibility to the respondents, therefore the respondents could be valuing health benefits as well as aesthetic benefits. Finally, there is no mention of how Tolley has measured the visual range in each photo (i.e., subjectively, or through instrumental methods such as telephotometry) and thus, it is difficult to comment on whether the visual range was measured accurately.

Tolley conducted a follow up study in selected cities to modify some of the problems with the original six cities study outlined above, and to test some different hypotheses. In the follow up study Tolley used photographs that depicted familiar local scenes to show local visibility programs to respondents instead of relying exclusively on scenes from Chicago. In addition, Tolley used 8 by 10 photographs, rather than the original 3 1/2 by 5 photographs, and tested the use of an electric bill addition as a method of payment instead of leaving the payment vehicle abstract. Finally, Tolley introduced the concept of valuing seasonal changes as opposed to changes over the entire year. Tolley estimated the original visibility valuation function under these new conditions. He concluded that the use of larger more distinct photographs using views familiar to local residents might result in higher bids (the t-statistics were not significant at the 95% level). In the test of the payment vehicle, the electric bill addition was found to reduce an individual's monthly bid by $2, ceteris paribus. In the test for seasonal difference, Tolley estimated that people would be willing to pay $0.75 per month more for a 10 mile improvement of typical visibility in the winter than during the summer. Although Tolley corrected the photos for local residents by showing familiar scenes, the remainder of the problems originally discussed still held true with the follow up study.

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PAPER NO. 13*

THE BENEFITS OF IMPROVING RECREATION QUALITY AND QUANTITY

1. INTRODUCTION

Of all valuation areas in environmental economics, studies addressing the nonmarketed services of recreation (mostly fishing and mostly salt water-based) are by far the most prevalent, owing to the early insight on valuation methods offered by the Clawson travel cost model; the theoretical complexities, and thus the academic attractiveness of estimating benefits in this area; many government funders; and widely available data. To give some idea of the magnitude of work in this area, Smith and Kaoru (1990) performed a meta-analysis on 77 studies of recreation demand, and Walsh, Johnson, and McKean (1988) reviewed 120 studies of the value of various types of recreation-activity days. Most of these studies pertain to individual sites or clusters of sites in a region. Some seek estimates of national recreation benefits.

Of all valuation areas in environmental economics, studies addressing the nonmarketed services of recreation...are by far the most prevalent...

The recreation literature may be split into two contexts: the value of adding or eliminating one or more recreation sites (quantity) and the value of changing quality characteristics (e.g., water quality or catch rates) at one or more sites (quality). Of course, these distinctions are more apparent than real. Consider the effects of a new hydroelectric plant. By damming a river, its character may be dramatically altered downstream; upstream, it becomes a lake. Recreation opportunities of a certain type are eliminated, while recreation activities of a different type are added. Does one label these changes "quality" changes or "quantity" changes? More straightforward is the treatment of pollution-related damages. In this case, quantity changes may conceptually be considered the logical extreme of quality changes, that is, when pollution is bad

*Based largely an a paper by Alan Krupnick and Rebecca Holmes.

enough, the site is effectively eliminated from use. To make matters more confusing, in the absence of data linking pollution to quality changes, some analysts make the upper-bound assumption that the recreation opportunity will be eliminated rather than degraded, that is, that there will be a quantity change (RCG/Hagler, Bailly, Inc. 1991).

The state of the literature in this area largely determines the organization of this paper. Surprisingly, while there are several comprehensive reviews of the quantity studies there are no recent reviews of the quality studies. Thus, we also split the literature review into quality and quantity studies. However, owing to the recent and voluminous work on recreation damages associated with acid rain — addressing both a quality change and a quantity change — we treat the acid deposition-recreation pathways together. Thus, the literature review has three parts: non-acidic-recreation quality, acid deposition-recreation, and non-acid-deposition recreation quantity. This paper does not yet contain a review of the last set of studies. Below is a discussion of measuring welfare changes when recreation quality changes.

2. VALUING CHANGES IN RECREATION QUALITY

2.1 THEORY

As a consequence of the research activity in this area, there are a variety of valuation approaches ranging from the very simple to the esoteric [consult Smith (1989) for a summary]. But, to obtain benefit estimates, all of these approaches need estimates from the physical sciences of a long chain of effects. For instance, to estimate the benefits of reducing SO_2 and NO_x emissions along the "acid deposition path," at a minimum one would need information on (i) the effect of these emissions on acid deposition by location and (ii) the effect of deposition on stream and lake quality. Some economic valuation models use these quality changes as a start point. Others, use changes in catch rate. For the latter models, additional scientific information includes (iii) the effect of the quality change on fish populations and (iv) the effect of changes in fish populations on catch rate. The ideal model would need to capture the effects of worse fishing at some sites on recreation choices at all sites and the effects (in the case of environmental improvements) on current nonparticipants that might be induced to participate. In the same model, consistent with welfare maximization, the value recreators place on these changes would also need to be captured.

The simplest valuation approach, called the "unit-day" value approach, is a two-step procedure. It requires, first, estimates of the effect of the change in recreation site characteristics (such as water quality) on recreation participation. Then, from numerous studies in the literature, "prices" for a day of recreation are multiplied by the change in recreational participation to obtain an estimate of benefits. Walsh, Johnson, and McKean (1988) offer a 20-year review of 102 studies using travel cost and contingent valuation (CV) approaches to produce unit values (per activity day) for camping, picnicking, swimming, hiking, boating, hunting, fishing of four types (cold, salt, warm, anadromous), nonconsumptive fishing and wildlife, and wilderness. Some of the values are available for the eastern and western United States.

As the affected sites may have unique characteristics, "average" unit values drawn from the literature may be inappropriate. In particular, the participation choice and valuation are really two sides of the same coin that cannot be neatly separated, as is implied by the unit-day value approach. More important, changes in utility to recreators who continue to participate are ignored by the unit value approach.

There are better approaches. Each of these approaches takes as its point of departure the insight that the WTP for improvements in recreation quality characteristics can be revealed by examining recreation choices across sites that have different levels of quality and that are located at different distances from recreators, that is, one can estimate the tradeoff between quality and quantity of recreation experience and the travel costs and time needed to obtain these experiences.

The three major approaches are (i) multiple site demand models, (ii) discrete choice models, and (iii) hedonic travel cost models [see Bockstael, Hanemann, and Kling (1987)]. The first, also called a varying parameters model, estimates a system of demand equations for all sites in the area, with travel costs, personal characteristics, and site characteristics as arguments (the last indirectly in a second stage). The consumer surplus associated with a change in site characteristics (the willingness to pay for this change) can then be computed. The drawbacks of this approach are that nonparticipation and substitution possibilities across sites are not handled very well.

The discrete choice approach addresses both of these drawbacks by seeking to explain recreator choices per occasion, that is, the allocation of participation across sites. It explicitly allows for an individual to participate at some sites but not at others. The drawback of this approach is that it takes total participation in the area as given. In practice, additional analyses are appended to explain overall participation, using results of the site allocation model as input to the appended analysis. A recent study of swimming visits at 30 Boston beaches is a successful application of this approach that yields benefits in terms of water quality characteristics (Bockstael, Hanemann, and Kling 1987). This approach is also useful in valuing changes in site availability — the quantity issue.

The final approach, the hedonic travel cost model (Brown and Mendelsohn 1984) assumes that each individual can choose among sites with many different bundles of attributes and can trade off some of one attribute for more of another. By observing the sites individuals choose, the price and, ultimately, the demand for each attribute can be recovered. The basic assumptions of this model, like the others, have been criticized. For instance, Bockstael, Hanemann, and Kling (1987) note that there is no market for providing alternative attributes of sites the way there is for houses (an application of hedonic techniques that is theoretically more satisfying), and thus it is questionable how much meaning one should attach to estimates of such prices as estimates of marginal valuation of an attribute. Indeed, values of desirable attributes estimated by this technique have frequently come up negative. Smith, Palmquist, and Jackus (1990) have made a series of modifications to this approach that improve its theoretical and empirical performance, however. They estimate benefits per trip for a 60% increase in catch rate of boat fishing parties in Albermarle-Pamlico Estuary in North Carolina of about $1, in the range found by others for sportfishing in Florida using a discrete choice model (Bockstael, McConnell, and Strand 1989).

The importance of using models that account for substitution between different recreation sites can be seen in the results of Morey, Rowe, and Watson (1991) for salmon fishing in Maine. They found that a single site model estimated WTP per angler per year to avoid elimination of this fishery in the Penobscot River that was three times greater ($2124 versus $764) than an estimate obtained from a discrete choice.

There is also a relatively small literature using contingent valuation approaches to obtain use values for water quality improvements or increases in catch at recreation sites. Some of these studies use a water quality ladder to convey improvements in a river (Smith and Desvousges 1986) and nationally (Carson and Mitchell 1988).

In spite of the vitality of research on recreation benefits, a convincing case has not been made that recreation choice is highly sensitive to changes in site quality or characteristics and, therefore, that changes in quality are highly valued. For instance, a highly competent study of the recreation benefits of pollution reductions at 30 Boston beaches (Bockstael, Hanemann, and Kling 1987) finds per recreator per season benefits from a 10 percent reduction in oil, chemical oxygen demand (COD), or fecal coliform ranging from $0.50 (fecal coliform) to $6.70 (COD) (1989 $s). As another example, a 20 percent increase in catch of striped bass in the Maryland portion of the Chesapeake Bay was estimated to result in annual benefits to bass fisherman of only $2.50 per person, while benefits to beach users of a 20 percent reduction in nutrients in the Maryland portion of the Bay results in benefits of $17 per trip. Contrast these estimates with acute health benefits of $140 per person for a 50-60 percent reduction in ambient ozone concentrations in Los Angeles (NERA 1990).

...a convincing case has not been made that recreation choice is highly sensitive to changes in site quality or characteristics and, therefore, that changes in quality are highly valued.

Given these small values, the major benefits of reducing pollution of recreation assets may well be in nonuse values, not because such values may be large per capita but because so many people may have them. A companion CV study of user and nonuser benefits to improving water quality in the Chesapeake Bay finds use values per household three times nonuse values ($94 per household per year versus $33) for those living in the Bay border States. If nonuse values extend to people living in other States, total nonuse values could easily exceed total use values. Of course, the Bay is a unique environmental asset. One would be surprised to find high existence values for common types of assets. In what is probably an amazing coincidence, a study asking for use and nonuse values for improving Minnesota lakes (Welle 1985) found the same result: the average user's WTP was about three times the WTP of the average nonuser. As all Minnesota lakes are included in the scenario, this group of assets may be viewed as unique in a similar way to that of the Chesapeake Bay.

An important complication for valuing recreational benefits of environmental improvements concerns the appropriate means of accounting for baseline conditions. Valuation studies of damages to lakes from acid rain assume that the historical baseline is appropriate rather than the current baseline — that damages already observed and related to a certain amount of deposition are a reasonable estimate of the benefits to be gained by reducing deposition (or slowing its increase) by the same amount. Far more useful to environmental costing would be to estimate the value of avoiding further deposition to lakes.[1]

[1] If our project were considering the benefits from reducing emissions below the current baseline, then the concept of irreversibilities would also be important to incorporate in the analysis.

3. LITERATURE ON NONACIDIC RECREATION QUALITY

No studies in the literature specifically survey studies valuing quality change, but they often provide sufficient information to distinguish such studies. The surveys we used included Bockstael, McConnell, and Strand (1989); Desvousges and Skahen (1987); Feenberg and Mills (1980); Fletcher, Adamowicz, and Graham-Tomasi (1990); Mitchell and Carson (1989); Smith (1989); Smith and Kaoru (1990); Walsh, Johnson and McKean (1988); Ward and Loomis (1986); and Yardas, Peskin, Krupnick, and Harrington (1984).

In addition to these sources, we contacted recreation demand researchers for studies and reviews and the EPA for a database of nonmarket valuation literature. We searched the bibliographies of recent recreation demand studies and a social science database at the Library of Congress for travel cost and recreation valuation. We reviewed the Subject Index of Articles of the *Journal of Economic Literature* from September 1989 to the present. Our objective in each case was to find recreation demand studies valuing environmental quality change. The result was a preliminary bibliography of about 150 such studies. Where the information was available, we recorded each study's method, the study location, the recreational activity, and the environmental quality measure, in addition to bibliographic information.

Because of the large number of studies available, we were able to discriminate among them and analyze only a subset. We focused first on the methodologically superior studies, those using the discrete choice or Random Utility Model (RUM). According to Parsons and Kealy (1990), "Random Utility Models appear to have emerged as the model of choice for recreation decisions."[2] After analyzing the RUM studies, we narrowed down the remainder by looking at studies valuing assets relevant to the reference environments. For instance, we excluded the many marine recreation analyses and focused on those valuing rivers, lakes, streams, etc. We also analyzed those works most frequently cited in reviews. Twenty-one studies were included in our literature tables — Tables 1 and 2.

The following information is included in the tables, chosen with an eye for assessing the quality of the study and its usefulness in a benefit transfer. There are two types of tables, general features and valuation features. In the general feature studies, the study characteristics are author/year, location/asset type, recreational activity, method (model), sample population/sample

[2]Parsons, G.R. and M.J. Kealy, "Measuring Water Quality Benefits Using A Random Utility Model of Lake Recreation in Wisconsin," report to the EPA.

Table 1. Key general features of recreation studies

Author(s) and Year	Location/Asset Type	Recreational Activity	Method	Sample Population/Sample Size Number of Sites: Geographical Extent
McConnell, Strand and Blake-Hedges (1991)	Maryland, Atlantic shore	Shore fishing: small gamefish	TCM (Discrete choice, simple)	Atlantic anglers living in Maryland/38 anglers taking 258 trips total 7 sites: Maryland counties or groups of counties
Bockstael, Hanemann and Kling (1987)	Boston-Cape Cod area, beaches	Swimming	TCM (Discrete choice, nested)	Random households in Boston SMSA/Sample size not reported (1975) 30 sites: Boston and Cape Cod
Bockstael, McConnell and Strand (1989)	East Coast of Florida, Atlantic Ocean	Fishing: big game, small game, bottomfish	TCM (Discrete choice, nested)	Subjects intercepted at sites on East Coast of Florida (Nov/Dec 1987)/158 people, 161 trips) 9 sites: Florida counties or groups of counties
McCarthy, Tay and Fletcher (1991)	Indiana, U. S. freshwater sites visited by Indiana anglers	Freshwater fishing	TCM (Discrete choice, simple, with portfolios of recreation trips as choice alternatives)	Indiana residents/573 observations 368 sites (five considered for each respondent): U.S. regions or zones
Parsons (1991)	Wisconsin, lakes larger than 100 acres, Michigan and Superior excluded	Boating, angling, swimming, viewing	TCM (Discrete choice, nested)	Wisconsin residents/nearly 1200, 1133 sites, sample of 24 considered for each respondent: lake in Wisconsin
Parsons and Kealy (1990)	Wisconsin, lakes larger than 100 acres, Michigan and Superior excluded	Boating, angling, swimming, viewing	TCM (Discrete choice, nested)	Wisconsin residents/over 1200 1133 sites, sample of 12 considered for each respondent: lakes in Wisconsin
Parsons and Kealy (1988)	Wisconsin, lakes larger than 100 acres, Michigan and Superior excluded	General (boating, fishing, swimming, picnicking: not valued separately)	TCM (Discrete choice, simple)	Wisconsin residents/105 for site choice model (~2600 trips); 330 for participation model 1100 sites (of these, lakes visited define choice set for individual): lakes in Wisconsin

Table 1. Key general features of recreation studies

Author(s) and Year	Location/Asset Type	Recreational Activity	Method	Sample Population/Sample Size Number of Sites: Geographical Extent
Morey, Rowe and Watson (1991)	Penobscot River, Maine	Salmon fishing	TCM (Discrete choice, repeated)	Maine residents holding Maine Atlantic salmon fishing licenses in 1988/168 8 sites: Maine rivers and groups of rivers, New Brunswick rivers, Nova Scotia rivers, Quebec rivers
Kaoru and Smith (1990)	North Carolina, Albemarle-Pamlico Estuary, Atlantic Ocean: tidal fresh and mixed fresh-saltwater sites	Sport fishing	TCM (Discrete choice, simple)	Fishing party leaders or boat owners fishing at the estuary/612 observations Scenario (a) 35 sites: estuary launch points Scenario (b) (Pamlico-Tar river subset of estuary sites) 9 sites: estuary launch points
Morey, Shaw and Rowe (1991)	Oregon, coastal waters	Marine recreational fishing	TCM (Discrete choice, repeated. Authors' modification of standard model)	Oregon resident anglers/5855 7 sites: coastal counties
Rowe, Michelson and Morey (1989) (Chapter 5, TCM model)	Penobscot River, Maine	Atlantic salmon fishing	TCM (Discrete choice, repeated)	Atlantic salmon fishing license holders in Maine/421 8 sites: Maine rivers and groups of rivers, New Brunswick rivers, Nova Scotia rivers, Quebec rivers
Caulkins, Bishop and Bouwes (1986)	Wisconsin lakes	General lake recreation. Type not specified in model due to data limitations.	TCM (Discrete choice, nested)	Lake users residing in northern Wisconsin/45 Number and definition of sites not reported
Milon (1988)	Dade County, Florida/Artificial marine habitat	Near shore and offshore sportfishing: not broken down by species	TCM (Discrete choice, nested)	Dade County boat registrants/887 respondents 13 sites: known "fishing spots", natural reefs, artificial reefs
Smith, Desvousges and McGivney (1983)	Monongahela River, Pennsylvania	Boating, fishing, swimming	TCM (Varying parameters)	Households in five Pennsylvania counties/69 respondents taking 94 trips 13 sites along the Monongahela River

Table 1. Key general features of recreation studies

Author(s) and Year	Location/Asset Type	Recreational Activity	Method	Sample Population/Sample Size Number of Sites: Geographical Extent
Brown and Mendelsohn (1984)	Washington state streams	Steelhead fishing	TCM (Hedonic)	Washington licensed fishermen/5500 Over 140 Washington rivers
Gramlich (1977)	Charles River, Boston, Massachusetts	Swimming	CVM, combination of iterative bidding and open-ended	Households in the Boston metropolitan area/165
Johnson and Adams (1988)	Oregon/John Day River, a Columbia tributary	Steelhead trout sport fishing	CVM, open-ended valuation questions	John Day River anglers/62
Smith and Desvousges (1986): CVM	Monongahela River, western Pennsylvania	Boating, fishing, swimming	CVM, direct question, payment card and iterative bidding	Households in 5 counties in southwestern Pennsylvania/301 interviews completed (smaller sample size for each valuation method: 54 for payment card, excluding protests and outliers)
Smith and Desvousges (1986): TCM	Army Corps of Engineers flatwater recreation sites nationwide (include one in TN reported here as an example)	Fishing, sightseeing, picnicking, camping, boating, swimming	TCM (Varying parameters)	Site users/1891 22 sites: Flatwater sites in 9 states (lakes and reservoirs)
Brookshire, Eubanks and Rowe (1977)	Wildlife population levels (antelope, deer and elk) in Eastern Powder River Coal Basin, Wyoming	Hunting	Hedonic price equation derived from household production function	Wyoming hunters/Sample size not reported Number of sites not reported: Hunt areas in Wyoming
Bouwes and Schneider (1979)	Pike Lake, Wisconsin	Not specified	TCM, pooled	Wisconsin lake users/195 8 southeastern Wisconsin lakes

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues
McConnell, Strand and Blake-Hedges (1991)(a)	Mean of catch from 1980-1988	Five percent increase in historic catch	Historic catch rate: mean catch per day of small game for period 1980-1988 at site	$10.36 (angler's mean WTP per trip)	Catch/hour; travel cost; travel time	Subjects were anglers surveyed by phone who agreed to participate in study. (1988) Aggregated sites: counties or groups of counties
Bockstael, Hanemann and Kling (1987) (b)	1975 water quality? Study not explicit	(a) 10% reduction in parameters at all sites (b) 30% all sites (c) 30% downtown Boston beaches	Water quality parameters: Oil, Chemical Oxygen Demand (COD), Fecal Coliform, Turbidity	(a) Oil: $ 2.54 COD: $ 7.02 Coliform: $ 0.50 (b) Oil: $12.35 COD: $18.95 Coliform: $ 7.55 All + turbidity: $31.91 (Benefits per household per season)	Stage I est: oil; fecal coliform; temperature; COD; turbidity; noise; public transportation; beach ethnicity; trip cost Stage II: price; household size; % children; swimming pool access Participation: cost and quality of rec. . opportunities; HH size; % children; sports equipment	Random survey of Boston households: participants and non-participants (1975) No site aggregation: all 30 used separately
Bockstael, McConnell and Strand (1989) (c)	Mean of catch and success rates from 1980-86	20% increase in catch and success rates for three species categories	Proxies for expected catch: catch rate--mean # of fish caught in Nov/Dec form 1980-86 (small and bottomfish) success rate--mean of proportion of anglers catching at least one fish in Nov/Dec from 1980-86 (big)	Small: $0.38 Bottom: $1.46 Big: $1.79 (Benefit per person per trip)	Fishing mode (shore, boat); species group; travel cost; travel time; boat ownership	Subjects were anglers surveyed by phone who agreed to participate in study. (1988) Aggregated sites: counties or groups of counties

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues
McCarthy, Tay and Fletcher (1991) (d)	1985 estimates of regional water quality indices	(a) 2% reduction in indices, assuming travel costs of 15 cents per mile (b) 2% reduction, 25 cents per mile	(see comments) Portfolio's water quality defined as sum of objective indices of each site's water quality, weighted by time consumer spends at site.	(a) TSS: $1,022,450 Phos: $1,560,900 Iron: $1,282,600 PCB: $747,780 FCB: $1,312,850 All: $5,953,200 (b) TSS: $1,704,890 Phos: $2,601,500 Iron: $2,601,500 PCB: $1,256,300 FCB: $2,187,680 All: $9,923,210 (Benefits of abatement to Indiana anglers: annual?)	Education; income; fish species; site type; coliform; TSS; iron; phosphorous; oxygen demand; travel distance	Uses 1985 survey of fishing, hunting, and wildlife-associated recreation, by the U.S. Census Bureau: sampling method unclear. Sites defined as 368 U.S. regions or zones: assume aggregated
Parsons (1991) (e)	1978 water quality	(a) High standard: high DO and clarity for all lakes (b) Low standard: no lakes have low DO	Dissolved oxygen (DO): high, moderate or low Clarity: high or low	(a) High standard: Boating: $8.44 Fishing: $1.88 Swimming: $11.64 Viewing: $11.06 (b) Low standard: Boating: $0.38 Fishing: $1.00 Swimming: $1.66 Viewing: $0.30	Travel cost; lake size; commercial facilities; remoteness; northern county; lake depth; boat ramp at lake; inlet in lake; dissolved oxygen; clarity	Random phone survey (1978) No site aggregation
Brookshire, Eubanks and Rowe (1977) (t)	Based on average of harvested animals for 3 year period 1971-73: Deer: pop. 16,800 Antelope: pop. 30,300 Elk: pop. 320	Uniform decline of deer and antelope populations across 9 antelope and 9 deer hunt areas in region by 1978: Deer: 5% decline Antelope: 9% decline Elimination of elk population	Deer, antelope and elk populations	$420 (Value per year for all hunters)	Hunting days per individual; stock of wildlife species; number of hunters in hunting group; density of all hunters at hunting site	

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues
Bouwes and Schneider (1979) (u)	LCI = 3	LCI = 10	LCI: Uttormark's Lack Condition Index for Wisconsin lakes. Index parameters are DO, Secchi depth, fish winterkill and macrophyte or algal growth. Index values range from 0 (best) to 23 (worst)	$0.50 (Benefit per year per recreator if damage is avoided)	Cost; perceived water quality (related to LCI: see comments); round trip time; income	Little information given on data
Parsons and Kealy (1990) (f)	1978 water quality	Scenarios 1 through 4, with increasing minimum water quality standards at all Wisconsin lakes	Water quality measures: Dissolved oxygen (DO); Secchi depth transparency; macrophytic algal growth	Scenario 1: $ 55.40 Scenario 2: $161.40 Scenario 3: $460.40 Scenario 4: $819.40 (Values per individual per season)	Site choice: lake size; public access; DO; clarity; winterkill; algal growth; price (incl. TC) Participation: lake property ownership; age; proximity to a Great Lake; importance of water quality to respondent; education; length of residence in Wisconsin	EPA random phone survey of Wisconsin residents (1978)

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues
Morey, Rowe and Watson (1991) (h)	1988 reported catch rate	(a) Doubling catch rate at Penobscot (b) Halving catch rate	Average catch per trip	(a) $719.40 (b) $336.60 (CV/angler/year, assuming 50 trips per year)	Income; fishing experience; membership in fishing club; age; trip costs; catch	Mail and telephone survey of Maine salmon license holders (1988) Aggregated sites: some are individual rivers, others groups of rivers, some including all rivers in a province. Authors state that catch varies less within groups than across groups. Source of catch data not reported: assume it is catch reported by surveyed anglers.
Kaoru and Smith (1990) (i)	1981-82 values of catch and water quality	(a) 1. Increase catch rate by one fish at all sites. 2. Increase catch rate 25% at all sites (b) Pamlico-Tar River Plan 1. 30% decline in N and Ph loadings, 30% increase in catch rates 2. 30% catch increase 3. 30% N/Ph decline	Catch rate: average over respondents of fish per person per fishing hour for each entry point Water quality: aggregates of discharges of nitrogen and phosphorus from point, nonpoint, upstream sources; discharges of BOD and TSS from plants within 10 miles upstream from sites	(a) 1. $6.79 2. $6.82 (b) 1. $11.26 2. $1.47 3. $5.89 (per person/per trip)	Travel cost; catch rate; type of boat ramp; area water classification (fresh, mixed); nitrogen loadings; phosphorous loadings	Intercept survey of estuary users Site aggregation: none in scenario reported

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues
Morey, Shaw and Rowe (1991) (j)	Charter boat catch rate: 1.27 Private boat catch rate: 0.70	Increased catch: 2.27 charter 1.70 private Salmon enhancement program funded by $5 charter boat tax: WTP measured for increased catch and tax	Offshore salmon catch rates (average per-person, per-trip) in Clatsop County	(By county of origin) Clatsop: $1.51 Tillamook: $0.94 Lincoln: $0.53 Lane: $0.26 Douglass: $0.17 Curry: $0.09 Multnomah: $0.83 Deschutes: $0.19 (benefit per year per angler)	Price; salmon, perch, smelt/grunion, flatfish, rockfish/bottomfish catch rates	On-site surveys of anglers (1981): authors make correction for selection bias aggregated sites. Incomplete data on individuals' site selections; no data on income, boat ownership, species preference, age, sex, race, education
Rowe, Michelson and Morey (1989) (Chapter 5, TCM model) (k)	1988 catch rates	(a) Doubling catch rate at Penobscot (b) Halving catch rate at Penobscot	Salmon catch rates	(a) $688 (b) -$328 (value/angler/year)	Travel cost; catch rate; experience; fishing club membership age; income	Mail and phone survey of randomly selected license holders (1988) Aggregated sites: some are individual rivers, other groups of rivers, some including all rivers in a province
Caulkins, Bishop and Bouwes (1986) (l)	Not reported	One LCI unit improvement in water quality at Shadow Lake	Uttormark and Wall's Lake Classification Index (LCI)	Not calculated. Calculates change in trips from change in water quality.	Site choice: travel cost; LCI; urban shoreline; lake depth Participation: travel cost; LCI; urban shoreline; lake depth; age; recreation income of HH; number of people in individual's recreation group	Uses statewide water quality survey (as do Parsons and Kealy) for data on recreation

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues
Milon (1988) (m)	1985 catch	10% increase in catch within site group (near shore, offshore-natural, offshore-artificial)	Catch: weight of fish caught	Not calculated. Study reports change in site group choice probability resulting from change in catch.	Site choice: travel cost; travel time; catch weight; catch variability; age of site Habitat choice: boat equipment; opinion on productivity of artificial habitat; race; years boating in Dad Offshore vs. Near-shore: engine size; boat equipment; age; income	Mail survey, stratified sampling proportional by zip code
Smith, Desvousges and McGivney (n)	Current DO levels, adequate to support boating	(a) Increase DO to fishable levels (b) Increase DO to swimmable levels	Dissolved oxygen (DO), percent saturation	(a) $9.76 (b) $20.50 (benefits per household per season)	Not specified for Monongahela application. If same as for Army Corps sites application: Stage I: travel cost; income Stage II: total shore miles; recreational/developed access areas at site; water area/site area; mean DO; variance in DO	Stratified cluster survey of PA households, but only those who had taken at least one trip were used
Brown and Mendelsohn (1984) (o)	Gives "average" price for unit of catch	See baseline	Catch: number of fish caught per ten days	$4.80/trip (average price of catch per ten days) $110/season (per-trip price x average number of trips)	Stage I: fish density, scenery rating Stage II: (inverse demand) income; experience; scenery; lack of congestion; fish density; number of trips	Random sample of license holders

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues
Gramlich (1977) (p)	Charles River water quality in 1973	Increase in water quality throughout river to level "B", guaranteed by state and local government. Payment vehicle: tax increase	Water quality level "B": "clean enough for swimming, fish and wildlife but not necessarily good enough for use as a drinking water supply without further treatment."	$89.79 (value per household per year, using sample mean values of independent variables)	Income; high school education; proximity to Charles (dependent variable is WTP for specific quality change: quality not itself an argument)	Survey of both users and non-users
Johnson and Adams (1988) (q)	Expected catch rate with current fish populations 9.3 hrs/fish	(a) 7.1 hrs/fish (33% increase in population) (b) 5.0 (67%) (c) 2.9 (100%) Payment vehicle: fee in form of steelhead stamps	Changes in angler's mean expected catch rate with increases in fish population	(a) $10.21 (b) $13.22 (c) $16.17 (WTP/angler for increased population/catch)	Catch rate	On-site sampling Personal interviews
Smith and Desvousges (1986) CVM (r)	1981 water quality, represented as Level D	Four possible changes in water quality levels: (a) avoid decrease D->E (b) increase D->C (c) increase C->B (d) increase D->B Payment vehicle: increase in taxes and prices	RFF water quality ladder levels: A: Drinkable B: Swimmable C: Fishable D: Boatable E: No recreation use possible	(a) $8.93 (b) $14.00 (c) $8.93 (d) $23.20 (values per person per year)	Age; willingness to pay cost of water pollution	Stratified cluster sampling: HH's divided into clusters of ~7, entire clusters selected; face to face interviews; users and nonusers included
Smith and Desvousges (1986) TCM (s)	Boatable water quality	(a) water quality change from boatable to fishable (b) boatable to swimmable	RFF Water Quality Index levels: boatable, fishable, swimmable. Each associated with a particular DO level.	(for Cordell Hull Dam and Reservoir, Tennessee) (a) $30.69 (b) $68.08 (benefit per season per user)	Stage I: travel costs (income not significant) Stage II: size of water pool/size of site (income parameter only); DO (intercept only); variation in DO (intercept only); (shore miles, access areas not significant)	On-site interviews. Used maximum likelihood estimator in estimating demand curves to reflect truncation and censoring.

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues
Brookshire, Eubanks and Rowe (1977) (t)	Based on average of harvested animals for 3 year period 1971-73: Deer: pop. 16,800 Antelope: pop. 30,300 Elk: pop. 320	Uniform decline of deer and antelope populations across 9 antelope and 9 deer hunt areas in region by 1978: Deer: 5% decline Antelope: 9% decline Elimination of elk population	Deer, antelope and elk populations	$420 (value per year for all hunters)	Hunting days per individual; stock of wildlife species; number of hunters in hunting group; density of all hunters at hunting site	
Bouwes and Schneider (1979) (u)	LCI = 3	LCI = 10	LCI: Uttormark's Lake Condition Index for Wisconsin lakes. Index parameters are DO, Secchi depth, fish winterkill and macrophyte or algal growth. Index values range from 0 (best) to 23 (worst)	$0.50 (benefit per year per recreator if damage is avoided)	Cost; perceived water quality (related to LCI: see comments); round trip time; income	Little information given on data

Comments Code

Note: For conversion to 1990 dollars, I assume study gives values in dollars of the year the data is collected, unless otherwise specified.

(a) Method: Models catch as random variable (Poisson): function of historic catch rate, on-site time and skill; does not model change in participation due to quality change. Treatment of time: On-site: no cost imputed: brings only utility. Travel: explicit cost: for anglers with flexible work hours, travel time is valued at the wage rate; implicit cost: opportunity cost of discretionary time. Authors model by including travel time directly in utility function. Value: calculated using expected catch which varies across individuals. Authors also calculate benefits using mean angler characteristics to determine average expected catch: value is $9.00. Subjects: anglers' age varies form 18 to 80, with mean 39.4 and SD 14.55.

(b) Method: Authors also estimate hedonic travel cost equations, but results are counter-intuitive (many negative hedonic prices for quality) and they cannot estimate welfare changes. Models change in participation due to quality change; derivation of travel costs, including treatment of time, unclear. Value: Authors also calculate values per choice occasion.

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues

(c) Method: Uses data on fishing trips in Nov/Dec only. Does not model change in participation due to quality change. Treatment of time: Travel: 80% of wage for wage-earners who can vary time; travel time included directly for those who cannot vary time (refer to Bockstael, Strand and Hanemann, 1987).

(d) Method: Defines portfolio of recreation trips as choice alternatives. Assumes all recreation decisions made at beginning of period: destination, frequency and duration decided jointly. Appears to model change in participation due to quality change ("increases in the level of pollutants present in the water decrease the demand for recreational fishing," p. 15). Treatment of time: Estimates effect of distance on choice, then measures welfare by assuming two different per-mile costs of travel. No explicit treatment of the cost of time. Value: Authors also report benefits of 1% reductions, which are about 50% less. Used sampling procedures (of destination, frequency and total number of trips) to limit number of choices available to each consumer; corrected for bias introduced by sampling. Assumes utility derived from a day in a region depends linearly on water quality. Subjects: Heterogeneous income, age, sex, marital status, education, employment (Table 2 of study). Assumptions: Utility of an n-day trip same as utility of n one-day trips. Utility depends linearly on water quality.

(e) Method: Same basic study as Parsons and Kealy (1990). Uses randomly selected subset of all sites as individual's choice set, to limit choices without site aggregation bias. Model differs from Parsons and Kealy (1990) in that: authors estimate models with different numbers of selected sites in choice set: 3, 6, 12, and 24 (vs. 12 in P&K). Authors are more confident of estimates for larger choice sets. However, some parameter estimates are stable even for 3 site model, and benefit estimates are fairly stable for 6 and greater-site models. Authors do not model change in participation due to quality change. Also estimate a model using only sites visited by at least one respondent, and find parameter and benefit estimates robust. Time: 1/3 wage rate. Value: Study also reports values for 3, 6, and 12-site models, and model with visited sites only. Assumptions: Independence of Irrelevant Alternatives holds within any nest in model.

(f) Method: Authors use randomly selected subset of all sites as individual's choice set: limits choices without site aggregation bias. Treatment of time: time valued at one-third wage rate. On-site time assumed constant, equal to 4 hours. Models change in participation due to quality change. Value: Values presented are upper bounds, assuming the increase in number of trips predicted by participation equation: also present lower bounds, assuming no change in total trips. Also present values per choice occasion.

(g) Method: Study defines individual's choice set as lakes he/she actually visited. For nonparticipants, this means their choice set contains no sites, and it is assumed that they will always be nonusers. Models change in participation due to quality change. Treatment of time: time valued at one-third wage rate: wage rate defined as family income/average number of working hours in a year. Travel time calculated as distance/average speed of 35 mph. Value: Values per choice occasion also reported. Spread of values reported. Assumptions: Independence of irrelevant alternatives (IIA). No diminishing marginal utility to visiting a given site during a season.

(h) Method: Models participation decision by including "not fishing" as one of choices . Alternative models: authors also estimate: 1) repeated discrete choice model without income effects; 2) standard logit model without income effects or nonparticipation alternative; 3) share model of site choice; 4) single-site linear demand model and find their model preferable to all. Authors use estimated equations to predict visits to Penobscot under current conditions: estimate is 9.75 vs. and actual 11.85 visits. Some indication of predictive power of model. Treatment of time: travel and on-site time valued at one-third the wage rate. Value: Value reported is mean value: median values 20-33% less. Also reports EV; values nearly identical.

(i) Method: Study investigates effect of site aggregation on benefit measures by using three site definition scenarios with different degrees of aggregation: 1) 35 sites: estuary launch points; 2) 23 sites: groupings of proximate launch points; 3) 11 sites: coastal counties within estuary. Small effects on estimated parameters, but significant effect on benefit estimates. Study tests hypothesis that behaviorally relevant quality measure is a composite of "objective" quality measures. Quality definition found to affect benefit estimates (Scenario (b)). Treatment of time: travel time derived from distance by assuming average speed of 40 mph. Cost of time estimated as wage rate using a 1977 hedonic wage model and 1978 data scaled to survey year with the CPI. For students, unemployed and retired subjects, cost of time estimated as minimum wage. Does not model change in participation due to quality change. Value: Values reported are derived from least aggregated (35 and 9-site) models using nitrogen and phosphorus loadings as quality measure. Authors also report results from models with different aggregation levels and using BOD and TSS as quality measure. Coefficients of BOD and TSS were not significant in estimated equations.

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues

(j) Method: Authors develop a version of a repeated discrete choice model which allows estimation with incomplete trip data and corrects for oversampling of "avid" participants due to on-site survey. Models participation by including "not fishing" as one of choice alternatives. Treatment of time: model includes the "value of time" in travel costs, but study does not describe its derivation. Estimation equation: coefficients of expected sign, except for flatfish and rockfish/bottomfish, with negative coefficients. Authors cite possible multicollinearity. T-statistics not given. Subjects: All counties in Oregon represented. Assumptions: Constant marginal utility of income: no income effect.

(k) Method: Models participation by including "not fishing" as one of choice alternatives. Treatment of time: valued at one-third the wage rate. Assumes travel time = distance/40 mph. Value: Mean value reported. Authors also report median, which is lower, and range.

(l) Method: Study compares RUM to pooled model. Models participation as function of water quality. Treatment of time: Values travel time at 1/4 hourly wage rate. Entire travel cost expression (transportation costs + time costs + fees) deflated by opportunity cost of time. Same data set as used by Parsons and Kealy (1990) (P&K, p. 3).

(m) Method: Study measures travel costs as costs in traveling from launch site to fishing site, not costs from home to launch site. Nested discrete choice model: 1) Offshore or near shore; 2) If offshore, natural or artificial; and 3) Site choice. Does not model change in participation due to quality change; assumes total trips taken is exogenously determined. Authors use model primarily to estimate the value of new artificial reef sites. Could also, presumably, use to value change in catch.

(n) Method: Authors estimate model using data on Army Corps of Engineers sites first, reporting estimation equations, but do not calculate benefits from quality change. Then they calculate benefits form quality change using Monongahela River data, but do not report estimation equations. Treatment of time: Travel time valued at wage rate, using a hedonic wage model to calculate wage. On-site time not considered as a cost. Value: Not clear whether value given is for change in quality at one or all sites. Spread of values also reported. Assumptions of model: Single purpose trips only. SE characteristics besides income not included in specification.

(o) Method: Hedonic travel cost method: based on the idea that consumers will travel farther to higher quality sites, method estimates demand for site characteristics: Stage I: regress travel cost on site characteristics to determine implicit prices for characteristics. Stage II: regress prices on level of characteristics consumed (and other variables) to produce inverse demand functions for characteristics. Treatment of time: valued at 30% of wage rate. (Gave stablest parameters.) Value: Survey year not reported: unclear in what year's dollars values are measured. Demand functions for scenery and congestion also calculated. Assumptions: No user fees.

(p) Method: Bias: Measurement: author not confident of accuracy of income measurement. Strategic: author believes it was clear to respondents that results would not affect policy. Hypothetical: No real cost to answer: upward bias. Abstract/unfamiliar good: downward bias. Starting point: author acknowledges possibility of bias in using iterative bidding game, but notes that, with starting point of $10 and average WTP of ~$30, bias will be downward. Sequence: author asks for WTP for Charles immediately after asking for WTP for all U.S. rivers. Value: Author reports 95% confidence interval for mean WTP. Author also reports values for: WTP for Charles for various levels of income and education, assuming typical values for other variables. WTP to increase water quality to level "B" for all rivers in U.S. WTP to increase water quality to level "B" for all rivers in U.S. excluding the Charles.

(q) Method: Study values incremental streamflow changes by estimating: 1) Steelhead fishery productivity as function of streamflow and 2) value of change in fishing quality (using CVM). CVM method: 1) Gave anglers information on average success rates for each of past five years at the John Day River. 2) Asked to estimate own catch rate at the river in an average year. 3) Gave 3 scenarios: 33%, 67% and 100% increase in steelhead stock. Asked to estimate expected increase in catch rate. 4) Asked maximum fee would pay for improvements.

(r) Method: Four valuation methodologies employed: 1) Iterative bidding: $25 starting point; 2) Iterative bidding: $125 starting point; 3) Direct question; and 4) Direct question with payment card. Statistically significant differences between values derived form 1. and 2. for some scenarios. Respondents asked to include use and option values in bids, then what portion of total they allocate to use. Value: This value for subset of survey for which direct payment with payment card method was used. Also, report values for other methods. Also report values for option + use value as well as use value alone. Use values were statistically different from zero for some, but not all, scenarios and methodologies.

Subjects: Education - Mean 12.75 - Standard Deviation 1.73
Race (white=1) - Mean 0.90 - Standard Deviation 0.30
Income (1990$s) - Mean 28,135 - Standard Deviation 18,985
Age - Mean 47.82 - Standard Deviation 18.34

Table 2. Values from recreation studies

Author(s) and Year	Baseline	Scenario	Damage Measure (Startpoint)	Value of Quality Change (1990 $'s)	Significant Variables	Data Issues

(s) Method: Treatment of time: used hedonic wage model estimated form 1978 Current Population Survey. Used predicted wage as opportunity cost of time. Did not include substitute prices or characteristics in site demand estimation. Authors also use Corps sites model to estimate values for changes in water quality at Monongahela sites, but find the results implausibly high (>30X CVM measures). They attribute this to the physical differences between Corps and Monongahela sites. Value: Values given here are Marshallian welfare measures: Hicksian measures are $100.53 and $299.55, respectively. Values also reported for each of 20 other sites. Subjects: Variety of Characteristics: for example, average income ranges from $9,199 to $29,571 ($19,870 to $63,873, 1990 $s).) Assumptions: Constant on-site time.

(t) Method: Basically a hedonic travel cost approach, except that time costs are not included in expenditure estimates. Characteristic demand functions do not include income as an argument, because of data limitations. Value: Authors reported PV of loss over 30 years at different discount rates (zero and 8%). I assumed value spread over 30 years beginning in 1978, determined annual benefit and converted to 1990$s assuming value originally in 1978$s. (If I assume instead that losses occur over 30 years starting from 1975, with no losses until 1978, and using 1975 as base year, the value is $566.) Assumptions: Authors use harvest of animals as proxy for populations in model, and assume that a given percentage change in population will cause an equal percentage change in harvest.

(u) Method: Authors estimate relation between perceived and objective water quality measures. In interviews at lakes, asked for respondents' perceptions of lake's water quality, using a scale of 0 to 23. They then regressed lakes' perceived water qualities on their LCI's to obtain an expression relating the two. Using this expression, they were able to measure the welfare change from a change in LCI using demand equations estimated using perceived water quality. Did not include cost of time in trip cost; instead, included time directly in demand equation. Value: I assumed values given were in 1977 dollars.

size/number of sites, and geographical extent. In the valuation features table, we include information on baseline, scenario, damage measure (start point), the valuation estimate, significant variables, and data issues.[3] Other comments are included in extensive endnotes on the table.

For the WTP estimates, we index all reported values to the same year and translate where possible into the same units; this allows us to see the range of values and whether they are consistent across studies. However, this may *not* always be possible. As noted above, inconsistent damage measures and baselines preclude comparison. Different scenarios do the same: if one study values a 10% increase in catch, and another a 50% increase, the values are not comparable.

The studies surveyed were primarily recent. Most were published (or written) after 1988; only three, before 1980. They valued a variety of assets, primarily freshwater lakes, rivers and streams. The majority studied fishing benefits, though some valued swimming, boating, viewing, picnicking, camping, or hunting. A few of the RUM studies were of marine recreation. Most of the studies used the RUM, with the rest divided between other travel cost and contingent valuation methods. Average sample size was 1014 (486 excluding the two highest and two lowest). Many of the travel cost studies included sites across a State, though some were less extensive (a river, several counties, a metropolitan area) and some more extensive (sites in nine States, several States and a province, etc.).

Damage measures were highly variable. Many valued catch rate for a fish species. Scenarios and methodologies differed greatly. For studies reporting values per person per trip, the average value was $7 for an increase in catch rate. Annual values ranged from $0.09/person to $719.40/person, with typical values in the hundreds. However, the scenarios valued vary so much that comparison is not very meaningful.

For studies reporting values per person per trip, the average value was $7 for an increase in catch rate.

The other predominant category of damage measures was objective water quality measures, most frequently dissolved oxygen. However, in these studies the scenarios are so variable that no comparison of the reported values is useful.

[3] A data issue specific to multiple site travel cost models is site aggregation. Because of computational requirements, the RUM studies are particularly likely to define sites as aggregates of sites; this may introduce bias if the aggregate sites are large and heterogeneous.

A particularly important study for our work is a recent study by Jones and Sung. Jones and Sung (1991) present results from a random utility travel cost model developed for the valuation of environmental quality at Michigan recreational fishing sites. Specifically, they calculate the damages to Michigan-licensed recreational anglers from fish kills caused by the operation of the largest pumped-storage plant in the United States, and they calculate the benefits of cleaning up PCB contamination in a river in Michigan. The strength of the study lies in the authors' improvements upon the methodology. The usefulness of the results is limited, however, by the imprecise quality of the data they use.

Jones and Sung note the strengths of the random utility model but point out that many issues need to be resolved concerning the correct specification of the model and the sensitivity of welfare estimation to specification errors.

Jones and Sung's improvements upon the standard methodology presented in the literature include the development of a consumer surplus measure that can accommodate changes in the predicted number of total trips resulting from policy changes. The standard methodology requires an assumption that the total trips made remain unchanged once the proposed policy is adopted. Jones and Sung also improve upon the treatment of the opportunity costs of time, which has been inadequate in previous studies.

Due to limited information on the frequency of participation, consisting only of the date of an individual's most recent trip and the survey return date, the authors had to develop a stochastic renewal model to infer the total trips made per season by all Michigan anglers. Although necessitated by their data limitation, the model allowed them to assess the dependency of a trip choice on the duration of the trip, which is lacking in many random utility model studies that restrict their analysis to day trips. Jones and Sung, by showing that two-thirds of the damages in their policy scenarios accrue to anglers taking trips of longer than one day, conclude that ignoring this relationship of trip choice on trip duration may lead to severe underestimation of damages.

...two-thirds of the damages...accrue to anglers taking trips of longer than one day...

Because the stochastic renewal model poorly matched the total trip days indicated by another less comprehensive survey, the authors issue a warning, stating that the total benefits or total damages they estimate "should not be treated literally." They do, however, say that the consumer welfare measures calculated on a per trip basis may be cited. The consumer welfare measures are reported for the first policy scenario, which is the termination of fish mortalities caused by the pump-storage power plant. For this scenario, which is expected to lead to a roughly 10% increase in all salmon catch rates, they calculate an average conditional compensating variation of $.12 per trip when considering all of Michigan. If restricted to those areas and anglers most affected by the power plant scenario, the compensating variation is $1.08 (1984 $) per trip. If the increase in catch rates were 20%, which is similar to the scenarios of other studies [Bockstael et al. (1988, 20%), Smith and Palmquist (1988, 25%)], the average compensating variation would be $2.92 per trip, which is of the same order of magnitude obtained in these and other studies. No similar comparison is made for the PCB cleanup scenario, although it appears that the compensating variations are roughly the same. Finally, the model can be manipulated to estimate the value of changing the fish populations over a given number of stream miles. Given a decrease in fish kills from the pump-storage facility that would increase fish populations over 100 stream miles, the average benefit is $0.04 per mile affected per trip.

4. ACID DEPOSITION-RECREATION PATHWAY STUDIES

We have examined six studies of this pathway. Two of the studies also examine nonuse values. Four apply to the Adirondacks [Englin et al. (1991) applies to all the Northeast], one applies to Minnesota, and another applies to Norway. The methods used span the full range. The scenarios are all for large changes in emissions or changes in a variety of impact measures or proxies (environmental quality index). Values per person (or angler) per year vary over a wide range.

Englin et al. (1991) is by far the most complete. Like most studies of acid rain damages to lakes, the Englin et al. report for NAPAP focuses on the Adirondacks. This region has the best documented evidence of damage. The study takes information on trips made by anglers from Maine, New Hampshire, Vermont, and New York and produces estimates of social welfare for a variety of scenarios. Anglers from New York City were excluded from the analysis.

Damages for eight scenarios were estimated: the standard NAPAP emission projections of S1 (no additional sulfur controls beyond those already legislated) and S4 (reduction of 10 million tons) in both 2010 and 2030, a 30% increase in deposition in the year 2030, no change by the year 2030, a 50% reduction in deposition, and current damages. By the year 2030, S1 and S4 will achieve just about the same level of control, with S1 retaining slightly higher levels of emissions.

The Englin et al. report makes all of the linkages back to deposition. Deposition is linked to an acidic stress index (ASI) for fish through calculations based on measured chemical characteristics of lakes, where possible, and through forecasts based on other observed characteristics of lakes where the chemical characteristics information is not available. The equation using the chemical characteristics was based on laboratory toxicity experiments on fish. Survival of fish fry is related to the pH, calcium concentration, and aluminum concentration of the lake. For lakes where this information was not available, a regressions was fit for characteristics of the lake that could be observed by the anglers. These nonchemical characteristics used to predict ASI were state, pond vs lake, percentage of watershed in leafy trees, percentage of watershed in pine trees, percentage of watershed in meadows, percentage of watershed that is agricultural land, subjective description of the weediness of the lake, visibility, and whether boating or swimming was included on the trip.

The Englin et al. report makes all of the linkages back to deposition.

Similarly, Englin used a regression to estimate the relationship between catch per hour and the biological response of fish to changes in water quality. Using actual average catch per hour as a base, the percentage change in ASI was used to calculate future catch per unit effort for each scenario.

Two methods were used to value the change in catch per unit effort: a hedonic travel cost model (HTC), and a random utility model (RUM). The effect of the change in catch per unit effort (CPUE) on participation in recreational fishing is also modelled. The participation model is somewhat problematic. It not only includes catch per unit effort for both bass and trout, but a range of demographic information as well. There is not enough information given in the report to isolate the effect of the acid rain from the changing population.

The WTP per person per year is quite different according to the method used. For instance, with no reduction in deposition from 1989 levels, WTP is \$1.54 using the hedonic travel cost approach, but only \$0.30 using the random utility model. The relationship between results turns completely around for the scenario involving a reduction in deposition of 50%. In this case, the hedonic model reveals WTP of \$0.24 while the RUM estimates WTP of \$0.82.

Mullen and Menz (1985) use data from the 1976 New York State Anglers' Survey to estimate a relationship between an individual's fishing days and site availability, which is measured as the sum of acres of fishable water (for four different kinds of fishing), weighted by the travel cost from each site (or collection of sites) to the individual's residence. This is converted to a demand function by adding a price term to the weights and using the estimated relationship to determine changes in visitation as a function of the price. In the next step they remove from the available fishable waters all sites where pH has fallen below 5 in measurements made since 1976, according to surveys of Adirondack lakes made since 1976. A new demand curve is calculated, and the economic losses due to acidification are estimated by the area between the two curves.

This analysis is not very useful for making estimates of external costs, although in fairness to the authors it should be noted that this was not their purpose. The analysis of Mullen and Menz has been vigorously criticized by Shaw (1989), who argues that their demand curve is not consistently derived from economic theory and is not really a demand curve at all. However, we think this argument is of secondary importance in evaluating the use of Mullen and Menz for purposes of making external cost estimates. Far more important are the authors' failure to make a linkage back to emissions and their inattention to the sunk cost problem. (1) There is no observed connection between physical damages and recreation behavior. Mullen and Menz assume a decline in recreation visits is associated with acidification; they do not observe it. Of the lakes removed from the supply of fishable water, it does not appear from their article that they can assert that these lakes are not being fished today or that they were being fished prior to 1976. (2) Mullen and Menz value the damages that they believe have already occurred to Adirondack lakes. As explained above, a far more useful concept is the valuation of lakes that would not become acidic if emissions were reduced.

Violette (1985) applies two models, a travel cost model with site characteristics and a simpler participation model, to fishing sites in the Adirondack Mountains of New York. This method only considers use values, although Violette notes that the nonuse values may exceed the use values. Acid rain was assumed to affect both the number of sites available for fishing and the characteristics of the sites. Only lakes are included in the analysis. Sites were characterized by the number of lakes they contained with certain characteristics, for instance acres of cold water, and two-story or warm water lakes. Site characteristics were obtained from the Adirondack Lake and Pond Survey. The economic damages were calculated by changing the characteristics of the sites, for instance by changing the fishable acres of cold water lakes. The characteristics did not include catch rates. For the fishermen, Violette uses the same data set as Mullen and Menz, the New York Angler's Survey for 1976-77. The survey was mailed to a 3 percent sample of fishermen licensed in New York State. It asked questions on fishing activities, expenditures, preferences, attitudes and opinions, and participant background. Violette designed the model to yield an upper-bound estimate for damages. For instance, he did not take substitutability of fishing sites into account. The estimates of damages from the travel cost model ($0.8 to $11.6 million) and the participation model ($1.7 to $10.5 million) are about the same.

Like Mullen and Menz, Violette's work appears to us to be interesting and competently done. However it is not useful for estimation of external cost because it too does not link his estimates of economic damages either to acid deposition in the region or to emissions. In addition, the Lake and Pond Survey is not a very comprehensive or up-to-date data set. Not all lakes and ponds are surveyed each year or necessarily within the last 20 years. Data on pH existed for only 35% of measured surface area, and information on alkalinity was available for only 52%. This lack of information is a problem since what the Lake and Pond Survey records as the state of the lake may have no relationship to its actual state at the time anglers were making a recreation decision.

Morey and Shaw, using the 1976-77 New York Anglers Survey — the same data as Violette and Mullen and Menz — estimate a share model for seven lakes or sets of lakes in the Adirondacks and for eight varieties of sport fish. Each fisherman allocates a fixed recreational budget to alternative sites, and the model estimates the effect of site characteristics on shares. The site characteristics Morey and Shaw consider are fishable acreage and average catch rate for the individual's first and second most preferred species at the site. Morey and Shaw estimate the willingness to pay for a 25% increase in catch rate per season at $8.68 per fisherman; for a 50 percent increase, $15.61.

Morey and Shaw did not link acidity to fish populations or fish populations to catch rates, so the study is only useful as a demonstration of the sensitivity of benefit estimates to catch rates. Because there is no link between physical and economic damages, the study does not in and of itself produce useable results for environmental costing. Once such links are available, however, this study may become quite valuable despite its limitations. One such limitation is the assumption of a fixed recreation budget. This is bound to underestimate benefits of improvements, owing to substitution between fishing and other activities. However, the effect is not likely to be large. The approach also assumes an exogenous total number of recreation trips.

Welle (1985), calculates existence values and use values for the environmental quality of lakes in Minnesota, which may be affected by acid rain.[4] About 1,000 Minnesota residents were asked how much they would be willing to pay to prevent declines in the environmental quality of Minnesota lakes from acid rain. The changes in environmental quality were presented as

[4]NAPAP, p. 6-34.

movements on an "environmental quality ladder." In addition, they were asked to characterize themselves as users or nonusers, and the bids of nonusers were characterized as existence values by Welle. When including protest bids of $0, the mean bids for nonusers ranged from $30 to $36 depending on the degree of damage, and for certain users the bids were $91 to $109. Excluding protest bids, the mean nonuser bid was $57 for both moderate and severe effects; the mean certain user bids were $102 for moderate effects, and $124 for severe effects.[5] The range of damages was unspecified in NAPAP's summary of the report. NAPAP described the study as limited in terms of the population and the scope of the effects of acid deposition to which they can be applied.[6] In addition, the questions asked in this study were only loosely tied to theory, so the existence values calculated may not be near the true existence values.

Navrud's (1989) study of willingness to pay of households in Norway for incremental improvements in freshwater fish populations calculated median nonuse values of $12 to $36 per fish per household per year. The author asked respondents to indicate the maximum amount of money they would pay for improvements from liming, and to divide their total bid into use, option, and existence and bequest values. The improvements were equal to the expected effects of 30%, 50%, and 70% reductions in sulphur emissions in Europe, although the empirical basis for this linkage is not discussed, nor are any literature citations provided. Median WTP for the 30%-equivalent reduction was 100 to 300 Kroner, depending on starting bid (i.e., the experiment suffered from mild starting-point bias). This translates into a range of $16 to $48 at the exchange rate in effect in April 1988, when the survey was conducted.[7] Respondents reported 12 percent of WTP to be recreational value, 12 percent option value, and 76 percent existence value.

Assuming Americans have the same preferences as Norwegians, it would seem that these results can be used in environmental costing applications in the United States. However, two vital pieces of information are still needed. First, we need to know how many people should be included in the population valuing the resource. Second, we still need the "transfer coefficient" relating changes in emissions to changes in water quality. The link between water quality improvements and the 30, 50, and 70 percent emission reductions assumed by Navrud are not justified in the paper, and even if they can be justified they relate only to European, not American, experience. Corresponding emission reductions in the U.S. are likely to be quite different.

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[5] Welle, p. 180.

[6] NAPAP, p. 6-34.

[7] Purchasing power parity would provide a more meaningful comparison but is not as readily available.

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PAPER NO. 14*

CALCULATING EXTERNALITIES FROM DAMAGES IN OCCUPATIONAL HEALTH AND SAFETY

This paper surveys the theoretical basis for the possibility that coal miner occupational health and safety damages are not adequately internalized into the production cost of mining coal and thereby impose an external cost on society.

There are primarily two mechanisms for occupational health and safety damages to be internalized into the firm's cost of production. One is through wages and benefits, and the second is through employer compliance with government programs, such as the Federal Black Lung Program and State Workers' Compensation Programs. These two mechanisms are evaluated in the next two sections.

...two mechanisms for occupational health and safety damages to be internalized...wages and benefits...compliance with government programs...

In the final section, the pertinent issues are summarized and a tentative assessment of the success of these mechanisms for internalizing expected damages is offered. This assessment concludes that these mechanisms are largely but not entirely successful, and the claim that occupational health and safety issues in the mining of coal imposes some external costs on society may have validity. However, a quantitative assessment of this cost is impossible given the information that currently exists. The most that we can accomplish is to organize information with regard to this issue for consideration along side quantitative values presented to policymakers.

Compensation Through Wages and Benefits

To varying degrees all employment opportunities include a risk, or expected damage, of harm to the health and safety of workers. One way that such damages can be internalized into the costs of production is through

*Based largely on a working paper by Dallas Burtraw and Jonathan Shefftz.

increased wages and benefits.[1] Benefits may take the form of reduced hours, workplace amenities, nonpecuniary compensation such as pensions and health services, employer contributions to union or government funds to deliver similar services, etc.[2] The economic theory of "compensating wage differentials" suggests in a competitive labor market that wages and benefits, taken together, will adjust so that at the margin workers are indifferent between more risky jobs and associated higher wages or benefits, and less risky jobs that will have lower wages or benefits.[3]

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One finding of research in this area is that workers sort themselves according to risk across industries and occupations, depending on their own attitudes towards risk. A second finding is that within each risk classification, compensating wage differentials for a wage-fatality risk tradeoff have been identified and measured across a variety of industries.[4] For example, Viscusi

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[1]Another avenue for internalization of costs is through employer liability. If employers must fully compensate an injured worker, or his or her family in the case of loss of life, so that the injured parties are "made whole," then there is no external cost. As made clear in a subsequent section, State Workers Compensation Programs effectively preclude this possibility by capping employer liability. Yet another avenue is through direct payments to the government in the form of fines, taxes, etc. Employer contribution to the Black Lung Programs and State Worker Compensation Programs are explored below.

[2]The National Bituminous Coal Wage Agreement of 1988 describes nonwage aspects of compensation.

[3]See Rosen (1986); Moore and Viscusi (1990). Rosen (1986) characterizes the market wage as the "sum of two conceptually distinct transactions, one for labor services and worker characteristics, and another for job attributes. The positive price the worker pays for preferred job activities is subtracted from the wage payment. The price paid by employers to induce workers to undertake onerous tasks take the form of a wage premium, a negative price for the job, as it were. The observed distribution of wages clears both markets over all worker characteristics and job attributes."

[4]"It is only fair to note, however, that these results do not prove that risk is allocated efficiently by the labor market. Rather, they indicate that complete market failure does not necessary exist. Any call for intervention must therefore be based on subjective judgments of the adequacy of the compensating differential and self-sorting market mechanisms," (Moore and Viscusi 1990, p. 15). See also Dillingham (1985) and Dickens (1990) for more general critiques.

(1979) found that within a sample of blue-collar workers, market-generated compensation for fatality risk averages almost $1,000.[5]

Unfortunately, this empirical literature has addressed the converse of the question that we address in this paper. While the empirical literature has been concerned with demonstrating the existence of a positive compensating wage differential, we are concerned with the degree to which wage differentials fail to compensate *adequately* for damage. In terms of the familiar metaphor of a half full/half empty glass of water, the literature has measured how far the glass is from empty. Our purpose is to assess how far it is from full. In the absence of empirical evidence on this question, we evaluate the coal mining industry in light of the theory of compensating wages to assess qualitatively the evidence for and against the internalization of damages.

A sufficient condition for the internalization of damages into worker compensation is the existence of a competitive labor market.[6] Four conditions particular to the coal mining industry shed doubt on the existence of a competitive labor market. One is the special sociology of the coal industry, in and around the Appalachian region in particular. The mines that serve as reference environments for this study are located in Daviess County in western Kentucky, on the periphery of Appalachia, and in northwest New Mexico. Many of the sociological characteristics that we discuss that pertain to Appalachia in particular will not apply strongly to these areas, but due to the strong occupational identity in the coal industry, many will be relevant.

A second condition that undermines the assumption of a competitive labor market is the difficulty workers may have in accurately perceiving their true risks. It is often asserted that workers' perceived risk is less than the actual risk, although in principle the effect could work in the opposite direction, with workers overestimating actual risk. A third condition is the monopsony influence of a single high-wage industry in the region. These three factors combine to suggest that workers could be at a disadvantage in a bargaining relationship with employers leading to an increase in occupational hazards. Kahn (1991) has confirmed this hypothesis with regard to the monopsonistic influence of the coal industry in Kentucky and corresponding safety records among nonunion mines. A fourth condition is the presence of organized labor

[5]This number is in 1988 dollars. The level of risk varies. See Moore and Viscusi (1990) for a comprehensive review.

[6]This condition is not necessary due to the possibility of countervailing influences in the labor market, such as two-sided market power on the part of a monopsony employer and of a monopoly labor union. We consider these offsetting influences below.

unions, which may be a countervailing force with regard to worker bargaining power. We consider each of these factors in turn.[7]

The Sociology of the Coal Industry and Worker Mobility

A thorough review of sociological findings on the Appalachian coal region is impossible in this brief presentation, so we focus on the most pertinent findings and refer the reader to other sources for more careful review. Two important features of the sociology of Appalachia are the prevalence of poverty in the region and relatively low educational attainment.[8] These characteristics are evidence that workers in the region have relatively few skills that are transferable to alternative employment inside Appalachia or in other regions.

Another special feature sometimes attributed to Appalachia residents is a high cost of geographic mobility.[9] Factors that have been identified as important to the willingness to consider both locational and occupational changes include the number of children in school, job and career tenure, income, housing demand and resident site inducements and other psychic costs, family health, etc.[10] Poverty and low educational attainment could be linked to immobility if social structures emerge and play a particularly important role. For example, nonmarket activities within the extended family may substitute for market activities among families with higher incomes. This characteristic may be summarized as "deep roots in the community." Reliance on nonmarket activities to meet economic and social needs raises the relative cost of relocation

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[7]For a general exposition of these issues see Gramlich (1981, chapter 11).

[8]In 1988, 12.1% of households participated in food stamp programs in Kentucky (fourth ranking nationally) and 9.1% in New Mexico (thirteenth ranking nationally). The national average was 7.7%. Average annual pay in Kentucky was $18,545 (twelfth ranking nationally) and $18,259 in New Mexico (eleventh ranking nationally). The national average was $21,871. (U.S. Department of Commerce 1990.)
In one study of miners in Southern Appalachia, the education median for miners was 12.0 years for men 34 years or less, and 8.7 years of school for men 35 or older. "Affirmation of education is improbable except for the child who shows extraordinary talent with "books" at school. Therefore, a consequence of such discouragement is *temporary delay, not permanent denial, or entry into the mine industry.*" (Althouse 1974, p. 26.) The difference among age groups in the study, and the age of this study suggest that this characteristic may be of diminishing relevance.

[9]Myers (1983) analyzed the period from 1973-1979 when U.S. coal production increased 17%. In Eastern (primarily Appalachian) coal counties employment growth rates exceeded population growth rates by a factor of four in this period suggesting most new jobs were filled by people already living there. In contrast, in the West (including New Mexico) and in the Interior (including Daviess County, Kentucky) employment growth was only twice that of population growth, suggesting many new jobs were filled by workers moving into coal counties.

[10]Linneman and Graves (1983).

to seek improved employment because of the cost of severing those roots.[11] In addition, the perceived cost or mobility may be high due to the lack of role models and contacts who have migrated previously.[12]

In sum, the combined sociological characteristics of relative poverty, low educational attainment, and high costs of mobility among coal miners may impart a bargaining advantage to employers.

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Perceptions of Occupational Risks

A second factor that contradicts the requirements of a competitive labor market is the potential inability of workers to perceive accurately their occupational hazards. There is ample psychological evidence that individuals fail to perceive risk in an accurate manner.[13] However, Robinson (1989) analyzes worker responses to occupational exposure to carcinogenic substances in terms of subjective assessment of risk, quit rates, and prounion attitudes. The results of his analysis indicate that "workers recognize at least some of the cancer risks they face on the job."

Dickens (1990) criticizes empirical studies of compensating wages on the grounds that workers simply ignore altogether factors which have a low

[11]The cost of severing one's ties to a community are not only personal but also spillover onto the community. For practical reasons we ignore this spillover, manifest in the willingness to pay of a family's neighbors to keep the family from leaving the area. However, to the extent that individuals are altruistic and internalize this effect already into personal decisions, and to the extent the region is characterized by extended families and communities, this spillover would help to explain the relatively high costs of mobility that are attributed to the region.

[12]Althouse (1974) reports that in a sample of miners in Southern Appalachia, 50 percent of all brothers of the sample are themselves miners (p. 24). Althouse reports a "folklore" of occupational immobility that is reinforced by black lung, represented in the quotation from a miner: "after ten years of mining a man can't pass the physical for another kind of job," (p.70). Fewer than 5 percent of all brothers in the sample are reported in any sort of business or professional occupations (p. 23).

"...(I)f one's male siblings are not mining, they probably migrated to blue-collar jobs out of state," (p. 24). "To the extent that the youths in the coal fields have kinfolk in urban places, an opportunity to acquire a desired occupation outside a situation of highly limited job choice may exist. Migration is often contingent upon the formation of kinfolk networks. The chance to move to a branch of them elsewhere can spur migration of some youths from the groups at home. But as long as job opportunities at home remain 'good' many, if not most, continue to enter the mines. Mining is the center of gravity in their style of life; it is 'in the blood' though they deny that mining is their chosen occupation," (Althouse 1974, p.26).

[13]For examples, see the symposium in *Science*, Vol. 236, April 17, 1987.

probability. This suggests that workers may underestimate risks such as mine accidents. However, in comparison with other industries the dangers associated with coal mining are not low probability events, and there is strong evidence that workers are cognizant of the risks in the industry.

On the other hand, it takes approximately thirty years for black lung disease to develop and such a large chronological gap between exposure and disease may lead workers to discount their personal probability of contracting the disease. In addition, legislation to promote safer conditions for coal mining may lull workers into a false sense of security, if recent evidence of widespread fraud and tampering with dust samples is accurate.[14]

However, in the absence of further evidence, the argument that workers underestimate occupational risks seems less compelling in coal mining than in other industries. Furthermore, given significant improvements in working conditions throughout the industry compared to a generation ago, the likelihood that workers underestimate true risks today is not any more compelling than the likelihood that workers overestimate true risks.

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Monopsonistic Influences in the Labor Market

A third factor that is cited as evidence against the existence of a competitive labor market in Appalachia is the narrow base for relatively high wage employment, such as is associated with coal mining.[15] The relative concentration of employment opportunities among few industries and employers would suggest that employers enjoy relative bargaining strength *vis-a-vis* their workers.

Trends in the mining industry over the last decade reinforce this possibility. There has been a long slide in coal employment after World War

[14]References on the dust fraud issue include: Kilborn (1991), U.S. Senate Committee on Labor and Human Resources (1987), McAteer (1991), Occupational Health and Safety Center (1991), and Stoltzfus (1991).

[15]There are "few occupational alternatives to coal mining which promise steady employment, and any amount of occupational diversity is negligible throughout the area. To a considerable degree, if a man doesn't work coal, he many not work regularly at all. Furthermore, a large share of work other than coal industry jobs in these counties is normally obtained on a patronage basis." Other jobs (construction, timber) are seasonal, or "closed along class lines," (Althouse 1974, p. 23).

II that is still continuing.[16] The employment increase of the 1970s was an exception to the continuing decline in coal employment. The recent increase in productivity in the industry is in part a trend toward mechanized longwall technology, which is less labor intensive than underground mining. Job loss between 1978 and 1984 was 31% in the industry.[17] Western Kentucky, which produces mostly high sulfur coal, has suffered a declining share of Kentucky's total output, from 39% in 1975 to 27% in 1984. Low sulfur coal in the eastern part of the State has driven recent increases in production. The coal industry as a whole will grow 3% per year between 1984 and 1995, but this does not imply a growth in employment.[18]

Monopolistic Influences in the Labor Market—Organized Labor

A fourth condition in the coal mining industry that sheds doubt on the existence of a competitive labor market is the role of organized labor. Typically organized labor may possess market power in the provision of labor services. In this specific industry it appears organized labor may be a countervailing force to the power of employers with regard to worker bargaining power with a predicted increase in safety. Approximately 42.3% of the labor force in coal mining is unionized nationally. Approximately 43.3% have contract coverage. Among all workers in Kentucky about 15.8% are union members, while in New Mexico about 7.2% are union members.[19] This geographic pattern applies to the coal industry as well, with greater percentages of workers unionized in eastern mines.

The standard daily wage rates in 1990, according to the National Bituminous Coal Wage Agreement of 1988, range from $115 to $132, corresponding to annual salaries of $30,000 to $34,400. Nonunion salaries are expected to be less, though we expect them to be influenced by the organizing activities of the union.[20]

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[16]Mining employment in the U.S. dropped from 415,600 in 1950 to 177,800 in 1984. In Kentucky the numbers dropped from 76,000 in 1950 to 34,400 in 1987. If Kentucky's coal employment were measured in full-time equivalent jobs, the 1987 figure would be 24,800. (MACED 1987, p. 20.)

[17]Kentucky coal jobs declined by 27% in this period. MACED, *Op. cit.)*

[18]MACED 1987.

[19]Curme, *et. al.* 1990.

[20]Hourly wages plus overtime and bonuses at larger mines have risen to the point that many miners make over $40,000/year, with some earning as much as $70,000/year. Wage levels this high may not be sustainable when competitors at smaller mines are paying $6.00 to $8.00 per hour. Some large mines have laid off employees and then have contracted out mine operations to companies which pay no benefits and half or less of the wages previously earned at the same site. (MACED 1987, p. 20.)

Extrapolating from existing circumstances one can reason that the mine in the New Mexico reference environment will not be a union while the mine in Daviess County, Kentucky will be union. It is noteworthy that nonunion miners (typically workers at smaller mines) are three times more likely to be killed on the job than are union members.[21] However, it does not seem appropriate to consider the role of the union in isolation from the performance record of the industry in general. For example, companies that run union mines also sublease portions of their properties, especially after productivity has begun to fall off, to smaller and typically nonunion companies to intensively work the remains.[22]

There are several plausible explanations for the observed correlation between unions and safety. One is that the union effect may simply be capturing the safety record of larger mines, which are typically more unionized, as opposed to smaller (nonunion) mines.[23] A second explanation for the differential in fatality rates between small and large mines (and therefore between unionized and nonunion mines) can be found in the incentives created by State workers' compensation plans. Larger mines are more likely to be experience rated in the insurance premiums they must pay for workers'

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[21]"Last year, the rate of fatal injuries for all regular coal mine employees was just about .03 per 100 workers, a record low. Independent contractor employees, however, were killed at a rate of .11 per 100 workers. The independent contractor employees' rate more than tripled that of regular miners, and the contractors, for the most part, work on the surface - a relatively low-hazard area," (O'Neal 1989).

The United Mine Workers Journal of January 1986, reports that through September 25 of 1985, "approximately 70 percent of all fatal coal-mine accidents in the United States occurred at mine employing fewer than 50 workers. Nearly half of all coal-mine fatalities occurred at mines employing 20 or fewer workers."

The opening statement of Senator Howard Metzenbaum on March 12, 1987, before the U.S. Senate Committee on Labor and Human Resources Full Oversight Hearings of the Mine Safety and Health Administration provides further evidence:

"The GAO [General Accounting Office] continuing investigation also uncovered a dramatic disparity in injury reporting between small mines - those employing fewer than 50 workers - and large mines. The GAO figures are startling. Small mines employed only 5% of the mine workforce, but accounted for 45 percent of mine fatalities.

The high fatality rate compares to an unbelievably low rate of reported injuries. Small mines on average reported only 3 injuries for every fatality, whereas large mines reported 51 injuries for every fatality, which is 17 times the injury report rate of small mines."

[22]Yarrow (1990, p. 40).

[23]Most coal mines that employ less than 50 miners are non- union, according to the United Mine Workers Journal of January 1985, meaning that the union/non-union differential may simply be capturing the effect of the presence of a disproportionately large number of small mines in the nonunion sector, not the actual safety effect (if any) of unionization. This is supported by the National Research Council (1982), which shows that their measured gap between overall union/non-union fatality rates (0.11 vs. 0.06 average number of fatalities per 200,000 employee-hours) essentially disappears if calculated for each size category (0.14 for both union and nonunion mines with 1-50 employees, 0.08 both with 51-150, 0.09 for nonunion and 0.07 for union mines with 151-250). Union mines do have a higher disabling injury rate, but this appears to be merely a reporting effect (and interpretable as a sign of nonunion mines' skirting of the law).

compensation plans, providing them with greater incentive to take care to avoid accidents (see the section on State Workers' Compensation Programs below).

Notwithstanding these alternative explanations, the correlation between large mines, union mines, better safety and higher wages erodes the compensating wage hypothesis Obviously this correlation is confounded by the union wage differential, and by technologies that exist in the larger mines. Nonetheless, the correlation sheds doubt on the assumption of a competitive labor market.

Trends in the industry appear to be undermining the influence of organized labor. In 1978, about 250,000 coal miners were employed and about 160,000 were union members. In 1990 employment had fallen to 150,000, of whom only 70,000 were union members.[24] Most of the contraction occurred in the union mines.

In summary, potentially high mobility costs, potential information problems, and monopsonistic demand for labor services in the region combine to empower employers in the coal industry. Organized labor functions to offset this power. The relative strength of these influences is open to speculation and controversy. If on balance these costs constitute a disadvantage for labor in a bargaining framework over wages and working conditions, then the wages and benefits component of the firms cost of production will inadequately reflect expected health and safety damages.[25]

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Employer Contribution to Government Programs

Compensation for workers is not the only manner in which damages may be internalized into the cost of production in the coal industry. Employer contribution to special government benefit programs are another manner in which damages may be internalized. There are two general frameworks which

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[24]Yarrow (1990).

[25]If market power is attributed to the employer, then whether it is health and safety damages or body sweat that is not being compensated, or a share of each, is an open but irrelevant question.

apply, federal and state black lung benefits and state workers' compensation programs.

Black Lung Benefits[26]

New claims under the federal Black Lung Benefits Program are administered under the Department of Labor.[27] The program provides monthly payments and medical treatment benefits to coal miners (and certain transportation and construction workers who were exposed to coal dust) totally disabled from black lung. Monthly payments to eligible surviving dependents are also provided if the death of the deceased claimant is proved to be due largely to black lung.[28]

A number of features of the program diminishes its ability to capture all expected Black Lung damages.

A number of features of the program diminishes its ability to capture all expected Black Lung damages. One is the limit on benefits outlined above. Current rates are \$387.10 per month for an individual miner or the surviving spouse of a deceased miner, and up to \$774.10 per month for a claimant (miner or surviving spouse) with three or more dependents.[29] Unlike most workers' compensation programs, the benefits are not based upon past wages, receipt of

[26]In addition to the sources cited in this section, this information was drawn from the following sources: U.S. Department of Labor, Annual Report 1989; U.S. Department of Labor, Fact Sheet No. ESA 89-14; Field Hearings on Black Lung, 1990; Telephone conversation with Davitt McAteer; Telephone conversations with officials in U.S. Department of Labor: Branch of Workers' Compensation Studies (523-9560); Employment Standards Administration, James Rechnitzer (523-6808); and Division of Coal Mine Workers' Compensation, Dan Peed (523-6792).

[27]The 1969 Federal Coal Mine Health and Safety Act created the Black Lung Benefits Program, assigning initial responsibility for administering the program to the Social Security Administration. In 1973, the Department of Labor assumed responsibility for the administration of the program. However, the Social Security Administration still pays and administers claims filed prior to July 1, 1973, as well as the claims filed by the survivors of the original claimants. (The payments are for income benefits only, as it has never paid medical benefits, unlike the Department of Labor.) The number of beneficiaries and the total monthly amount of benefits processed by the Social Security Administration only (presently 174,268 and \$72 million) is gradually declining. (Not a single new claim was filed with the Social Security Administration in 1989, while the number of present beneficiaries is declining due to death.)

[28]However, in the case of a deceased claimant who originally filed the claim before 1982, the death may be due to any cause, even totally unrelated to black lung.

[29]These amounts are based upon 37.5% of the pay rate for Federal employees in GS-2, step 1.

benefits does not preclude employment lighter than coal mining or its equivalent, and retirement does not preclude receipt of benefits.[30]

Second, coal mine industry pension plans typically have clauses which offset the pension by the amount of Federal black lung benefits, especially if the benefits are being paid by the responsible mine operator.

Third, in order for a claim to be approved, the disability must be total.[31] This is in contrast to State workers' compensation programs, where workers can receive payment for a permanent partial disability which allows continued employment but nevertheless causes a loss of physical function. Total disability for the purposes of the Black Lung Benefits Act means that the claimant is no longer able to perform the equivalent of the types of tasks previously performed, either the specific jobs actually performed during the period of employment in coal mining industry or the physically equivalent jobs in other industries.[32]

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[30]These benefits are offset by a State or Federal workers' compensation award received for black lung. Federal black lung benefits are also reduced when a miner or surviving parent, brother or sister has earnings in excess of a maximum allowable level. Since by definition beneficiaries must be totally disabled from coal mine employment or its equivalent in terms of exertion, and since beneficiaries are typically of retirement age, only 25 claims are currently being offset due to excess earnings.

Combined State and Federal workers' compensation offsets amount to approximately $5.9 million a year. Currently 782 claims are being offset because of concurrent Federal benefits. These are mostly claims filed by former inspectors of the Federal Mine Safety and Health Administration who have contracted black lung from their frequent inspections of mines. As Federal employees, they are also eligible for benefits under the Federal Employees' Compensation Act, which offset their benefits received from the Black Lung Benefits Program.

Currently 3097 claims are being offset because of concurrent State benefits. The original plan at the time of the creation of the Federal program was to provide benefits only until the State programs caught up in compensation levels, and then to refuse to accept further claims from those States which were now certified as providing adequate benefits. In the major coal producing States, such as Pennsylvania, Kentucky, and West Virginia, black lung is by law a compensable disease under worker's compensation programs. However, in other States it is not and/or widows' benefits are not up to Federal levels (often being merely a lump-sum payment). Therefore, no State has ever been certified as meeting Federal standards, nor does it look as if any State ever will be, thus ensuring the Federal government's continuing role. State workers' compensation programs usually set their income benefits at two-thirds of the individual's former wages, in contrast to the Federal black lung program, which is a fixed amount. However, State compensation laws also dictate maximum and minimum benefit levels for permanently totally disabled workers. In 1989, all the top twenty-five coal producing States had maximums larger than the Federal black lung benefits level, while twelve of the States had minimums lower than the Federal level.

[31]Eligibility standards for the receipt of benefits were tightened significantly by the Black Lung Amendments of 1981. Essentially, under the present 718 series regulations, the burden is on the claimant to establish total disability due to black lung, whereas previously the 727 series regulations made rather liberal presumptions for the presence of black lung. Furthermore, the medical guidelines for the establishment of total disability are much more strict under the present 718 series regulation than under the previous 727 series regulation.

[32]However, claimants may still be capable of performing—and may actually be performing—lighter work, although this is seldom the case as the majority are of retirement age.

Fourth, the totally disabling respiratory impairment must have arisen solely from coal miners' black lung, and not from a combination of black lung and other factors, such as smoking or asthma. Since the medical profession generally agrees that the proportion of a miner's respiratory disease resulting from black lung can not be determined, it is quite easy for a mine operator facing a claim to find at least one doctor to testify that the miner's respiratory ailment is not solely caused by black lung.[33] Therefore, the combination of a totally disabling respiratory impairment and a lifetime of work in the coal mines does not constitute sufficient proof for the receipt of Federally mandated benefits. The vast majority of claim denials is based on the claimant's inability to prove total disability due to coal miners' black lung or (for survivors) the miners' death due to black lung.

Presently mine operators dispute around 90% of the claims lodged against them. The approval rate in 1989 for claims initially filed after 1981 was only 3.4%. This compares to a 47% approval rate from 1973 through 1989 for claims originally filed between 1973 and 1978 and 33% approval rate for claims originally filed between 1978 and 1980.[34]

A fifth feature of the program that diminishes its ability to capture damages is the difficulty for workers to obtain legal representation. The Federally mandated legal fee is only $1,500 to $3,000, which is received only if and when the claim is approved. For an eventually successful case, the attorney may represent the client (who is barred from providing direct compensation to the attorney) for years without receiving any payment. For an unsuccessful case, the attorney may put in years of work with no compensation at all.[35] Obviously, such a situation makes it very difficult for a potential claimant to obtain proper legal representation.

A sixth obstacle to compensation is the Department of Labor's practice of routinely demanding the return of benefits awarded for claims which were later disallowed. Totally disabled former miners have received interim benefits before their disability could be proved to be solely caused by black lung before

[33]Many X-ray examinations may be ordered, and each X-ray may be reinterpreted many different times by many different doctors. As only one negative opinion is required (even after numerous positive findings) and coal mine operators obviously have deeper pockets than any individual miner, mine operators have been very successful in dismissals of claims. (Field Hearings on Black Lung 1990.)

Perhaps appropriately from the standpoint of economic theory, mine operators may also introduce evidence regarding the claimant's cigarette smoking habits, for example, in order to show that other known causes for the claimant's respirator ailment exist. (See Mohring and Boyd 1971.)

[34]It should be noted, however, the approval rate for the first two years of the program, 1970 and 1971, when it was first administered by the Social Security Administration, was in the single digits.

[35]Field Hearings on Black Lung 1990.

a formal hearing. Upon the denial of the claim due to the efforts of the responsible mine operator in gathering countervailing evidence in the interim period, the now denied claimant is served a notice that he owes tens of thousands of dollars payable within thirty days after which interest will begin to accumulate.[36] The total amount collected from both overpaid claimants/beneficiaries and from providers of medical goods and services was $4.7 million in 1989, with an additional $1 million committed to future repayment.

A seventh reason that the Black Lungs Program may fail to internalize all of the expected damages is that historically neither the mine owner nor the industry absorb all the cost of benefits awards. The Department of Labor black lung benefits are provided by the "responsible coal mine operator," defined as the last coal mine employer of at least one year's duration. Another sizable amount of the benefits is provided by the Black Lung Disability Trust Fund.[37] The Trust Fund finances the cost of black lung claims filed with the Department of Labor if the responsible coal mine operator cannot be identified, typically because of bankruptcy.[38] The major source of revenue for the Trust Fund is an excise tax on mined coal sold or used by producers. The tax is imposed on a per ton basis, with no experience rating for the black lung claims previously filed against the operator. The per ton rate is now $1.10 for underground coal and $0.55 for surface mined coal, with a cap of 4% of sales price.[39] In the past, the Trust Fund has run substantial yearly deficits, leading to the accumulation of a cumulative debt to the U.S. Treasury of over $3 billion at the

[36]Field Hearings on Black Lung 1990.

[37]The Trust Fund was first established by the Black Lung Benefits Revenue Act of 1977 and now jointly administered by the Secretaries of Labor, Treasury and Health and Human Services. As of 1989, only 11,000 claims in the Department of Labor Black Lung Benefits Program were being paid by responsible mine operators while 84,000 were being paid by the Trust Fund (although a small number of those claims paid by the Trust Fund are in interim status and may eventually be transferred to responsible mine operators). In total there were 144,000 beneficiaries in 1989. Of this total, over 49,000 originally filed their claims before July 1, 1973 and are thus concurrently receiving income benefits from the Social Security Administration (which has never provided medical benefits), but have subsequently filed for medical benefits with the Department of Labor and are now receiving only medical benefits from the Trust Fund.

Although far more claims are currently drawn against the Trust Fund than against responsible mine operators, this will soon change. Of the 199 new claims approved in the past fiscal year, only 77 were against the trust fund with the remaining 122 against responsible mine operators.

[38]On the other hand, the Trust Fund also finances claims if the miner's last coal mine employment was before 1970. It also covers the administrative costs of the Black Lung program, which amounted to over $56 million in 1989.

[39]This differential implies a 50% lower probability of contracting black lung in a surface mine, which may or may not be accurate.

end of 1989.[40] In the year 2014, the tax will revert to its original, pre-1981 rates of $0.50 and $0.25, with a 2% cap.

The existence of yearly deficits in the Trust Fund represent a historic public subsidy to the industry.[41] On the other hand, through this subsidy and through the presence of the excise tax, current taxpayers, consumers and producers of coal are paying in part for the coal mining externality from the past. These payments offset liabilities incurred against a previous generation of coal miners. However, the industries debt may be largely irrelevant in a forward-looking assessment of expected damages because they do not necessarily represent the marginal expected damage of new investments in the industry.

[40]In 1989, total Department of Labor Black Lung program obligations (which does not include claims paid by the Social Security Administration responsible mine operators) were $650 million (over $590 million in claims plus almost $60 million in administrative costs), but the final audited Trust Fund collection amount was only $605 million. Nevertheless, this situation represents a vast improvement over the early years of the program, before the existence of the tonnage-supported Trust Fund and the practice of having the last responsible operator pay the claim. Between 1970 and the end of 1984, the federal treasury had paid out more than 600,000 claimants a sum approaching $15.6 billion (over $12 billion by the Social Security Administration and over $3 billion by the Department of Labor) (MACED 1986).

[41]Over $12 billion that has been paid out by the Social Security Administration (which will still be added to at the rate of tens of millions of dollars each year for many more years to come until all recipients die off) and also a large portion of the over $3 billion paid out by the Department of Labor will remain an unpaid debt to society created by the coal industry of the past. It should be noted that the creation of this debt is no accident. During the Congressional debate over the 1969 Federal Coal Mine Health and Safety Act it was recognized that the high cost of a black lung benefits program necessitate by the coal industry's actions in the past would represent a large share of future coal industry revenues. At the same time, the Act was to create significantly strengthened and wide ranging safety regulations for the coal mining industry, including lower coal dust standards. As it was politically impossible to impose both the costs of preventative measures (dust standards) and compensating measures (black lung benefits) on the industry at the same time, the federal government ended up footing the bill for all of the compensation program (MACED 1986).

Eventually the higher tonnage tax levels should close the Trust Fund yearly deficit and perhaps even be able to contribute toward retiring the accumulated Trust Fund debt. Or, more realistically, the tax revenues may be able to pay off the interest charges on the Trust Fund debt, which reached a peak of almost $275 million in 1985 before the Consolidated Omnibus Budget Reconciliation Act of 1985 imposed a five year moratorium on the accrual of interest. The Department of Labor's budget projection model now forecasts continuing Trust Fund insolvency through the year 2030. Earlier forecasts showing solvency by the year 2014 had served as the rationale for Congress's reversion of the coal excise tax in that year to its original levels. (Telephone conversation with Jim Rechnitzer.)

State Workers' Compensation Programs[42]

Workers' compensation programs vary across different states. However, it is possible to provide a generalized picture of the most important components of these programs.[43] Since employers are responsible for benefits paid under these programs they serve to internalize the expected damages of occupational hazards. However, a number of features of these programs may serve to limit the extent to which this internalization actually is achieved.

The employer is liable for the costs of any work-related injury, regardless of who is to blame for the original incident. However, the imposed liability is exclusive, as the injured worker is not allowed to sue for additional compensation (pain and suffering, etc.). Therefore, workers' compensation benefits are a type of no-fault insurance which gives employers limited liability for their agreement to pay certain benefits to injured employees.[44]

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[42]In addition to specific citations, information in this section has been garnered from the following sources: U.S. Department of Labor, October 1990; U.S. Department of Labor, January, 1991; Chelius and Smith 1983; Ehrenberg 1988; and Ruser 1985.

[43]Workers compensation laws are State laws (with Federal Employees' Compensation Act covering Federal employees) that make employers liable for all of an injured worker's medical expenses and a portion of lost wages. For permanent total disability (where an individual is permanently prevented from ever working at all) and temporary total disability (where an individual is temporarily prevented from working at all but expected to recover fully), the income replacement-rate is typically set at two-thirds, constrained by minimum and maximum levels which are often tied to State wage averages. For permanent partial disability (where an individual is permanently physically impaired to some degree short of total disability), scheduled injuries have ex ante fixed benefit levels while nonscheduled injuries may have ex post benefit levels dependent upon a fraction of lost earnings, with the mix between scheduled and nonscheduled injuries varying by State.

In seven States (plus Puerto Rico and the Virgin Islands), employers are required to purchase insurance from a State fund to cover their liabilities. In fourteen States, employers have the choice of purchasing insurance from a competitive State fund or a private carrier. In the remaining twenty-nine States (and the District of Columbia), no State fund exists, and thus insurance must be purchased from a private carrier. In most of these States, self- insurance by the largest firms is also allowed. Insurance through a group of employers is another possible option in some States.

In Kentucky, no State fund exists, and insurance is through a private carrier, with insurance through the individual employer or a group of employers also allowed. (Branch of Workers' Compensation Studies 1991.)

Workers' compensation benefits are not taxable (at least as of 1988), and Income benefits accounted for 61.5% of all workers' compensation benefits in 1981, while permanent partial disability and temporary total disability benefits accounted for 80% of income benefits. Temporary total disability claims are the most numerous. (Ruser 1985; Ehrenberg 1988)

In Kentucky, both temporarily and permanently totally disabled workers receive two-thirds of their wage, with a minimum of $72.41 per week (20% of the State's average weekly wage) and a maximum of $363.03. Permanent partial disability payments for nonscheduled injuries are two-thirds of the worker's wage, with a maximum of $271.51 (75% of the State's average weekly wage), payable for a maximum period of 425 weeks. (Branch of Workers' Compensation Studies 1991)

[44](Ehrenberg 1988.)

The determination of insurance premiums is perhaps the most important component for our purposes, as it plays a large role in determining the incentives firms face in reducing their accident rates.[45] The procedure is fairly similar across most States, as it is set by the National Council on Compensation Insurance or closely follows the Council's procedures.

A firm is first categorized according to one or more of the 600+ industrial-occupational classifications. It is then assigned "manual rates," reflecting the average conditions found in each industrial-occupational classification. The manual premiums are calculated by multiplying manual rates by the payroll for each industrial- occupational class. The smallest firms, which constitute 85% of all employers but only 15% of covered employment, simply pay this unmodified manual premium. These firms therefore face absolutely no incentive through the premium calculation procedure to reduce the incidence of occupational accidents.[46] However, if the manual premium exceeds a certain amount (as is the case with the largest 15% of employers), then the firm is experience-rated, with the manual premium modified to reflect the firm's own past loss experience.

An implication of experience rated workers' compensation programs is that employers will take more fully into account the expected damages of occupational hazards. This internalization will cause a decrease in the injury rate, as the employer's marginal cost of occupational injuries is now higher. The higher the benefit levels, the lower the injury rate. However, the experience rated employer will also have an incentive to dispute workers' compensation claims. This introduces a new and potentially socially wasteful cost into the activities of the firm.

Just as employers have an incentive to dispute claims that are invalid, employees have an incentive to file claims that are not valid. Higher workers' compensation benefit levels may conceivably lead to increases in the reported injury rate which are totally attributable to reporting effects. Injuries occurring outside the workplace may be reported as occupational injuries, the severity of minor ailments may be exaggerated, or previously unreported minor injuries may be officially reported.

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[45] Ruser (1985) provides an excellent summary and synthesis of the literature on insurance premiums calculation.

[46] This may help explain the large differential in fatality rates between small mines and larger ones. Empirical evidence on this hypothesis is scarce. See the efforts by Chelius and Smith (1983) and Ruser (1985).

Summary and Analysis

We identified two mechanisms through which the expected occupational health and safety damages of coal mining may be reflected in the costs of production. The first is through the wages and benefits that employers pay to employees. The theory of compensating wages suggests that in a competitive market the compensation that workers receive reflects the attributes of their occupation, including health and safety risks. We consider four conditions in the coal industry, and in Appalachia in particular, that may undermine the existence of a perfect competition in the labor market.

Three of these conditions suggest that employers may be at an advantage in a bargaining situation with workers. The first is high mobility costs of coal miners relative to blue collar workers in other industries and regions. High mobility costs prevent workers from readily changing locations or occupations. The second is the possibility that workers fail to perceive accurately occupational risks and discount these risks, although it is also possible they exaggerate risks. The third is the narrow base of employment in coal regions that creates a shortage of comparably paid occupations. Employment and productivity trends in the industry are exacerbating this condition.

A fourth condition is the presence of organized labor in many mines, especially large mines, which may serve to offset the advantages attributed to employers. Organized labor is associated with larger mines, which have superior performance records with regard to health and safety issues. The mines that serve as reference environments for this study fall into this category, although we have not assumed that the New Mexico mine will be a union facility. Nonetheless, it is difficult to entirely disassociate the particular reference environments from conditions that may exist throughout the industry, since they are related through competitive pressures in the coal market. In addition there is evidence that when productivity at larger mines begins to fall, production practices change and safety conditions deteriorate.

On balance we surmise that the potential market power of employers, due to the shortage of comparable employment opportunities for workers, and the potential market power of employees due to the muscle of organized labor would be largely offsetting, at least in that segment of the industry that is unionized. Furthermore, the role of information seems to be relatively unimportant in this industry where occupational hazards are a commonly recognized part of the occupational culture.

However, the relatively high mobility costs—with regard to both geography and job change—isolate the labor market and undermine the assumption that wages and benefits reflect working conditions in a manner suggested by the theory of compensating wages. While wages and benefits would still be expected to adjust in response to changes in occupational risks in the industry, they may vary from expected damages by a factor determined by the cost of geographic and occupational mobility for workers.[47]

The second mechanism for the internalization of expected damages into the costs of production is employer compliance with governmental programs. We have examined the Black Lung Benefits Program and State Workers' Compensation Programs. Each of these programs provide compensation to workers who have been harmed due to workplace accidents or exposures through various mechanisms they are largely funded by employers, thereby incorporating much of this cost into the industry's cost of production.

The Federal Black Lung Benefits Program provides a modicum of compensation for workers who have been totally disabled. It is financed through three mechanisms primarily. One is a charge against the responsible mine owner. The second is an excise tax on the entire industry which reimburses the Trust Fund for payments when a responsible mine owner is not identified. The third is tax payer contribution to the Trust Fund. In the future, tax payer contribution is expected to be insignificant.

The distinction between the identification of a responsible mine owner and an excise tax on the industry in general is important from the standpoint of economic efficiency because when the responsible mine owner is not identified, the owner is not held directly accountable for the health and safety record of the mine. Consequently the owner may have insufficient incentive to take care.

However, the problem we are concerned with here effectively takes current industrial practices as a given and focuses instead on the question of whether the expected damage that results is internalized into the cost of production. Therefore the distinction between payment by a responsible mine owner and an excise tax is unimportant. As long as the industry as an entity is footing the bill for expected damages, then those damages are reflected in the cost of the product. Furthermore, from a forward looking viewpoint one can argue that the current levels of payment by the industry could be too high to the extent that the current excise tax is partly dedicated toward retiring obligations created by benefit awards for past exposures.

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[47]In a competitive labor market wages and benefits (W + B) should equal the value of the marginal product of additional workplace risk (VMP_r). A high cost of mobility (M) separates the cost of labor and the value of its marginal product. At the socially optimal level of production: $W + B + M = VMP_r^*$. However, since M is an unrealized cost for the firm it increases risk per unit of product until the value of the marginal product equals its pecuniary expense—just wages plus benefits: $VMP_r = W + B < VMP_r^*$. The firm's production decision (VMP_r) does not adequately represent worker exposure to occupation hazards.

Nonetheless, for many reasons it appears that the current level of benefits provided under the Federal Black Lung Program are insufficient. One reason is the limit on benefit awards. A second is the strict standards for qualification that are currently in place. A third is the discouraging structure of legal representation, and potential financial liability for reversals in qualification rulings. A fourth is the alleged falsification of dust measurements. On balance, current regulatory practice appears to favor industry, leading to an insufficient level of benefits awards for the purpose of internalizing damages.

State Workers' Compensation Programs are another vehicle for internalizing expected health and safety damages. As was mentioned with regard to the Black Lung Benefits Program, the importance that is placed on an experience-rated program for achieving efficient levels of damages is not directly relevant to the question of whether current levels of expected damages are adequately internalized. The reality that benefit levels are fixed under State Workers' Compensation Programs is more relevant because it provides a liability limit for the firm and allows for the possibility that economic damages exceed benefit awards. This is important in determining the degree damages are internalized.

...possibility exists that each of the mechanisms that exist for the internalization of expected mine occupational damages may fall short.

In summary, the possibility exists that each of the mechanisms that exist for the internalization of expected mine occupational damages may fall short. Labor market imperfections seem to favor the idea that wages and benefits do not fully reflect occupational health and safety hazards. Governmental programs are important add-ons which may internalize a portion of what remains. However, governmental programs do not seem entirely adequate, and to some degree they may crowd out some of the wages and benefits that are privately provided, as in the case of the substitution of black lung benefits for pension benefits. We conclude that it is certainly possible that some residual of the expected occupational health and safety damages that are integral to the coal fuel cycle are external costs borne by society and not internalized into producers' costs. Whether this is the case is an empirical question, and unfortunately there is insufficient evidence on which to base a quantitative conclusion.

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PAPER NO. 15*

COAL TRANSPORTATION ROAD DAMAGE

1. INTRODUCTION

Heavy trucks are primarily responsible for pavement damage to the nation's highways.[1] In this paper we evaluate the pavement damage caused by coal trucks. We analyze the chief source of pavement damage (vehicle weight per axle, not total vehicle weight) and the chief cost involved (the periodic overlay that is required when a road's surface becomes worn).

This analysis is presented in two stages. In the first section we present a synopsis of current economic theory including simple versions of the formulas that can be used to calculate costs of pavement wear. In the second section we apply this theory to a specific example proximate to the reference environment for the Fuel Cycle Study in New Mexico in order to provide a numerical measure of the magnitude of the costs.

Theory

Pavement Cost

Road overlays define the endpoints of a pavement's life. The configurations and number of axles on a vehicle matter -- as a rule, the more axles that a vehicle has to distribute its weight the less damage it will cause.[2]

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*Based largely on a working paper by Dallas Burtraw, Ken Harrison, and JoAnne Pawlowski.

[1]We refer specifically to trucks weighing more than 80,000 lbs. *Heavy Vehicle Cost Responsibility Study, Report of the Secretary of Transportation to the United States Congress Pursuant to Section 931 of the Deficit Reduction Act of 1984,* U.S. Department of Transportation, Washington D.C., November 1988, p. iii.

[2]Many State laws, however, penalize trucks with a greater number axles. Fuel taxes punish, because trucks with a greater number of axles require larger engines and get lower fuel economy. Many State turnpikes charge more for a given weight if it is carried on a vehicle with many axles. From: Clifford Winston, "Efficient Transportation Infrastructure Policy", *Journal of Economic Perspectives,* Vol. 5, No. 1, Winter 1991, pg. 116

The life of a road surface (i.e., the interval between road overlays) is affected by the number and type of the axles that pass over it.

The following equation yields the number of axle passages that the road will withstand before requiring an overlay:[3]

$$Nj = \frac{A_0 \, (D + 1)^{A1} \, (L2j)^{A3}}{(L1j + L2j)}$$

Where:

L1j = the weight borne by axle j (in thousands of pounds)

L2j = the type of axle weight. L_2=1 for single axles, L_2=2 for tandem axles (two axles close together),

D = the road's durability. (For rigid pavements, D equals the pavement's thickness in inches. For flexible pavements, D is a linear combination of pavement, base and subbase thicknesses with coefficients 0.44, 0.14 and 0.11. [i.e., D = 0.44(pavement) + 0.14(base) + 0.11(subbase)].

A_j = structural coefficients that describe the durability of rigid and flexible pavements, derived from an empirical study by the American Association of State Highway Officials.[4] For rigid pavements, $A_0 = e^{13.505}$ or 733,073; A_1 = 5.041; A_2 = 3.241; A_3 = 2.270. For flexible pavements, $A_0 = e^{12.062}$ or 173,165; A_1 = 7.761; A_2 = 3.652; A_3 = 3.238.[5]

C = the cost of the overlay per mile

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[3]Kenneth A. Small, Clifford Winston and Carol A. Evans, *Roadwork: A New Highway Pricing and Investment Policy*, The Brookings Institution, Washington D.C., 1989, p.24

[4]The study evaluated 264 rigid and 284 flexible experimental pavement sections, using previously estimated values of N as dependent variables. Cited in *Roadwork*, Small, Winston and Evans, p. 25, from Highway Research Board, *The AASHO Road Test: Report 5, Pavement Research*, Special Report 61E (Washington, D.C.: National Research Council 1962) pp. 36-40.

[5]Small, Winston and Evans, *Roadwork*, p. 27. The authors reanalyzed and revised figures from the AASHO report.

This cost estimate per mile includes all the lanes that must be resurfaced. Typically, for a multiple-lane highway all lanes will be resurfaced simultaneously whenever the lane in greatest disrepair requires resurfacing.[6] The examples we consider in this study involve two-lane highways, so we assume both lanes will be resurfaced when the lane bearing the fully-loaded coal truck requires resurfacing. If this can be avoided, for example, if the road is a divided highway, then separate calculations should be done for each leg of the truck's round-trip journey.

To accurately describe the pavement damage one truck can cause, we need to find values for N for *each* of the truck's axles. These values (Nj) are evaluated against the cost of the overlay per mile. The following equation yields the cost of pavement wear per axle per mile[7]:

$$C/Nj = Cj$$

We sum the values of Cj to obtain R, the cost of the pavement wear per *truck passage* per mile.

To understand what this figure tells us about the social efficiency of electricity produced with coal, we relate the damages caused by coal during transport to the energy coal produces in a power plant. To begin, we divide pavement wear costs by the weight of the load of coal carried by the coal truck (W) to obtain a cost for each round trip mile that a pound of coal travels on the road way:

$$R/W_{coal} = C_{wt.m} \text{ (\$/lb miles)}$$

We divide $C_{wt.m}$ by the heat value of the coal (measured in Btu's) and then multiply that figure ($C_{Btu.m}$) by the heat rate of the electrical generating facility (i.e., the efficiency of the utility plant measured in Btu/kWh):

$$\frac{C_{wt.m}}{H_{val}} = C_{Btu.m} \text{ (\$/Btu miles)}$$

$$C_{Btu.m} \times H_{rate} = C_{kWh.m} \text{ (\$/kilowatt hour miles)}$$

[6]See Small, Winston and Evans (1989), p. 15.

[7]Modified from *Roadwork*, p. 15.

To achieve a cost per kilowatt-hour for a site-specific analysis, we multiply $C_{kWh.m}$ by the number of one-way miles travelled in a single journey by a fully loaded truck:

$$M \times C_{kWh.m} = C_{kWh} \quad (\$/kWh)$$

A Hypothetical Case In New Mexico

The Quemado mine in New Mexico uses trucks to transport coal along a thirty mile stretch of U.S. Route 60. The trucks possess five axles (a single axle plus two tandem axles) and carry 48,000 lbs of coal. They weigh 80,000 lbs when fully loaded.[8] We use this example to suggest the possible method of transporting coal to the reference power plant in New Mexico.

Pavement Costs. The likely route for coal transport, like U.S. Route 60 in the case of the Quemado mine, would involve a two-lane asphalt highway. A recent contract for resurfacing 10.5 miles of U.S. Route 60 cost $402,334 per mile. This sum included all materials, labor, and road preparation necessary for the overlay. A highway engineer for the State of New Mexico suggested that a better figure to use would be $485,000 per mile on average for the State, which reflects inflation and the remote location of our reference facility. We assume the surface includes 4 inches of pavement, 8 inches of base, and 8 inches of subbase.

Table 1. Characteristics of axles on a fully loaded coal truck

Axle	Single Steering	Tractor Tandem	Trailer Tandem
L1: Weight (thousand lbs)	12	17	17
L2: Axle type (1 = single 2 = tandem)	1	2	2
N_j: number of passages	2,686,869	6,340,078	6,340,078

[8] Anecdotal and "off-the-record" information suggests that in many cases, though not necessarily in New Mexico or involving the Quemado mine, coal trucks will be loaded far in excess of their legal limit. Since damage to roadway surface increases exponentially with the weight per axle, excess weight can have a significant impact.

To satisfy our first equation, we need to specify D and $L1_j$ for each axle j. Since the road has an asphalt pavement, its durability, D, is measured by summing the pavement, base and subbase thicknesses with coefficients 0.44, 0.14, and 0.11.

$$D = 0.44\ (4) + 0.14\ (8) + 0.11\ (8) = 3.76$$

Table 1 reports the characteristics of axles on a fully loaded coal truck. L1 takes a value of 12 (thousand pounds) for a single axle and 17 (thousand pounds) for each axle in a tandem. The final row of the table reports the numbers N_j representing the number of passages the roadway surface will withstand for each axle of type j for a fully loaded truck. These calculations are:

$$N_{\text{sngl axle}} = \frac{173{,}165\ (3.76 + 1)^{7.761}(1)^{3.238}}{(12 + 1)^{3.652}}$$

$$= 2{,}686{,}869$$

$$N_{\text{dbl axle}} = \frac{173{,}165\ (3.76 + 1)^{7.761}(2)^{3.238}}{(17 + 2)^{3.652}}$$

$$= 6{,}340{,}077$$

Combining these estimates and accounting for each axle on the truck produces a number NT = 996,922, the number of passages of the fully loaded truck until the road will require resurfacing, absent any other vehicle traffic. The annual needs of the coal power plant will require 79,167 passages of a truck per year. Absent the coal trucks, the roadway will require resurfacing about once every ten years, so current vehicle traffic imposes damage equivalent to about 99,692 coal trucks per year. Including the coal trucks, total vehicle traffic will be equivalent to 178,859 coal trucks per year.

These numbers provide a basis for estimating the present discounted cost of the additional truck traffic per mile of travel for a fully loaded truck, accounting for anticipated resurfacing schedules. Recall that the cost of resurfacing the two-lane highway is estimated to be $480,000 per mile. If we assume the roadway surface is halfway through its useful life at the start of the analysis, then the present discounted cost of an infinitely lived ten year resurfacing schedule (the scenario absent the coal trucks) is $984,261. When the coal truck traffic is included the road must be resurfaced every 5.57 years. Assuming that coal truck traffic continues over the forty-year life of the facility, after which it ceases and total vehicle traffic reverts to the pre-truck scenario, the present discounted cost of the infinite resurfacing schedule is $1,665,372 per mile. The difference is the net present discounted damage estimate per mile attributable to the coal truck traffic, or $681,110.

The estimated distance to be traveled from the mine mouth to the generation facility by the fully loaded coal trucks is 30 miles. The total annualized cost of the damage that results is $1,190,816 per year. Expressed as a levelized cost per kilowatt-hour this estimate of road damage is equal to 0.354 mills/kWh. This is the midpoint estimate of maintenance costs for the roadway that are likely to occur, but it is an under-estimate of total damage due to the absence in this analysis of related effects on congestion, road safety, vehicle maintenance, and other factors that have not been quantified.

There are two uncorrelated sources of uncertainty in this analysis. One is the calculation of Nj, the number of passages the road surface can withstand for each axle type, and the other is the cost of resurfacing. We focus exclusively on the calculation of Nj, which is the greater uncertainty. Making use of standard errors reported in Small, Winston and Evans (1989) we calculated an upper and lower bound for Nj. A 95% confidence interval ranges within factors 0.28 and 3.52 of the point estimates of Nj. Carrying these bounds through the calculations provides lower and upper bounds for a 95% confidence interval for the estimate of damages in this example ranging from 0.101 mills/kWh to 1.241 mills/kWh.

PART V

ISSUES

PAPER 16[1]

ENERGY SECURITY EXTERNALITIES AND FUEL CYCLE COMPARISONS

1. INTRODUCTION

Externalities related to "energy security" may be one way in which the full social costs of energy use diverge from the market prices of energy commodities. Such divergences need to be included in reckoning the full costs of different fuel cycles. In this paper we critically examine potential externalities related to energy security and issues related to the measurement of these externalities, in the context of fuel cycle comparisons.[2]

We begin by attempting to specify what is meant by an energy security externality for the purposes of this study. Energy security can be defined in various ways, but our primary focus is on economic issues related to the behavior of energy markets. We do not address broader military, diplomatic, or foreign policy issues where economics and energy are only a part (sometimes a small part) of the bigger picture.[3] While these non-economic issues may be vital concerns, they also are non-marginal. Changes in individual fuel cycle decisions will have little or no effect in the consequences or resolution of these issues.

We do not address broader military, diplomatic, or foreign policy issues ...

We seek to identify potential energy security costs associated with one or more fuel cycles that are not reflected in current producer and consumer prices and that may be avoidable by a different choice of fuel cycle (the importance and avoidability of the costs are among the major issues to be determined). In keeping with most of the literature on the subject, we consider two broad sets of issues. The first concerns potential effects associated with

[1]Based largely on a working paper by D. Bohi and M. Toman.

[2]A companion paper (Bohi and Toman 1993) considers a wider range of issues at the national level.

[3]Nuclear proliferation and terrorism issues are thus excluded from our discussion.

dependence on foreign energy (mainly petroleum) supplies. These issues involve potential excess payments for imports, and indirect consequences of imports for the economy as a whole. The second set concerns the effects on the economy of fluctuations in energy costs. These fluctuations may derive from disturbances in the world petroleum market or from other shocks of a local or regional nature, such as component failures in the natural gas or electricity delivery systems. In addition, we consider the intersection between energy and market failures in the supply of research and development (R&D).

In this study we are not concerned directly with the effects of public utility regulation, though some indirect effects of utility regulation may be broadly similar to the energy security concerns outlined above. For example, utilities may not provide socially preferred levels of service reliability across customers because pricing regulation does not allow for "unbundled" pricing of different reliability levels (Smeers 1991). The result may be unnecessary disturbances in the flow of electrical energy and fluctuations in its cost. In addition, regulatory imperfections could lead to insufficient long-term capacity construction for gas or electricity delivery and result in unnecessarily scarce supplies. However, while such regulatory shortcomings cause deadweight losses in electricity markets, they are not energy security issues as that term normally is defined.[4] Nevertheless, the consequences of unreliability in energy delivery systems for the whole economy is an energy security issue, similar to the effects of oil price shocks. Similarly, the distortions in the rest of the economy resulting from inflexible electric utility prices can be thought of as an energy security issue. The question remains, however, whether such concerns can be addressed through the choice of fuel cycle. We return to these points subsequently in the paper.

The next section of the paper briefly summarizes the history of instability in the world oil market -- the focus of many energy security concerns -- over the past two decades. The purpose of this discussion is to dispel some common misunderstandings about what has happened that can influence judgments about the existence and magnitude of any market failures. In the succeeding three sections of the paper we turn to a critical review of potential energy security externalities, examining both conceptual arguments and empirical evidence. We consider spillovers related to import dependence, energy price variability, and research on and development of new technology. The sixth and final section briefly summarizes our conclusions.

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[4]Moreover, they may well be endemic problems across fuel cycles that cannot be ameliorated by altering fuel cycle decisions and that require nonmarginal (and difficult) changes in the whole structure of utility regulation to be rectified.

2. PAST EXPERIENCE IN THE WORLD OIL MARKET

The price of crude oil in the United States has been remarkably stable over the past 100 years, as seen in Fig. 1, except for the period between 1973 and 1990. Disturbances in world oil markets during those years are associated with five events: the 1973 "Arab Oil Embargo," the 1979 Iranian Revolution, the start of hostilities between Iran and Iraq in 1980, the precipitous 1986 drop in oil prices, and the 1990 invasion of Kuwait by Iraq. The first two events were marked by rapid increases in the price. The third event was notable for the absence of a price increase, the 1986 experience was a vivid example of a negative price shock, and the 1990 war demonstrated that the oil market does not always panic in a crisis. The discussion below provides a brief summary of these episodes; further details can be found in the references cited.

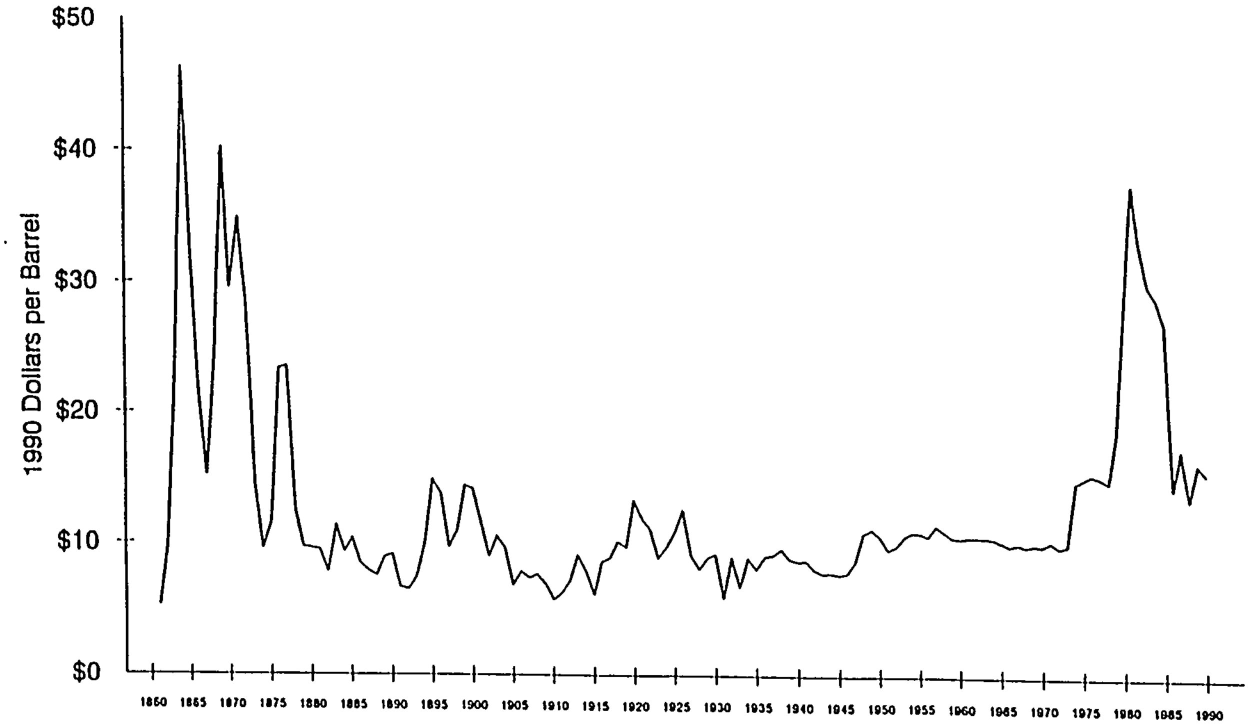
The first oil supply disruption began with the outbreak of the Yom Kippur War in October 1973. Later that month, Saudi Arabia and other Arab states proclaimed an embargo on oil exports to the United States and the Netherlands because of their support for Israel. Actual free world oil production declined from 47.8 million barrels per day (MMbd) in September 1973 to 43.2 MMbd in November, a 9.6 percent decrease. In December the production cuts were partially rescinded, and in mid-March of 1974 the Arab oil ministers agreed to restore production to pre-October levels. As shown in Fig. 2, the spot price of Arab Light oil on the Rotterdam market increased sharply in the first two months of the disruption and then, as the disruption ended, leveled off at over three times the pre-disruption price. What is most perplexing about these price increases is that they occurred without much of a change in world oil supply. As seen in Fig. 3, total OPEC and world crude oil supply increased in 1973, did not decline much in 1974, yet the price rose by over threefold.[5]

Four years of nominal price stability followed the 1973-74 oil embargo, while oil prices decreased significantly in real terms. The calm ended with the strike of Iranian oil workers in October 1978.[6] Iranian production declined to zero by late December as the revolution against the Shah unfolded. In the first months of the revolution the oil price was not greatly affected (see Fig. 2) because other countries (especially Saudi Arabia) increased production to make up for the lost Iranian production. Indeed, total free world oil production was

[5]OPEC: Organization of the Petroleum Exporting Countries.

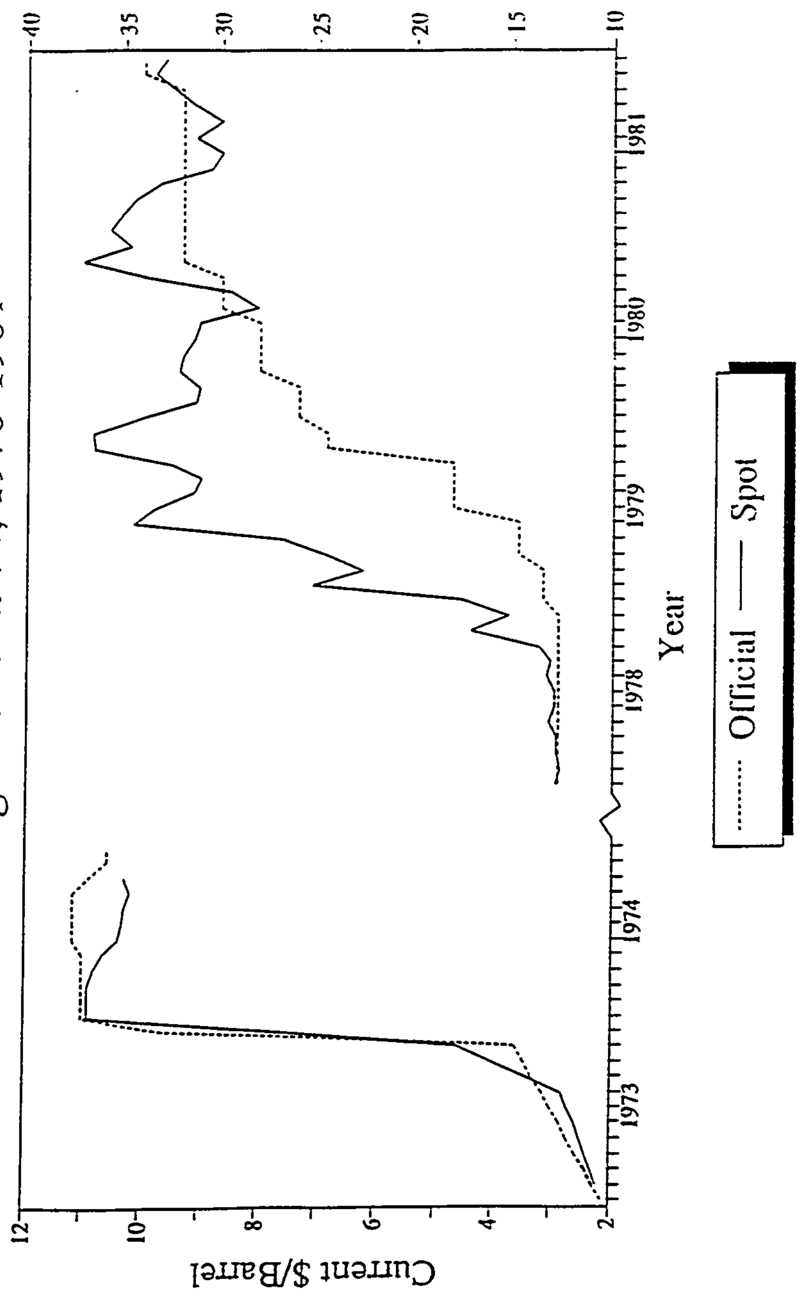
[6]The discussion which follows draws on Badger and Belgrave (1982), Mork (1982), Verleger (1982), Bohi (1983), and Charles River Associates (1986).

Figure 1: U.S. Crude Oil Prices



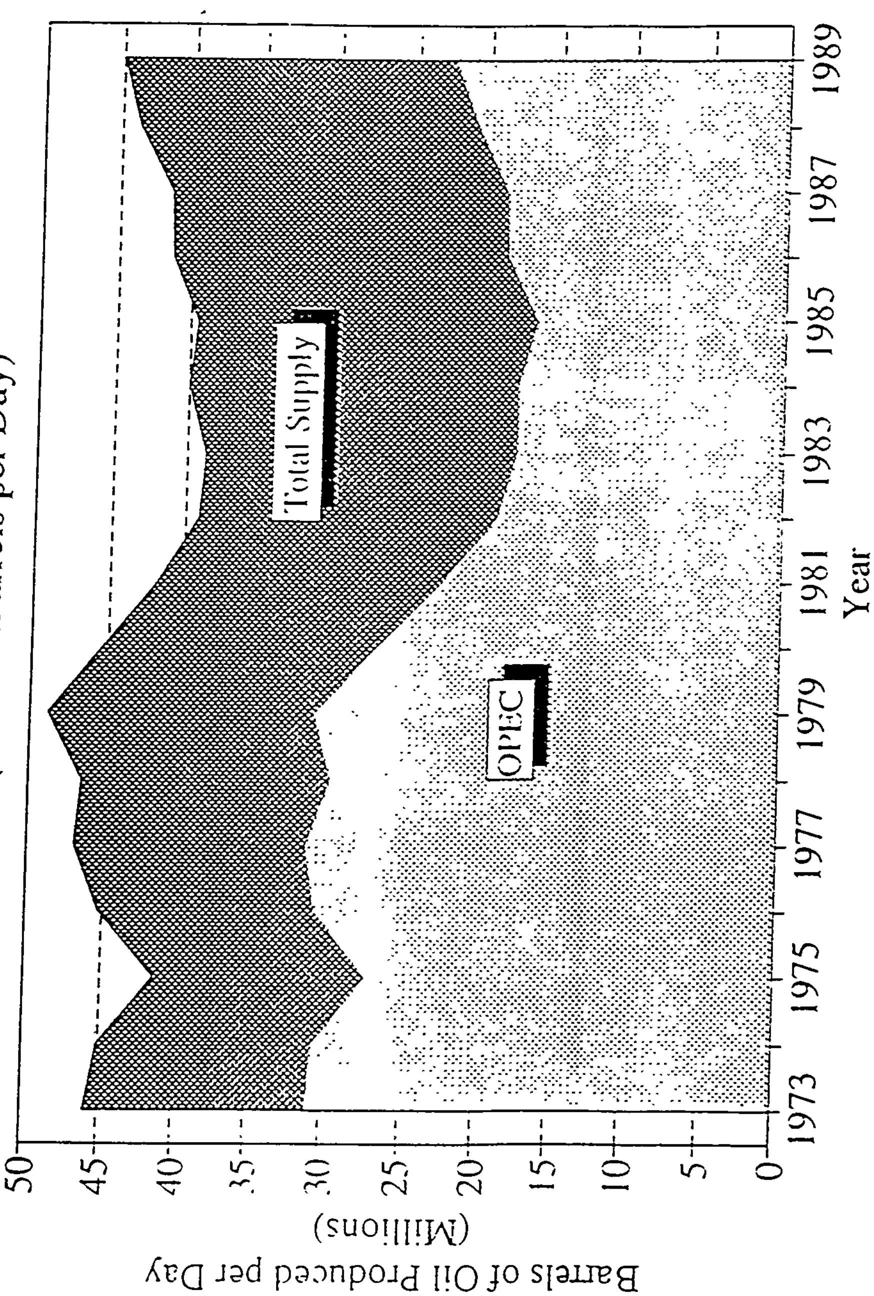
Source: *U.S. Bureau of the Census,* Historical Statistics of the United States, Colonial Times to 1970, Part 2, M138-142, pages 593-594
Energy Information Administration, Petroleum Marketing Monthly, Domestic Crude Oil First Purchase Price, April and July 1990
U.S. Bureau of the Census, Wholesale Price Indices, Producer Price Index, converted to 1990=100

Figure 2: Official and Spot Oil Prices
Arab Light 1973-1974, 1978-1981



Source: Petroleum Intelligence Weekly, various issues.

Figure 3: World Crude Oil Production
(Million Barrels per Day)



Source: Energy Information Administration, Monthly Energy Review

1.4 MMbd higher than in the fourth quarter of 1978 than in the third quarter. However, petroleum product stocks held by refiners within the Organization for Economic Cooperation and Development (OECD) were rapidly drawn down as stocks were transferred downstream to distributors and consumers. In the first quarter of 1979, Saudi Arabia limited its oil production and a significant rebuilding of OECD product stocks began. By March 1979, when Iranian production resumed at a reduced level, the average cost of oil imported into the U.S. was 12 percent above the October 1978 level.[7]

Prices and inventories continued to rise through 1979. By the end of 1979, average import costs had increased 98 percent above the October 1978 level. The price increase occurred despite the fact that actual OPEC (and world) crude oil production increased in 1979 over 1978. The explanation for the price increase is found on the demand side of the market (Bohi 1983). Fearing a repeat of the episode in 1973-74, oil refiners, marketers, and distributors increased their inventories of petroleum products in response to the revolution. The increase in inventories reduced supplies available for consumption, created a shortage, and resulted in a price shock that is usually blamed on OPEC behavior rather than on demand.

Spot prices were especially volatile during 1978-79 as the volume of transactions dried-up in what was a relatively thin market even under normal conditions. In particular, there appeared to be little attempt to arbitrage gaps of up to $20/bbl between spot and official prices. The lack of arbitrage reinforces the belief that sales by OPEC members were made at effective prices greater than official levels, and that average import costs provide a more accurate indicator of true oil prices during this period.

The first nine months of 1980 were relatively calm: OPEC production declined in gradual steps and final demand also fell, though stocks continued to be built up at an unprecedented rate. Spot prices of Arab Light declined, and the gap between official prices and spot prices narrowed (Fig. 2). In September 1980, war broke out between Iran and Iraq. By November, all oil exports from Iraq and a portion of Iranian exports were halted, leading to a total reduction of 3.6 MMbd

... spot price increases were short-lived ...

[7]The reported spot price rose 60 percent between these dates, while official government prices lagged substantially behind spot values; in March 1979 they were only 7 percent above their October 1978 level. During the 1970s reported prices are less accurate than the average cost of imports as a roxy for actual prices.

(from 59.5 to 55.9 MMbd) in free world production. As a result of this cut, spot prices of Arab Light reached $40/bbl in mid-November, while official prices increased only slightly to $32/bbl. The spot price increases were short-lived, however, as they gradually decreased to their 1980 levels by May of 1981.

Although the reduction in OPEC exports was much greater in 1980 following the Iran-Iraq war than in 1978 after the revolution, there was no price shock in 1980. The reason is that market conditions were much different by 1980 so that the war did not produce a panic on the demand side of the market like that after the revolution. By the outset of the war, petroleum stocks were well above normal levels and, because oil consumption was beginning to fall in response to high prices, stockholders were trying to reduce their inventories. However, the decline in stocks did not keep up with the decline in consumption, so that spot prices fell below official prices by 1981. The overall experience with OECD product inventories during 1978-1981 suggests that stockholders seriously miscalculated the crisis, perhaps because of limited experience in responding to oil price volatility. Stocks were accumulated when prices were rising and released when prices were falling, indicating that stockholders were suffering losses. Their behavior also served to destabilize the price, by making the price spike higher than it needed to be.

The price collapse in early 1986 represents yet another curious twist in the evolution of the oil market.[8] Prices were steady or declining during the four years preceding the major drop in price, as non-OPEC supplies increased and purchasers reacted to the previous price hikes by continuing to reduce consumption. In March 1983, the first drop in the (nominal) official OPEC selling price occurred. However, the $5.00 per barrel reduction was not enough, as the members of OPEC would not strictly abide by their production quotas and excess production continued to soften spot prices. As the volume of spot trading grew substantially after 1980, the official price increasingly became just a lagging indicator of spot values. During the four years prior to the price collapse at the beginning of 1986 the real dollar price of oil dropped by roughly 40 percent.[9]

OPEC output began to increase in August 1985 at a rate 25 percent above the level a year earlier, providing the first tangible event that triggered the price collapse in January 1986. Saudi Arabia was responsible for the largest share of the increase, although official production statistics are not considered reliable during this period. The second tangible event preceding the collapse

[8]The paragraphs that follows draw on Gately (1986).

[9]The real cost to countries other than the U.S. dropped during this period less because of depreciation of their currencies relative to the dollar (Huntington 1984; Gately 1986).

was the switch to "netback" pricing by OPEC members in the last quarter of 1985. The official price was now set equal to a composite of spot prices for refined products less the cost of refining and transportation. This move relinquished official control of oil pricing to the market, in recognition of the reality of market pressures and their inevitable effect on export prices. The direction of the effect on crude oil prices at the beginning of 1986 was predictable, even if the exact timing and duration were not. The spot price dropped by roughly 50 percent, from $26/bbl to $13/bbl, in the first two months of 1986, and dipped below $10/bbl by mid-1986 before recovering to around $17/bbl in mid-1987 (see Fig. 4). From 1986 until August 1990, the oil price level fluctuated in the $14-18 range.

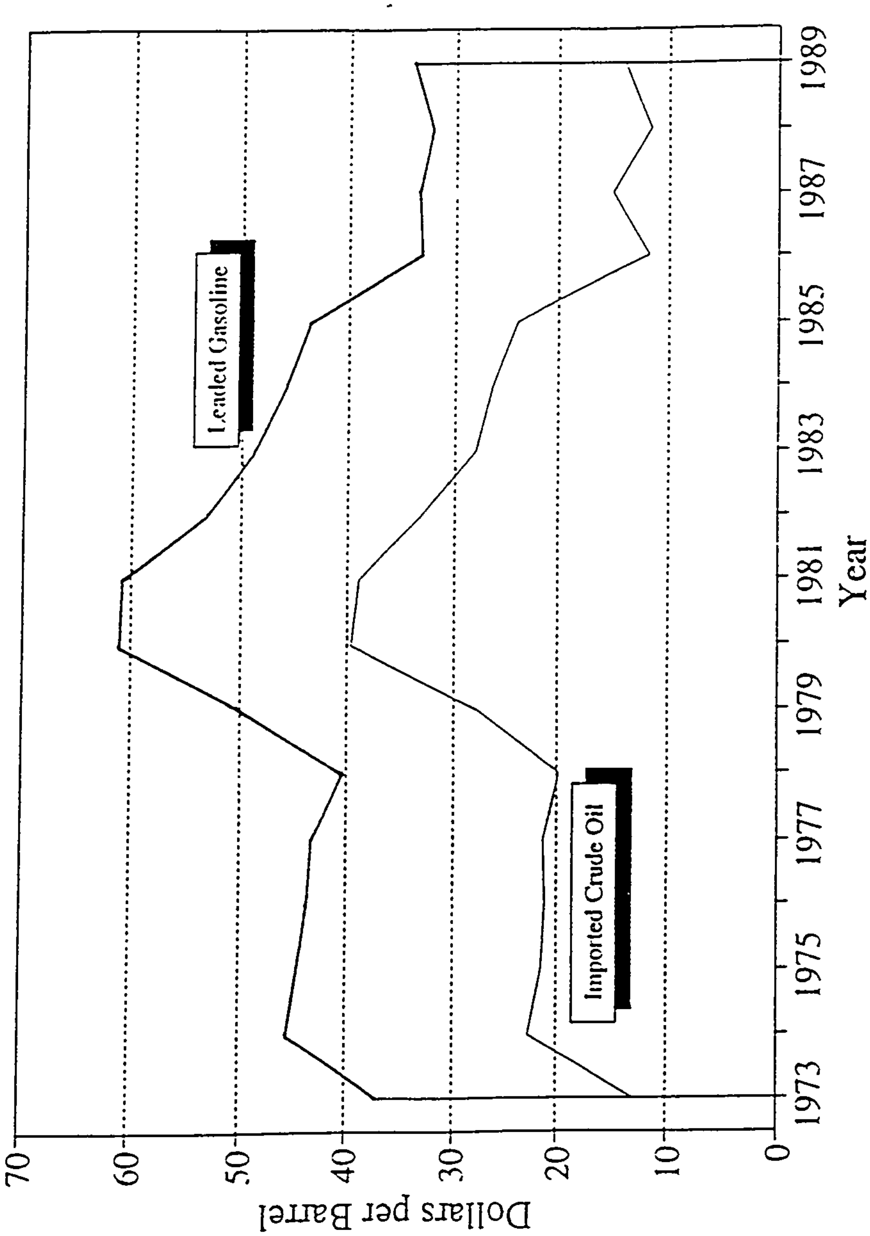
The Iraqi invasion of Kuwait on August 2, 1990 resulted in the immediate reduction of 4.3 million barrels per day (MMbd) of crude oil normally supplied to the world oil market from the two countries, or nearly 18 percent of OPEC exports and nearly 10 percent of total world supply. The spot price of crude oil in U.S. and world markets rose from $21 per barrel the day before the invasion to $28 per barrel within a week afterward (an increase of 33 percent).

In the ensuing weeks, the price of oil fluctuated with changes in expectations about war risks and with increases in production from other sources (see Fig. 5). The crude oil price rose dramatically in the last week of September, for example, in response to Saddam Hussein's threat to destroy Kuwaiti and other Middle Eastern oil fields if war should begin. By the end of November, oil production in Saudi Arabia and other countries had increased by enough to offset the original 4.3 MMbd interruption in supply. Saudi Arabia also announced plans for further production increases, including the development of a newly discovered giant oil field, that were expected to raise Saudi output an additional 1 MMbd by March 1, 1991. The crude oil price remained below $30/bbl during December and January, until just before the start of allied air strikes on January 16, 1991. The next day witnessed a record one-day drop in spot oil prices, and within five days the price of most crude oil was below $20/bbl.

The Iraqi War provided the first opportunity to observe oil futures market behavior during a market crisis.[10] It is of interest to note that throughout the conflict, the futures market consistently discounted crude oil prices for future delivery relative to spot prices. The price for delivery in one month was often $1/bbl below today's price, the 2-month price was another dollar cheaper, the

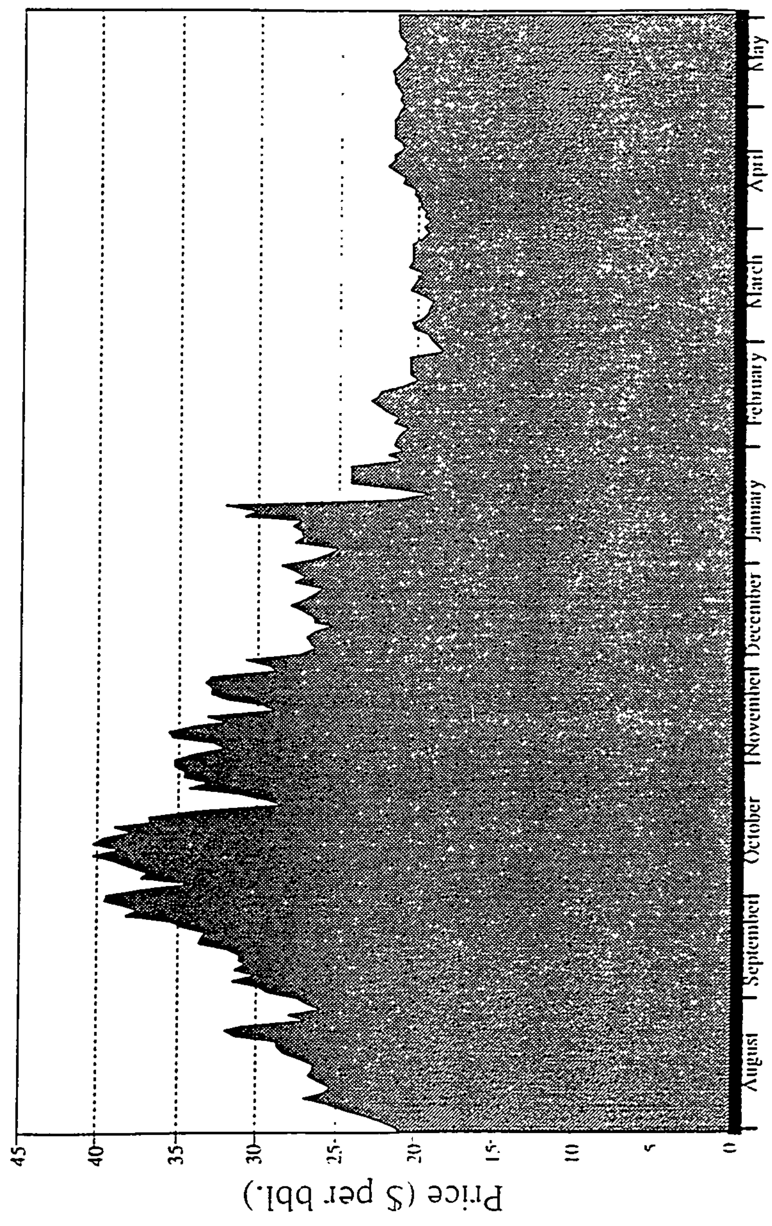
[10]The New York Mercantile Exchange futures market did not begin significant trading in oil futures until 1980.

Figure 4: Gasoline and Imported Crude Oil Prices
Constant (1982-84) Dollars



Source: Energy Information Administration, Monthly Energy Review

Figure 5: Daily Spot Price of Crude Oil
July 30, 1990 - May 24, 1991



Source: New York Mercantile Exchange

3-month price was cheaper still, and so on for 12-18 months into the future. This pattern of price behavior indicates that the market regarded the effect of the war to be temporary with no major impact on the long term level of prices. It also indicated that traders with ongoing oil purchase requirements could significantly reduce their costs relative to spot prices by buying forward. The collapse of the oil price immediately after the start of hostilities is consistent with the view that the crisis would soon end.

The war also gave us our first experience with the use of the Strategic Petroleum Reserve (SPR). On September 6, President Bush responded to growing pressures to sell SPR oil by ordering a one-time test sale of 5 million barrels from the reserve. The announcement not only failed to reduce oil prices, prices actually rose the next day. Soon after the beginning of hostilities on January 16, the U.S. proposed selling over 22 million barrels from the SPR within one month. However, the rapid drop in oil prices after January 16 eliminated any interest in the sale.

3. POTENTIAL EXTERNALITIES RELATED TO ENERGY IMPORTS

As noted in the Introduction, we distinguish between two sources of potential externalities: those related to changes in the quantity of oil imports,[11] and those related to volatility in the price of energy. The first category, discussed in this section, generally involves adjustments that occur over medium to long periods of time, while the second category, discussed in the next section, refers primarily to short-term adjustment problems. In this section we explain why externalities involving oil imports may be present, summarize the arguments concerning whether significant externalities exist, and examine whether they are relevant for fuel cycle decisions.

Direct Cost of Oil Imports

In a perfectly competitive international market the price of a commodity like oil is a complete measure of the worth of a transaction for individual actors. The outflow of $1 of wealth in payments for oil gives rise to an inflow of oil whose value to the buyer (including consumer surplus) is at least $1.

In a market where sellers exercise some market power, the price may lie above the competitive ideal. It is often argued that this is the case in the oil

[11]Under current circumstances this issue primarily concerns foreign sellers of oil, since oil is by far the most widely traded energy type. However, the scope of this issue could expand with an increase in world gas trade through LNG or methanol fuel.

market because of the actions of the OPEC. Nevertheless, the price of oil remains a complete measure of the value for an individual purchaser whose actions do not affect the world price: $1 worth of wealth outflow occurs only if the value of oil purchased is at least $1.

The situation may be different if a buyer's decision may affect the market price. If the market price is positively related to the total volume of purchases, then an increase in purchases raises the cost of all imports.[12] The effect on the cost of inframarginal imports is not reflected in the market price.

The assumption here of a positive relationship between price and total purchases is crucial. This assumption is reasonable if sellers behave competitively, so that there is a well-defined aggregate supply schedule. However, it may not be appropriate when sellers possess potential market power. The United States as a whole may possess this kind of "monopsony" power in the world oil market, even though individual buyers do not. If oil import demand could be coordinated it might be possible to drive down the world price of oil, benefitting all oil users in the U.S. (and elsewhere). In the absence of such coordination the effects of individual oil users in driving up the prices paid by all might be viewed as an externality.

Usually monopsony effects are thought to be only "pecuniary" externalities, effects that redistribute rents but do not bear on market efficiency. Indeed, such effects are ubiquitous in efficient markets where short-term or long-term scarcity (increasing supply cost) causes a bidding up of prices with increasing demand. When the rent redistribution involves rent transfers out of the purchasing country, the size of these wealth transfers may be a concern for policymakers even if the market is efficient from a global perspective. It does not necessarily follow, however, that a potential monopsony position should be exploited. The U.S. eschews the exercise of monopsony power in a number of international markets out of a belief that it is in its best interest to do so. To argue for the exploitation of monopsony in the world oil market, it is necessary to conclude that the policy decision can affect world prices and that it will not provoke a retaliation by exporters which would leave the U.S. worse off.

... monopsony effects are thought to be only "pecuniary" externalities, effects that redistribute rents ...

[12] See Bohi and Montgomery (1982) for further exposition. The phenomenon can be simply illustrated in mathematical terms. Let Z denote the volume of imports, and let $P(Z)$ denote the price. The effect of an increase in imports on the total cost is given by $d(PZ)/dZ = P(Z) + Z\,P'(Z)$. The second term measures the effect of the price increase on inframarginal imports.

Should monopsony effects be included in fuel cycle evaluations?[13] They are not relevant to individual local fuel cycle decisions because monopsony effects operate only at a national scale. The petroleum purchases of an individual electric utility are too small relative to the scale of world oil transactions to have any perceptible effect on world oil prices; for all intents and purposes, the individual utility faces a perfectly elastic supply schedule for oil imports.[14]

The question of whether monopsony effects should influence fuel cycle evaluations at the national level is more complex. They cannot be addressed directly in the absence of some means for coordinating oil demands at a national level, since this is the only level at which monopsony effects are meaningful. However, the national government could take concerns over oil import costs into account in the design of R&D policies, by favoring research into fuel cycles (and other energy technologies) that use energy sources other than oil or that promote domestic oil supply. Presumably this approach would be sufficiently indirect that the risk of retaliation by oil exporters would be small.[15]

The petroleum purchases of an individual electric utility are too small relative to the scale of world oil transactions to have any perceptible effect on world oil prices; ...

As we discuss further in a subsequent part of the paper, there is an argument for government involvement in R&D generally on a variety of grounds. In particular, R&D that curbs oil consumption may be one way to reduce the cost imposed by any exercise of market power by petroleum

[13]It is important to recognize the narrowness of this question. Almost all debate over oil import policy has been concerned with direct efforts to restrict imports (tariffs, quotas) or enhance domestic oil supplies (tax inducements) at a national level (see Toman 1991, Bohi and Toman 1993, and references therein). At this level of consideration, the capacity to influence world oil prices is at best one necessary condition for limiting oil imports as we have noted above. Other factors that must be considered include: (i) do sellers exercise market power, providing a justification for a strategic trade response (versus just acting to "beggar one's neighbor); and (ii) could sellers credibly retaliate to a trade limit (as opposed to causing themselves more harm than good)? In Bohi and Toman (1993) we argue that the degree of market power actually exercised by OPEC has been uncertain at best; and to the extent that OPEC possesses significant latent market power, import limits risk galvanizing a retaliatory response.

[14]Even if an individual fuel cycle decision did have an effect on oil prices, so little of the benefit would accrue to the individual decisionmaker that it would not be worthwhile for the individual decisionmaker to incur the cost of curbing oil use.

[15]Nevertheless, it is at least conceivable that some strategic R&D could be subject to scrutiny as a non-tariff barrier under the General Agreements on Tariffs and Trade (GATT).

exporters. Reduced consumption diminishes the loss imposed by oil prices above competitive levels, even if monopsony effects on the price of oil are not that substantial.[16] In addition, R&D directed at all energy demands, not just oil, may have salutary effects in limiting energy security costs associated with energy price volatility (discussed below).

In fact, however, even at a national level the capacity of the U.S. to influence world oil prices by curbing demand on imports is likely to be limited. U.S. demand is only a fraction of world oil use (roughly 30 percent), and imports are only a fraction of total U.S. oil demand (between 40 and 50 percent). Any drop in oil prices from a decline in U.S. demand will be partially offset by increases in other countries' demands. Walls (1990) calculates that any monopsony benefits from an incremental curbing of U.S. oil imports likely are well under $1/bbl of imports. Even proponents of an active oil import policy, like Broadman and Hogan (1986, 1988), attribute only about $2/bbl to the "monopsony wedge."

Indirect Costs of Oil Imports

Even without the presence of monopsony power or the exercise of market power by oil exporters, transfers of wealth for oil imports could have secondary effects on the economy that are not reflected in the price of oil and constitute a potential externality in the oil fuel cycle. The payments for oil imports have an unfavorable effect on the U.S. merchandise trade balance, which could in turn have a negative effect on the international exchange value of the dollar and on the cost of all imported goods. It also has been argued that higher oil prices could aggravate "structural" inflation that leads to adverse macroeconomic consequences. Broadman and Hogan (1986, 1988) attribute a substantial proportion of the total "import premium" they calculate to these effects.[17]

We note from the beginning that even if these effects constitute real externalities and are significant in magnitude, they are not amenable to policy responses at the level of individual fuel cycle decisions. Like the monopsony issue, responses to exchange rate and inflation issues are meaningful only at the national level because the impact of isolated decisions is negligible. Thus, the discussion that follows is relevant only to fuel cycle evaluations at the national level, such as for guiding R&D.

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[16]As noted above, the degree of market power exercised by OPEC is uncertain.

[17]Broadman and Hogan's figures derive in turn from earlier estimates and judgments in Nordhaus (1980) and Hogan (1981).

The argument that higher oil prices translate into depreciation of the dollar seems intuitively appealing. An increase in the price of oil means (assuming oil demand is price-inelastic) that total payments for oil rise and (assuming all other trade is fixed) the current account will move toward deficit. A current account deficit leads to an overall balance of payments deficit (assuming no change in capital flows) which in turn implies an excess supply of dollars in foreign exchange markets. Consequently, the international value of the dollar will fall and all U.S. imports will be more costly -- the U.S. must export more goods to buy the same amount of imports. While this is a pecuniary effect, it could be viewed as relevant to U.S. national welfare in the same way that U.S. interests are related to monopsony power -- limits on U.S. oil imports could curb the cost. However, while the argument may have appeal, the necessary sequence of assumptions is not likely to hold.

The conclusion of two complementary analyses of the balance of payments effects on prices, and the behavior of exchange rates after each oil price shock, is that it is inappropriate to attribute an exchange rate externality to oil imports. One analytical approach is concerned with real terms-of-trade effects of higher oil prices, as in Marion and Svensson (1986). The terms of trade refers to the amount of imports a given unit of exports will command in the international market; thus, a rise in the price of oil means that the U.S. must export more goods to buy the same amount of oil. Marion and Svensson demonstrate that the terms-of-trade effect of higher oil prices can be positive or negative for any individual oil importing country, depending on special circumstances for each country.[18]

The second analytical approach looks at the effect of oil prices on the monetary exchange rate, as in the work of Krugman (1983). Like Marion and Svensson, Krugman shows that the relationship between a country's exchange rate (or, for that matter, its current account position) and the price of oil is ambiguous in general. All oil importing countries will experience an initial current account deficit when the price of oil rises, but the effect on exchange rates among oil importing countries will depend, initially, on the willingness of the oil exporting countries to hold different foreign currencies (that is, on relative capital flows). If oil exporters prefer to hold more dollars than other currencies, for example, the dollar exchange rate will rise. Over time, the

[18]Their analysis includes the counterintuitive result that those countries least able to adjust to higher oil prices (because of rigidities in the way oil is used in their economy) will tend to experience the greatest reductions in domestic output and, assuming other factors constant, will experience an <u>improvement</u> in their terms of trade. This outcome is reached because a decline in home output relative to that of other countries means that there is now a relative shortage of home country goods in world markets. The relative shortage will cause an improvement in the home country's terms of trade. This is one way in which the terms of trade may be inversely related to economic performance (the improvement in the terms of trade presumably is of cold comfort to the country experiencing the deeper recession).

exporting countries will spend their foreign currencies on goods or assets, and the countries of preference for these expenditures will experience currency appreciation.

A study of exchange rate behavior by Trehan (1986) finds only weak empirical support for the view that higher oil prices lead to an appreciation of the dollar. A more defensible conclusion, in view of the weak statistical results, is that the price of oil is a poor predictor of the dollar exchange rate, either positively or negatively. A look at the history of the dollar/SDR exchange rate in Fig. 6 corroborates these findings. The SDR represents a composite of other currencies. The SDR exchange rate with the dollar shows that the value of the dollar is not harmed by oil price increases nor helped by oil price reductions.

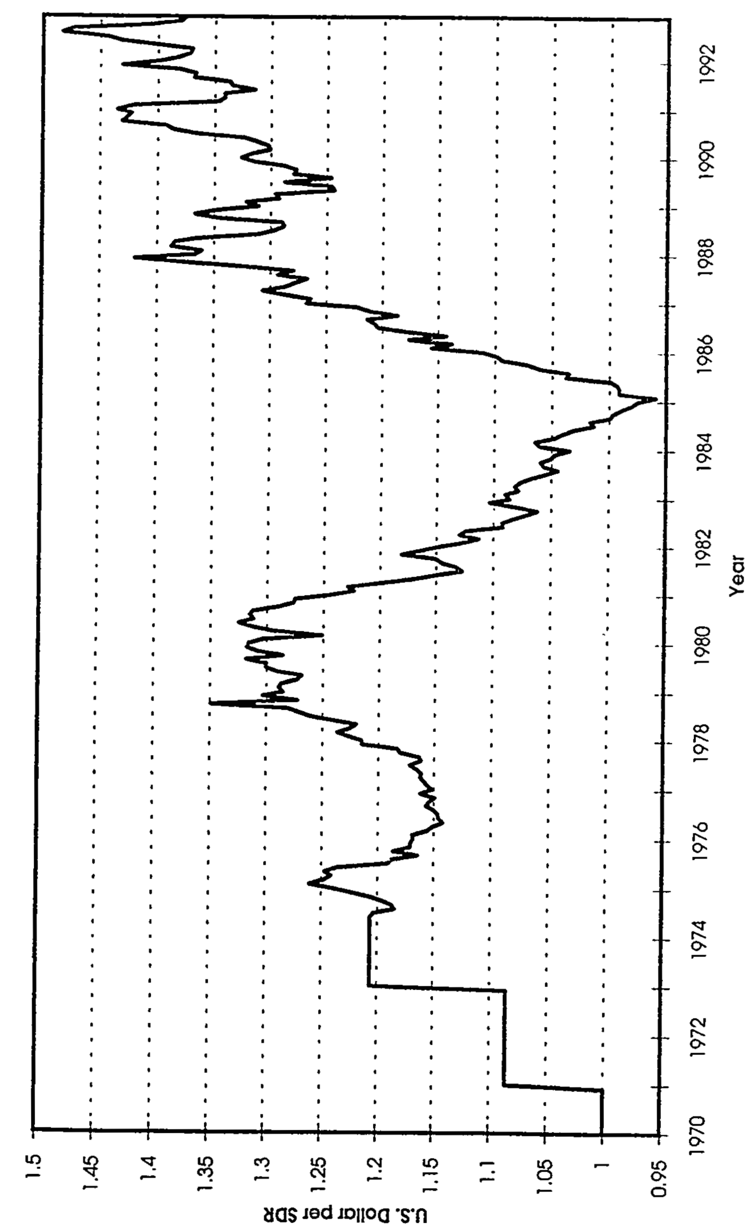
We turn next to the question of the connection between oil prices and inflation. Higher oil prices will no doubt raise all prices somewhat, but unless oil prices continue to rise there is no ongoing inflationary process in the long term, only an increase in the price level (though that increase may be spread over time). Similarly, a rise in oil prices will aggravate an inflationary process that is already in motion, but higher oil prices are not the cause of that inflationary process. The distinction between inflation and a rise in price, and between cause and effect, is important in establishing whether higher oil prices cause a need for a deflationary policy response.

To attribute an inflationary side effect to increases in oil prices, the increase must be in the rate of growth of oil prices, not merely in the level of oil prices. Our understanding of world oil markets is not complete enough to rule out the possibility of a boost in oil price growth rates. However, a rise in growth rates is inconsistent with the predictions of resource supply theory and with the record of actual oil prices. Resource supply theory predicts that increased producer market power will likely result in a rise in the initial price level, and that the price thereafter will likely grow more slowly over time compared to the price in a competitive market.[19] Actual oil prices exhibit surprising little trend of any kind, as indicated by the history of crude oil prices in the U.S. since 1889 shown in Fig. 1. The overall trend is flat over the past 100 years and, apart from temporary crises, the price has fluctuated on a fairly narrow band of $10-$15 per barrel.

... any connection between oil prices and inflation seems dubious ...

[19]Bohi and Toman (1984), Chapter 5.

Figure 6
U.S. DOLLAR/SDR EXCHANGE RATE



Note: Prior to July 1074, the SDR (Special Drawing Rights) were valued relative to the U.S. dollar. Since July 1974, their value is based on a weighted basket of currencies.
Source: International Financial Statistics, International Monetary Fund.

Another possibility is that an ongoing bout of inflation can result from an overzealous monetary authority who is seeking to accommodate a rise in oil prices. The monetary authority could err in estimating what is required to adjust to a higher level of oil prices, thereby setting in motion an inflationary spiral that requires a deflationary jolt to the economy to stop. However, inflation scenarios that rely on planning errors by the monetary authorities can be triggered by any number of events, and to focus policy on the triggering event rather than the cause of the problem seems misguided at best. Nor is there a reasonable second-best argument for attaching an inflationary spillover cost to oil prices, since the first-best alternative of educating the monetary authority is feasible and more likely to avoid further costs than a policy aimed at oil prices or imports. In short, any connection between oil prices and inflation seems dubious and would reflect at most a policy failure, not a market failure.

4. POTENTIAL EXTERNALITIES RELATED TO ENERGY PRICE VARIABILITY

Externalities may arise from difficulties experienced by the domestic economy in adjusting to rapid fluctuations in energy prices.[20] Sudden changes in energy prices could result from a variety of causes, including changes in world petroleum markets and fluctuations in local energy delivery capabilities (e.g., gas well freeze-ups, power station breakdowns, transmission system failures). The effects could be local, regional, or national.

Among the possible causes of sudden increases in energy prices, the most serious from a national perspective inevitably will involve a shift in world oil supply or demand. Disruptions in oil markets represent the only event capable of seriously taxing demand-supply balances in national and world energy markets. Moreover, given substitution possibilities between petroleum and other energy sources, oil price shocks are capable of causing a simultaneous disruption in all energy markets.[21]

Energy price shocks can have both long-term and short-term effects on the economy, though only the short-term effects will likely involve externalities. Long-term adjustments will take place in the amount of energy-related

[20]As discussed below, gradual changes in energy prices are unlikely to generate externalities because private markets have the capability to adjust over time without serious efficiency problems.

[21]See Mork (1985) for discussion of how shocks in one energy market propagate on other energy markets.

investment and in the rate of innovation. These adjustments could have significant effects on the level of productivity and the economy over time.[22] These adjustment costs will be fully internalized in private decisions. Only in limited circumstances is there a priori reason to believe the costs could be avoided by correcting a market failure (such as that related to R&D, as discussed below). Thus the following discussion of price-related externalities deals exclusively with the problems of adjusting to a sudden change in energy prices.[23]

The most important short-term adjustment problem concerns the possibility that real wages will not adjust to maintain employment when energy prices suddenly rise. A rise in energy prices will reduce the use of energy and (when energy and labor services are complementary inputs in production) will lower the marginal productivity of labor. Lower productivity implies an increase in the cost of labor, which employers will seek to reduce. If wages cannot be reduced (for institutional reasons such as periodic labor contracts), employers have no choice but to reduce the amount of employment. The decline in employment is an indirect effect of higher energy prices that lowers total output of the economy.

A second possible indirect effect of an energy price shock is a reduction in capital services because of premature obsolescence of energy- using capital stocks. A sudden increase in energy prices will make some energy-inefficient capital goods superfluous, either because of competition with more efficient capital goods or because the demand for more expensive energy-intensive end products declines. A decline in capital services implies a reduction in productive capability throughout the economy, and thus a reduction in potential output.

Employment and output losses may be further aggravated by difficulties in reallocating factors in response to changes in the mix of final demand brought about by changes in product prices. For example, commodity price rigidities could cause inefficient inventory accumulations in some sectors and unwanted decumulations in others. Another source of commodity price rigidity is public utility regulation in the natural gas and electricity industries. Since prices are set administratively in these industries and will not easily adjust to changes in market conditions, energy supplies will not necessarily flow to their highest valued uses, and productivity can be adversely affected.

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[22]See Jorgenson, Gollop, and Fraumeni (1987); Jorgenson (1988); and Hogan and Jorgenson (1991).

[23]See Bohi (1989) and references therein for further discussion of these issues.

These macroeconomic adjustment problems are fundamentally different in character from the problems associated with oil imports discussed above. The problem of excess wealth transfer for imports depends on the level of energy prices and the volume of energy imports, whereas the macroeconomic adjustment problems depend on the size of the change in energy prices and the volume of energy consumption. In addition, the wealth transfer problem operates at a fundamentally national scale, whereas the oil price adjustment problems could vary significantly from region to region depending on the causes of the disturbance, the flexibility of local energy production and consumption, and the effects of energy prices on local labor markets and capital utilization.

We first examine the extent to which these macroeconomic effects may be thought of as externalities. The conclusion is that some market failures may be present, but it is difficult to distinguish between externalities and private frictional adjustment problems that are accounted for in private market decisionmaking. The empirical section that comes next looks at gross macroeconomic effects of energy price shocks, since costs attributable to externalities are impossible to distinguish from those that are internalized, and concludes that even at a national level there is also considerable ambiguity in ascertaining the gross effect of energy price shocks on the economy. It follows that local effects are even less clear. Consequently, little can be said in practice about the subset of gross macroeconomic costs that are attributable to market failures and should be included in comparing the social costs of different fuel cycles.

Distinguishing Externalities

To what extent can macroeconomic effects of energy price shocks as described above be thought of as externalities relevant to fuel cycle comparisons? To address this question we must distinguish two types of reductions in economic activity that might result from an energy price increase. The first type of reduction, which is not an externality, occurs because more resources are required to pay for energy. Part of this direct resource cost is the income transfer abroad to pay for energy imports and part is the diversion of resources away from other economic activities and into domestic production of energy. These effects occur because of private sector responses to changes in energy prices, so they represent costs that are internalized in private decisions.[24]

[24]The income transfer abroad could include monopoly profits if OPEC exercises its market power, in which case these excessive profits constitute an external cost. However, these external costs, to the extent they occur and can be identified, are already included above in connection with the quantity of imports and should not be counted again here.

As noted previously, national income may drop below the level that would obtain in a perfectly frictionless economy if the energy price shock results in unemployment of labor and capital.[25] Some of these effects represent relatively clear cases of imperfection in factor market adjustment that would be lessened with less exposure to energy price volatility.[26] In other cases factor markets may be operating as well as possible given real-world institutional constraints, but adjustment problems still can be ameliorated by some outside policy action. In either case we can say that an externality exists to the extent that the parties to the labor and capital transactions themselves cannot fully avail themselves of means to anticipate and respond to the energy price shocks. Conversely, to the extent that the costs are anticipated and coped with, the effects of energy price instability are internalized.

To illustrate the various distinctions among effects of energy price volatility, an increase in labor unemployment after an energy price shock may at least partly reflect the existence of institutions like union contracts that encompass a degree of wage rigidity not suited to an environment of energy price variability. Individual workers as well as employers might prefer more flexible arrangements, but, in practice are unable to achieve them. In this case, an energy price shock triggers a failure in the market for labor services and induces unemployment. The resulting increase in unemployment after an energy price shock reflects a market failure.[27]

Some rigidities may reflect labor contracting practices that are efficient ex ante, given real-world informational and institutional constraints. In particular, downward real wage adjustments after an energy price shock may reduce labor productivity, because lower wages cause the average quality of the workforce to decline. This can happen because the best workers quit and the cost of being caught shirking declines.[28] In this case, wage rigidity reflects institutional arrangements in the labor market that represent the best choice among currently available methods for structuring labor services transactions.

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[25]In addition to effects reflecting relative factor prices, some authors would include short-term Keynesian contractionary effects resulting from the failure of energy exporters to respend their proceeds (see e.g., Pindyck 1980, Solow 1980). However, we view these effects as far more debatable and do not include them here.

[26]We emphasize lessened exposure to energy price volatility because that is the main effect of fuel cycle decisions. Other policies at the national scale, like the Strategic Petroleum Reserve, can alter the volatility of prices that is experienced. See Toman (1991) and Bohi and Toman (1993) for discussions of these measures.

[27]Unemployment shock also can result from sticky commodity prices in monopolistically competitive product markets. Sticky commodity prices magnify the burden of adjustment on output quantities and thus on employment when energy price shocks change relative costs of production.

[28]For an excellent survey of alternative explanations of wage and price rigidity and unemployment, along with copious references, see Mankiw (1990); see also Blinder (1991) on price rigidity.

Nevertheless, a choice of fuel cycle with less volatile costs could imply less of an adjustment burden due to wage rigidity, so the effects of the rigidity would remain relevant for fuel cycle comparisons.

Problems of identifying externalities also arise in analyzing a decrease in capital services from accelerated obsolescence of energy-intensive capital after an energy price jump. Private agents who anticipate the risk of future energy price shocks will have an economic incentive to hedge by adopting less energy-intensive technologies, or technologies which allow greater short-term flexibility in substituting away from energy inputs (see, e.g., Bohi and Montgomery 1982, Chapter 4 and references therein). Hedging also can be accomplished by expanding energy storage capability so that energy can be purchased at lower cost for use during a disturbance.

Even with these private hedging activities, society may experience adjustment costs from decreased capital utilization after an energy price shock. Hedging opportunities will not work perfectly, even under the best of circumstances. Private agents will not have perfect foresight of energy changes. Nor will capital be perfectly malleable across sectors and uses, even with investments in flexibility. Again, fuel cycles with less volatile costs might lessen the difficulties. At issue here is the extent of the market failures which impede the hedging process, and their relevance to fuel cycle comparisons.

One possibility is the existence of substantial, systematic price forecasting errors that bias private actors away from efficient investments. No doubt there have been episodes of significant forecasting error in the past, notably in the response of U.S. refiners to price signals in late 1978 and early 1979 during the Iranian Revolution (Bohi 1983). However, pricing efficiency in petroleum markets -- the degree to which prices embody relevant information about market conditions -- seems to have increased (Green and Mork 1991). The literature does not provide strong support for the existence of systematic problems in forecasting energy prices. Even if such errors could be identified, it is problematic that government intervention (other than its current activities in publishing energy statistics) could improve upon the situation.[29] Moreover, forecasting error will plague the evaluation of all fuel cycles at least somewhat.

Another, possibility is the existence of scale economies in energy storage which result in less than the socially efficient amount of storage being undertaken. This is one of the justifications for government investment in the Strategic Petroleum Reserve. An additional justification stems from the notion

[29]Since gas and electricity prices are subject to federal and state regulation, one source of price unpredictability is changes in regulation. While stability of the regulatory environment is useful for holding down adjustment costs, there often are compelling arguments for regulatory change to mitigate current inefficiencies.

that the private rate of discount exceeds the social rate because of tax distortions, barriers to risk-pooling, and other factors (Lind 1982). A high private discount rate will lead to less oil storage and will discriminate against more capital-intensive, less energy-intensive production technologies if capital and energy are substitutes, or against capital expenditures that promote energy flexibility. Unfortunately, however, the degree of capital-energy substitutability remains a matter of dispute.

Our definition of externalities includes adjustment costs in the rest of the economy resulting from the pricing of services provided by public utilities, where regulation generally impedes adjustment of commodity prices to changing market conditions. Because of regulation, an increase in oil prices will not lead to efficient adjustments in natural gas and electricity prices. These price rigidities in turn may cause adjustment problems throughout the economy.[30]

While distortions in gas and electricity markets from economic regulation may be endemic, society has not yet developed practical alternatives to current forms of regulation that will simultaneously constrain market power and allow regulated prices to adjust the changes in market conditions. Nevertheless, energy policies <u>may</u> be able to ameliorate the <u>consequences</u> of output price rigidities in utility markets, and the spillover effects of these rigidities to labor and capital employment, by helping to reduce input price variability or enhance adaptability. Thus, from the standpoint of energy security we would view spillover effects of commodity price rigidity in utility markets no differently than other commodity price rigidities.

To summarize, there are plausible theoretical arguments for the existence of externalities from energy price volatility that may be relevant to fuel cycle decisions. However, theory also reveals ambiguities in calculating the costs of volatility based on the causes of rigid adjustment and the degree to which volatility of energy prices is accommodated ex ante. These ambiguities are compounded in empirical analyses of volatility costs, to which we turn next.

... there are plausible theoretical arguments for the existence of externalities from energy price volatility that may be relevant to fuel cycle decisions. However, theory also reveals ambiguities in calculating the costs ...

[30]Note that automatic fuel adjustment clauses avoid this problem, but they also give rise to moral hazard problems in cost-minimization by utilities Baron and DeBondt (1979), Isaac (1982).

Empirical Findings

Empirical studies of the macroeconomic effects of energy price shocks do not try to distinguish between internalized and externalized costs. The best that we can do, therefore, is to try to assess the importance of the gross macroeconomic costs of energy price shocks and draw inferences about the empirical significance of the externality component.

Perhaps surprisingly, the evidence about the gross costs at the national level is mixed. The coincidence of timing of the two oil price increases and two recessions during the 1970's leads many observers to believe that the effects of energy price shocks on the economy are large.[31] This view is best represented by an extensive simulation analysis conducted by the Energy Modeling Forum and published in Hickman, Huntington, and Sweeney (1987).[32] The study compares estimates of the effects of various oil shock scenarios on U.S. GNP calculated by a group of 14 macroeconometric models, including standard large-scale models used by forecasting and consulting companies. While there is considerable variation among the findings among individual models, there is a consensus that the calculated GNP losses are substantial.[33]

The authors of the EMF study take pains to enumerate caveats which must be considered when interpreting their results. They indicate that the individual model results vary significantly because of differences among the models with respect to the relationship between GNP and the overall price level, and with respect to the link between oil prices and the general price level. Nevertheless, most of the models have the same basic mechanism for the transmission of energy shocks. Increased energy costs cause firms to increase their price markups, and higher prices depress aggregate spending. The reduction in aggregate spending reduces the demand for labor, but wages cannot fall fast enough to trim labor costs in line with reduced demand. Consequently, employment declines.

The main source of skepticism about the results of these models is that the equations of the models employ parameters estimated from limited experience with price shocks over the 1950 to 1980 period. During this period real oil prices were stable or falling except for the two brief explosions during

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[31]Reports by DOE (1987, 1988) reinforce this view.

[32]U.S. Department of Energy (1988) also estimates large macroeconomic costs due to energy price shocks.

[33]Broadman and Hogan (1986, 1988) also calculate a macroeconomic disturbance premium equal to 10-12 percent of the normal-market oil price, though the basis for their calculation is different and they mistakenly attribute the premium to oil imports rather than to total oil consumption.

the 1970's. Thus, the conclusions of the models regarding the relationship between oil price increases and GNP will be determined by the experience with the two recessions that followed the 1970's price shocks, although this experience may not be representative of the true energy-economy relationship. As noted below, the recessions experienced in some countries could be explained by factors other than energy prices, such as differences in macroeconomic stabilization policies. It is possible, in other words, that the econometric models are confusing the effects of deflationary macroeconomic policies with those of changes in oil prices.

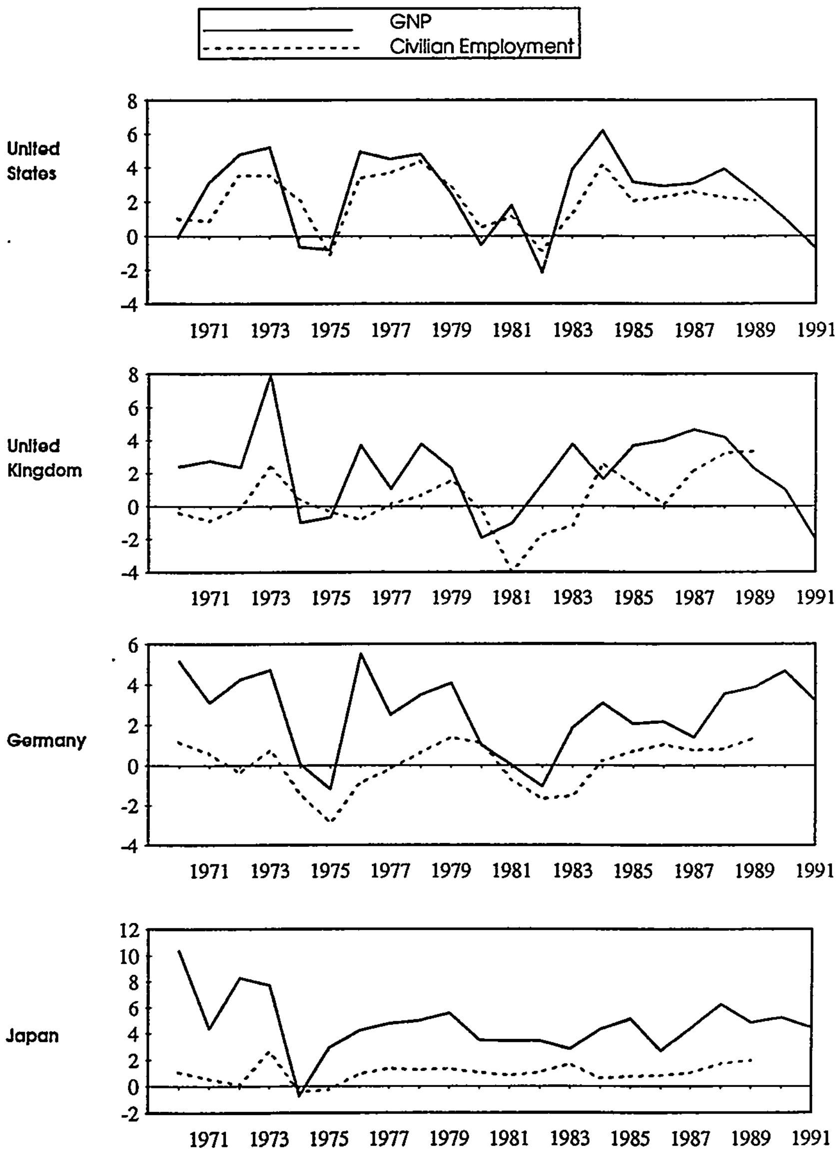
Another reason for skepticism in blaming energy prices for the recessions of the 1970's is that the price collapse of 1986 did not cause an economic boom in the U.S. and other industry countries (see Fig. 7). The absence of a positive boost to GNP in the major industrial countries suggests again that the experience of the 1970's gives a misleading impression of the energy-economy relationship. Additional corroboration is provided by comparing the conclusions of Hamilton (1983) with Mork (1989). Hamilton looks at correlations between energy prices and GNP before 1981 and concludes that higher energy prices cause recessions, while Mork finds that the same statistical methods applied to data extended through 1986 give a different impression.

... skepticism in blaming energy prices for the recessions ...

Doubts about the meaningfulness of statistical results based on aggregate economic variables led Bohi (1989, 1991) to examine disaggregated industry data for the U.S., Germany, Japan, and the UK for explanations of the experiences of these countries during the 1973-1974 and 1979-1980 shocks. The results of this analysis suggest that energy prices may have had little to do with the macroeconomic problems of the 1970's. To begin with, one might expect more similarity in the way different sectors of the economy were affected, both across the two recessions and across the four countries, if the common cause was energy prices. Within each country, the industries hit hardest are quite dissimilar from one recession to the next and, and for each recession, the industries hit hardest are dissimilar across the four countries.

Another set of statistical tests examined the relative economic performance of different sectors to see if the effects of the shocks are more pronounced among the more energy-intensive sectors. The tests reveal no significant negative correlations between energy intensity and changes in output, employment, or capital formation for any of the four countries. Nor does the evidence suggest that adjustment costs caused by changes in the composition of final demand are more severe in energy-intensive sectors. Finally, in contrast with the rigid wages argument, changes in real wages appear to vary negatively

Figure 7
ANNUAL PERCENTAGE CHANGES IN OUTPUT AND EMPLOYMENT



Note: Output is measured in GNP for Germany and Japan and in GDP for the United States and the United Kingdom. Figures on Germany refer to western Germany.

Sources: OECD Department of Economics and Statistics, "Labour Force Statistics," 1989, 1990. IMF "International Financial Statistics Yearbook." 1992.

with energy intensity in the two shock periods, suggesting that wages were more responsive in labor markets where unemployment has been more serious.

These and other findings suggest that one should look to factors other than energy prices to explain the macroeconomic failures during the 1970's. The alternative hypothesis suggested in Bohi (1989) is that the industrialized countries were already combatting inflation when the oil price shocks hit and, except for Japan in 1979, further deflated their respective economies to mitigate increases in their general price levels. Given that Japan was the only industrial country to avoid a recession after the 1979 oil shock, it is plausible that the monetary authorities rather than energy prices are to blame for the recessions.

In empirical analyses of the recessions it is of course difficult to separate the influences of the monetary authorities from those of oil prices, and research to date fails to make a credible distinction.[34] A great deal more study is required before we can begin to understand the nature of energy-economy interactions at the national and regional levels. If nothing definitive can be said about the gross economic costs of energy price shocks, it follows that even less can be said about the magnitude of any embedded externalities that are relevant for comparing fuel cycles at a local or national level.

5. POTENTIAL EXTERNALITIES RELATED TO R&D

Most economists would agree that market signals alone do not generate a socially efficient level of investment in research for acquiring basic knowledge for the development of new technologies. The basic problem is that information has attributes of a public good, with benefits to many other agents beyond those who bear the costs of information acquisition (notwithstanding institutions such as patents and copyrights). Since those who bear the costs of information acquisition generally cannot appropriate all the benefits, too little information acquisition is undertaken (Cornes and Sandler 1986).

... energy security externalities and R&D externalities provides strong support for a government role in supporting R&D activities ...

The above argument applies to energy-related research in general, just as it does to research in other areas. There are, however, specific aspects of energy security which intersect with R&D externalities. As previously noted, research and

[34]For example, on this issue compare Hickman, Huntington, and Sweeney (1987), Sachs (1982), Bruno (1986), and Helliwell (1988).

development of cost-effective alternative energy sources and less energy-intensive technologies may be the strongest tools over the longer term for countering any market power exerted by energy exporters. Enhanced energy conservation and flexibility in energy storage provide means for mitigating any adjustment costs associated with energy price shocks. Thus the presence of both energy security externalities and R&D externalities provides strong support for a government role in supporting R&D activities, including R&D related to fuel cycles, even though the importance of the energy security spillovers remains unresolved. If the spillovers are important, the most effective government support should aim to increase the elasticities of supply and demand for oil by expanding the range of substitutes on both sides of the market.

The government has already responded to the public good argument with considerable support for energy R&D. In view of this effort, together with the contribution of existing patent and copyright laws, it is conceivable that the R&D externality has already been adequately addressed in the design of Federal policy.[35] Indeed, we have little basis for saying whether the level of effort is deficient or excessive.

6. CONCLUDING REMARKS

We have examined the potential externalities associated with the volume and price of oil imports, with variability in the price of oil, and with the private incentives for R&D that may be relevant to fuel cycle comparisons. Any spillovers related to the volume and price of oil imports are irrelevant to *incremental* fuel cycle decisions because taken individually these decisions may have a nonnegligible effect on the price. Collectively, however, these decisions may *not* have a negligible effect, which may justify national policy aimed at coordinating or guiding individual decisions. Nonetheless this reasoning provides only weak guidance for national policy regarding fuel cycles.[36]

With regard to variability in the price of energy, there is a plausible conceptual argument for the existence of externalities at local and national levels, especially associated with institutional and technological constraints on wage and price flexibility. However, it is difficult in practice to identify the extent of any

[35]The cost of the commitment is borne by tax payers, so it may not be adequately internalized into price signals considered by energy consumers. However, an empirical assessment of the cost of this commitment is unavailable at this juncture.

[36]We emphasize again that we have not addressed the full range of issues related to national import policies in this paper; see Bohi and Toman (1993). For a sharply contrasting view of the social costs of oil imports, see Greene and Leiby (1993).

externality relative to internalized adjustment costs. The empirical literature provides no information on the magnitude of the externality, and even wide disagreement about the magnitude of the gross adjustment costs. Finally, there is an unambiguous energy R&D externality, that is in part related to energy security, but the magnitude of any uninternalized residual is unknown. In short, there is not enough solid empirical information on R&D externalities to give an informed policy recommendation (other than to invest in economic research that will help resolve some of the ambiguities identified here).

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PAPER 17[1]

THE MEASUREMENT OF EMPLOYMENT BENEFITS

1. INTRODUCTION

The consideration of employment effects and so-called "hidden employment benefits" is one of the most confused and contentious issues in benefit-cost analysis and applied welfare economics generally. New investments create new employment opportunities, and often advocates for specific investments cite these employment opportunities as alleged benefits associated with the project. Indeed, from the local perspective, such employment opportunities may appear to be beneficial because they appear to come for free. If there is unemployment in the local area, then new investments create valuable employment opportunities for those in the local community. Even if there is full employment in the local area then new investments create incentives for in-migration from other locations that may have pecuniary benefits locally through increased property values, business revenues, etc.

The focus in this study is on net economic benefits from a broad national perspective.

However, in reality benefits that accrue locally often do not come for free, but displace employment opportunities in other locales or draw on public sector revenues collected on a broader regional basis. If a job created in community A causes a job to be lost in community B, then the net gain from the national perspective is zero. Similarly, higher property values and new business revenues in community A come at the expense of those who must pay those revenues. These gains are offset by losses elsewhere in the economy, and consequently are referred to by economists as transfer payments or pecuniary externalities.

The focus in this study is on net economic benefits from a broad national perspective. From this perspective, many of the alleged employment benefits at the local level are offset by lost benefits at other locales, and do not count as benefits according to economic theory.

[1]Based largely on a working paper by D. Burtraw.

The clearest example of this concerns the creation of a new employment opportunity in a competitive economy that is fully employed at the national and local level. In this case, a worker will be paid the value of her marginal product, that is, the value of her marginal contribution to the economic activity of the firm. If the new investment takes place, then the value of the worker in the new employment must be greater than in previous employment, but if the economy is fully employed then it is likely the difference between the value of the worker in these two jobs is small. For the worker to take the new job, her previous job and associated economic activity must be eliminated from the economy (unless it is filled by another worker, who leaves her previous job, etc.). So, on net, economic welfare is only slightly improved or essentially unchanged. The value of the worker's contribution to the economy through her new employment is offset by the lost value of the worker's contribution through her previous employment.

The misuse of the term "employment benefits" by advocates of new investment projects, particularly pertaining to large hydroelectric projects in the western states, fueled a reaction in the 1960s among academic economists that led to efforts to discredit the consideration of employment benefits generally. The pervasive argument in the academic community, especially during the period of relatively low unemployment in the 1960s, was that there are few if any real economic gains associated with employment benefits. This was popularized in the policy community by the notion that since the concept was so likely to be misused, it might be preferable to discredit its use at all.

From a theoretical perspective, neither the presumed inclusion of new employment opportunities as "hidden benefits," nor their exclusion, is a proper position to take in general. It is fair to say that most in the economics profession view the performance of labor markets as usually adequate and consequently, there may be few employment benefits associated with most new investments, from the perspective of net economic efficiency. However, this point of view is not doctrine or ideology, but rather a rebuttable presumption subject to empirical analysis.

This paper outlines a methodology for testing this rebuttable presumption with empirical data pertaining to labor markets that would be affected by a specific new investment. The theoretical question that is relevant is whether the social opportunity cost of new employment is less than the market wage. This would be the case, for example, if one expects unemployment or underemployment to persist in a specific region of the economy or occupational category affected by the new investment. In this case, new employment opportunities produce a net increase in social wealth rather than just a transfer of income.

The method that is used in this investigation is to estimate the social opportunity cost of hiring new workers by taking into account the relevant unemployment rate for those workers, and calculating the probability that the new employment will attract some workers from the pool of previously unemployed or under-employed. Since these unemployment rates and associated probabilities depend on the specific geographical region and occupational category that is affected, it is possible that some fuel cycles would generate employment benefits while similar investments in other fuel cycles would not.

The method...is to estimate the social opportunity cost of hiring new workers by taking into account the relevant unemployment rate for those workers, and calculating the probability that the new employment will attract some workers from the pool of previously unemployed or under-employed.

This paper illustrates this methodology in the evaluation of new investments in the coal fuel cycle in the Southeast and Southwest Reference environments. We find that only a small fraction of the employment impacts associated with the new investments would count as employment benefits. Nonetheless, these small fractions are significant compared to the magnitude of other externalities investigated in this study. However, it also bears repeating that similar benefits of varying maginitudes may accrue from investments in each of the fuel cycles. This methodology is applied to other fuel cycles, and the resulting estimates appear in the text of the fuel cycle documents.

2. SOURCE OF EMPLOYMENT BENEFITS

The term *shadow price* is used to describe the social value of goods and services that must be sacrificed in order to redirect resources to new activities.[2] Under the familiar assumption of perfectly competitive markets, shadow prices and market prices coincide. However, a wedge may be driven between the two if markets function imperfectly, for example, due to the failure of prices or wages to adjust to conditions of unemployment. When inputs to the production of energy services stand idle or under-utilized at their current market price or wage their market prices will not represent social costs. In this discussion we focus exclusively on labor markets and the possibility that workers are

[2]A seminal discourse of the use of shadow prices for investment decisions is found in Lind (1982).

previously unemployed or under-employed. In this case, society's opportunity cost of employing workers in new activities is less than their wage. Equivalently, it is sometimes stated that there are hidden benefits that result from new employment in this activity.

Unemployment is a transitional state (people typically do not remain unemployed forever), unemployment rates fluctuate, and they differ across occupational categories and regions. To account precisely for the extent to which the employment of labor services make use of previously under-utilized resources it would be necessary to trace each unit of labor employed to its source and to inquire into its alternative use. This discussion follows the general literature in proceeding under the assumption that there is insufficient information to allow such a precise accounting.[3] Instead we assert that it is sufficient to observe persistent unemployment (above the natural rate of unemployment) in relevant labor markets in order to conclude that employment benefits exist.[4]

Unemployment must be estimated for the relevant employment sector and region for each occupational category affected by the new investment.

Unemployment must be estimated for the relevant employment sector and region for each occupational category affected by the new investment. In principle, one would prefer to use statistical techniques to forecast unemployment into the relevant time horizon. A reasonable first-order approximation can be obtained through the use of long-run unemployment rates amended by information about investment and growth in the affected region.

The estimated unemployment rate will include an element that is sometimes termed "frictional unemployment," the "nonaccelerating inflation rate of unemployment" (NAIRU), and more generally, the "natural rate of unemployment." This natural rate reflects the expectation that at any one time there will always be a segment of the population that is in transition between jobs, perhaps looking for a new job or to acquire new skills. Recent estimates of the natural rate of unemployment range from 4.7 to 6.5 percent (although, in

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[3]This analysis utilizes a partial equilibrium approach, in which the labor market is modeled in isolation from other segments of the economy. (See Ward and Deren, 1991). A more rigorous technique is to construct a general equilibrium model that allows individuals to optimize in response to price changes and adjust their own behavior accordingly. (See Squire and van der Tak, 1975.) However, general equilibrium models of regional economies are unlikely to exist and those that do are unlikely to capture features of particular concern to the Fuel Cycle Study.

[4]See Haveman (1970), Gramlich (1981) or Sassone and Schaffer (1978) for additional exposition.

principle, they can vary by occupation and region).[5] Consequently, many economists describe a fully employed economy as one in which the unemployment rate is in this range. Persistent unemployment rates that are above this range reflect a shadow price (social cost) of labor services that is less than the market wage.

... a probability that a new worker was ultimately drawn from the pool of previously idle workers,...linking this probability with the prevailing rate of unemployment was developed by Haveman and Krutilla (1967).

When a new job is created in a local economy, due in the present circumstance to new investment in a fuel cycle, there is some probability that the worker who is hired will be drawn from the pool of previously unemployed workers, and some probability that the worker will be drawn from other existing employment. In the latter case, there is a probability that someone to fill the worker's old job will be drawn from the pool of previously unemployed workers, and some probability that, again, the worker will be drawn from another existing job. After this chain of possibilities is played out, there is a probability that a new worker was ultimately drawn from the pool of previously idle workers, or that some old job was eliminated from the economy. The general relationship linking this probability with the prevailing rate of unemployment was developed by Haveman and Krutilla (1967).

If P represents the probability of drawing a new worker from the pool of previously unemployed, (1-P) represents the probability that no previously unemployed worker will be hired. This latter probability can be used as a coefficient on observed market wages (W) to calculate the expected shadow value (expected social cost) of diverting workers from other employment.[6]

It is important to note that unemployed individuals also attach a positive value to their time (denoted L), even if it is not spent in the workplace. Some individuals may be providing productive services such as child care, others may be enjoying leisure. An appropriate estimate of the marginal value of an unemployed person's time should be included in order to fully account for the

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[5]See Johnson and Layard (1986). The range of estimates results from different theoretical formulations of the labor market. However, there is broad agreement that there has been a secular increase in the natural rate of unemployment since the early 1950s.

[6]Another cost that may be important is the cost of geographical relocation for a worker that has to move to new employment.

opportunity cost of new employment. The probability P can be used as a coefficient on this value to calculate the expected shadow cost (expected social cost) of diverting workers from nonemployment activities.

The full shadow cost of employing one additional worker is specified to be: $S = P L + (1-P) W$. Where P is greater than zero (and under the assumption that $L < W$) there can be said to be "employment benefits" because the social cost of labor services are less than their private cost.

New employment opportunities will have another effect on a regional economy. If new *primary* employment is created, for example in the construction of a electricity facility, *secondary* employment will be created as well, for example, in the manufacture of materials for the construction project. Finally, *induced* employment will result from the increment in the indirect demand for goods and services, for example, at restaurants servicing the persons employed in the primary and secondary sectors. This "respending effect" is often called the "multiplier effect" since the introduction of new investment will stimulate economic activity by some factor greater than one. A standard representation and calculation of these indirect effects is captured in input-output analysis that tracks the flow of dollars through the national or regional economy and that presents an estimate of the change in the demand for final goods and services as a function of new economic investment.

The multiplier effect does not necessarily represent an increase in social welfare. If the economy is fully employed prior to the introduction of a new investment and associated multiplier spending, then the new demand for final goods and services will be realized only through a decrement in previously supplied goods and services.[7] The concept of opportunity cost applies in precisely the same manner with regard to respending effects as to the primary employment effects of new investment. In order for the respending effects to generate a net social benefit, it must be the case that the shadow price or social value of labor is less than its market wage. In principle, one would want to take into account the possible existence of pockets of persistent unemployment in order to evaluate the net benefits of respending effects. In practice, information typically is not available at a sufficiently disaggregated level to allow one to trace the respending effects precisely. However, regional input-output analysis does allow one to trace respending effects through broad occupational categories. Information on unemployment rates in these broad occupational categories will allow one to calculate expected social costs to a first order approximation.

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[7]Or potentially through immigration of capital or labor.

3. DATA AND RESEARCH APPROACH

Data for the unemployment rate in thirty-seven industrial categories for the nation and for regions of the country were gathered from annual publications of *Employment and Earnings* and *Geographic Profile of Employment and Unemployment* published by the U.S. Bureau of Labor Statistics.[8] Table 1 presents numerical references to the data in these documents. Unemployment rates by industrial sector for the United States, for the East South Central region which includes the State of Tennessee, and for the Mountain region which includes the State of New Mexico, for selected years between 1981 and 1991, are presented in Tables 2, 3 and 4.[9] The averages of these data for each industrial category are presented in the final column of the tables. These averages are taken to represent a forecast of the persistent rates of unemployment over the 40-year life of the facility.

The year 1989 is selected as a modern estimate of a full employment period with low inflation. The year 1989 has the lowest average unemployment rate (5.3%) for the U.S. of any year in this period. Inflation for all items less food and energy was 4.5%, the midpoint between 1983 and the present when inflation ranged from 4% to 5% annually.[10] Unemployment rates observed in this year in each industry are taken to be industry-specific natural rates of unemployment.

Johnson and Layard (1986) survey recent estimates of the natural rate of unemployment ranging from 4.7% to 6.5%, and the average rate for 1989 (5.3%) falls well within this range. As an alternative and more conservative estimate, calculations are presented using the year 1987 as a base. The average unemployment rate for the U.S. was 6.2% in 1987, near the upper bound suggested by Johnson and Layard, while inflation was 4.0%, the lowest of any year in the decade.

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[8]These publications actually provide data for thirty-seven industries. The mapping from the reported industries to the thirty-seven industries we use, along with an index of the numerical references are presented in Table 1.

[9]The years 1982 and 1984 were omitted in these series in order to place greater weight on recent observations. These estimates should be viewed as illustrative of the methodology. More careful estimates can be obtained using an econometric model to forecast future rates of unemployment and growth.

[10]Energy is excluded because prices have been heavily influenced by international markets. Food is excluded because fluctuations are due in large part to weather.

Table 1. Matching of the input-output and unemployment table categories

Category in Input-Output Table	Category No. for reference to other tables	Category from which the data were taken in the unemployment data
Agricultural products and agricultural,	1	Farming, forestry and fishing*
Forestry and fishery products	2	Farming, forestry and fishing*
Coal Mining	3	Mining
Crude petroleum and natural gas	4	Mining
Miscellaneous mining	5	Mining
New construction	6	Construction
Maintenance and repair construction	7	Construction
Food and kindred products and tobacco	8	Food and kindred products
Textile mill products	9	Textile mill products
Apparel	10	Apparel and other textile products
Paper and allied products	11	Paper and allied products
Printing and publishing	12	Printing and publishing
Chemicals and petroleum refining	13	Chemicals and allied products
Rubber and leather products	14	Rubber and misc. plastic products

*Farming, forestry and fishing is an occupational category. All of the other unemployment categories are industrial sectors.

Table 1. Matching of the input-output and unemployment table categories (Cont'd)

Category in Input-Output Table	Category No. for reference to other tables	Category from which the data were taken in the unemployment data
Lumber and wood products and furniture	15	Average of Lumber and wood products and Furniture and fixtures
Stone, clay, and glass products	16	Stone, clay, and glass products
Primary metal industries	17	Primary metal industries
Fabricated metal products	18	Fabricated metal products
Machinery, except electrical	19	Machinery, except electrical
Electric and electronic equipment	20	Electrical machinery, equipment and supplies
Motor vehicles and equipment	21	Motor vehicles and equipment
Transportation equipment, except motor vehicles	22	Transportation equipment (include motor vehicle)
Instruments and related products	23	Professional and photo equipment, watches, etc.
Miscellaneous manufacturing industries	24	Manufacturing Average
Transportation	25	Transportation
Communication	26	Communication and other public utilities
Electric, gas, water, and sanitary services	27	Communication and other public utilities
Wholesale trade	28	Wholesale trade

*Farming, forestry and fishing is an occupational category. All of the other unemployment categories are industrial sectors.

Table 1. Matching of the input-output and unemployment table categories (Cont'd)

Category in Input-Output Table	Category No. for reference to other tables	Category from which the data were taken in the unemployment data
Retail trade	29	Retail trade
Finance	30	Finance, Insurance, and Real Estate
Insurance	31	Finance, Insurance, and Real Estate
Real estate	32	Finance, Insurance, and Real Estate
Hotels and lodging places and amusements	33	Services, excluding households
Personal services	34	Services, excluding households
Business services	35	Professional services
Eating and drinking places	36	Services, excluding households
Health services	37	Medical services, including hospitals
Miscellaneous services	38	Services, excluding household
Households	39	No corresponding entry

*Farming, forestry and fishing is an occupational category. All of the other unemployment categories are industrial sectors.

Table 2. United States unemployment rates by industrial sectors for the years 1981, 1983, 1985-1990*

Sector	1990	1989	1988	1987	1986	1985	1983	1981	Average
3	4.8	5.8	7.9	10.0	13.5	9.5	17.0	6.0	9.3
4	4.8	5.8	7.9	10.0	13.5	9.5	17.0	6.0	9.3
5	4.8	5.8	7.9	10.0	13.5	9.5	17.0	6.0	9.3
6	11.1	10.0	10.6	11.6	13.1	13.1	18.4	15.6	12.9
7	11.1	10.0	10.6	11.6	13.1	13.1	18.4	15.6	12.9
8	7.1	7.4	8.3	8.6	9.9	9.8	13.1	8.4	9.1
9	5.9	4.9	5.3	6.7	7.6	9.9	9.6	10.1	7.5
10	9.3	8.4	8.2	9.7	10.7	11.4	12.4	10.6	10.1
11	4.1	3.5	3.2	3.9	4.1	4.5	6.8	11.5	5.2
12	4.2	4.0	4.2	4.4	4.8	5.5	6.7	5.3	4.9
13	3.5	3.1	2.9	4.1	5.1	4.7	7.3	5.4	4.5
14	6.2	6.2	5.6	5.4	7.9	7.6	10.8	5.1	6.9
15	7.6	6.1	7.9	8.0	10.3	11.6	15.0	12.6	9.9
16	5.6	5.3	5.4	6.2	7.8	9.6	11.2	8.6	7.5
17	4.9	4.1	4.5	7.3	10.0	11.3	20.0	8.5	8.8
18	7.0	6.6	5.6	7.0	8.3	8.8	14.4	9.6	8.4
19	4.6	3.6	4.1	5.2	6.3	6.2	12.2	5.9	6.0
20	5.6	4.8	4.9	4.9	6.4	7.8	8.9	6.8	6.3
21	9.0	6.2	6.3	7.9	6.9	7.3	12.6	14.7	8.9
22	6.4	4.8	5.3	5.8	5.2	5.8	10.9	10.4	6.8
23	3.9	3.4	3.3	3.9	4.3	4.4	7.0	5.9	4.5
24	5.8	5.1	5.3	6.0	7.1	7.7	11.2	8.3	7.1

*Unemployment rates taken from Table 5, Geographic Profile of Employment and Unemployment, published by the Bureau of Labor Statistics, for years shown above.

Table 2. United States unemployment rates by industrial sectors for the years 1981, 1983, 1985-1990* (Cont'd)

Sector	1990	1989	1988	1987	1986	1985	1983	1981	Average
25	5.1	5.0	5.2	5.9	6.7	5.9	8.6	5.2	6.0
26	2.0	2.2	2.1	2.6	3.0	2.5	3.6	6.1	3.0
27	3.8	3.9	3.9	4.5	5.1	5.1	7.4	8.9	5.3
28	4.5	4.0	4.3	4.5	5.3	5.2	7.5	2.3	4.7
29	6.8	6.5	6.7	7.6	8.1	8.2	10.7	8.1	7.8
30	3.0	3.1	3.0	3.1	3.5	3.5	4.5	3.5	3.4
31	3.0	3.1	3.0	3.1	3.5	3.5	4.5	3.5	3.4
32	3.0	3.1	3.0	3.1	3.5	3.5	4.5	3.5	3.4
33	7.2	7.0	7.1	7.7	8.8	8.7	11.3	9.4	8.4
34	7.2	7.0	7.1	7.7	8.8	8.7	11.3	9.4	8.4
35	3.2	3.0	3.2	3.6	4.0	4.2	5.5	4.7	3.9
36	7.2	7.0	7.1	7.7	8.8	8.7	11.3	9.4	8.4
37	3.2	3.0	3.2	3.6	4.0	4.2	5.5	4.7	3.9
38	7.2	7.0	7.1	7.7	8.8	8.7	11.3	9.4	8.4

*Unemployment rates taken from Table 5, Geographic Profile of Employment and Unemployment, published by the Bureau of Labor Statistics, for years shown above.

Table 3. East South Central region unemployment rates by industrial sectors for the years 1981, 1983, 1985-1990*

Sector	1990	1989	1988	1987	1986	1985	1983	1981	Average
3	7.6	7.6	8.0	11.5	22.3	11.4	24.2	9.1	12.7
4	7.6	7.6	8.0	11.5	22.3	11.4	24.2	9.1	12.7
5	7.6	7.6	8.0	11.5	22.3	11.4	24.2	9.1	12.7
6	14.8	10.8	13.6	15.5	17.9	18.0	23.7	15.1	16.2
7	14.8	10.8	13.6	15.5	17.9	18.0	23.7	15.1	16.2
8	10.3	7.3	8.6	8.7	11.3	10.8	15.2	10.4	10.3
9	6.5	9.3	6.8	7.1	8.4	12.2	7.5	10.5	8.5
10	7.5	8.7	8.5	13.4	11.8	13.0	12.1	10.5	10.7
11	2.6	3.5	4.7	2.7	2.7	2.1	8.9	5.8	4.1
12	4.3	3.7	4.3	5.2	5.9	6.0	4.5	5.8	5.0
13	2.5	1.3	2.5	3.0	7.2	6.5	7.2	5.0	4.4
14	4.8	3.6	5.5	6.8	6.6	7.3	8.9	6.3	6.2
15	4.8	6.5	9.8	9.1	10.4	12.6	17.4	13.8	10.5
16	8.2	4.3	7.2	7.3	8.7	11.9	14.8	11.6	9.3
17	5.6	5.5	6.6	3.6	10.5	8.5	14.8	10.8	8.2
18	4.0	7.2	4.6	5.8	8.1	10.2	15.1	14.3	8.7
19	7.6	3.7	5.5	6.3	8.1	7.4	11.9	7.0	7.2
20	6.1	5.2	7.8	6.3	9.2	11.3	8.2	11.7	8.2
21	8.1	7.5	7.6	8.9	8.8	7.0	14.7	14.5	9.6
22	8.5	8.8	7.3	8.6	9.0	9.6	17.2	11.2	10.0
23	6.1	6.1	7.2	7.3	9.3	10.4	14.8	11.6	9.1

*Unemployment rates are taken from Table 5, Geographic Profile of Employment and Unemployment, published by the Bureau of Labor Statistics (BLS), for the years presented above. A blank cell indicates either that the value did not meet BLS reliability standards or that the category definition in that year was inconsistent with the rest of the years.

Table 3. East South Central region unemployment rates by industrial sectors for the years 1981, 1983, 1985-1990* (Cont'd)

Sector	1990	1989	1988	1987	1986	1985	1983	1981	Average
24	6.2	6.2	7.0	7.7	9.3	10.1	12.7	10.0	8.7
25	4.7	4.3	5.6	7.1	7.8	7.4	9.4	6.7	6.6
26	1.9	2.7	1.9	2.4	3.6	1.7	3.4		2.5
27	1.9	2.7	1.9	2.4	3.6	1.7	3.4		2.5
28	4.7	4.3	4.4	4.6	7.7	4.5	8.7	10.1	6.1
29	7.5	7.8	7.8	8.9	11.3	10.7	14.2	11.3	9.9
30	1.3	3.2	3.0	4.5	4.0	3.0	5.1	4.1	3.5
31	1.3	3.2	3.0	4.5	4.0	3.0	5.1	4.1	3.5
32	1.3	3.2	3.0	4.5	4.0	3.0	5.1	4.1	3.5
33	5.2	5.2	6.1	6.5	7.8	6.5	10.3	6.7	6.8
34	5.2	5.2	6.1	6.5	7.8	6.5	10.3	6.7	6.8
35	3.4	3.8	4.3	4.1	4.8	4.6	7.5	5.1	4.7
36	5.2	5.2	6.1	6.5	7.8	6.5	10.3	6.7	6.8
37	3.3	3.8	4.6	4.7	4.8	5.4	6.7	6.7	5.0
38	5.2	5.2	6.1	6.5	7.8	6.5	10.3	6.7	6.8

*Unemployment rates are taken from Table 5, Geographic Profile of Employment and Unemployment, published by the Bureau of Labor Statistics (BLS), for the years presented above. A blank cell indicates either that the value did not meet BLS reliability standards or that the category definition in that year was inconsistent with the rest of the years.

Table 4. Mountain region unemployment rates by industrial sectors for the years 1981, 1983, 1985-1990*

Sector	1990	1989	1988	1987	1986	1985	1983	1981	Average
3	3.3	4.6	5.9	9.6	12.4	8.9	14.1	4.2	7.9
4	3.3	4.6	5.9	9.6	12.4	8.9	14.1	4.2	7.9
5	3.3	4.6	5.9	9.6	12.4	8.9	14.1	4.2	7.9
6	11.1	12.5	14.8	14.7	14.8	13.9	17.9	12.8	14.1
7	11.1	12.5	14.8	14.7	14.8	13.9	17.9	12.8	14.1
8	7.1	9.1	7.5	9.7	12.0	9.0	12.7	9.7	9.6
9	5.4	6.7	7.0	8.3	9.7	6.8	11.4	7.9	7.9
10	5.4	6.7	7.0	8.3	9.7	6.8	11.4	7.9	7.9
11	5.4	6.7	7.0	8.3	9.7	6.8	11.4	3.8	7.4
12	5.4	4.4	5.4	5.8	6.2	3.4	6.1	3.8	5.1
13	3.7	5.2	7.0	8.3	9.7	6.8	11.4	7.9	7.5
14	5.4	6.7	7.0	8.3	9.7	6.8	11.4	7.9	7.9
15	8.9	5.9	11.4	10.0	12.9	13.8	13.6	6.3	10.4
16	4.9	4.4	5.4	4.9	4.4	8.0	9.3	6.3	6.0
17	4.9	4.4	5.4	4.9	5.9	8.0	9.3	5.0	6.0
18	5.1	4.3	4.1	5.0	8.9	8.0	9.3	6.3	6.4
19	4.3	4.0	4.1	3.8	4.4	8.8	8.3	5.3	5.4
20	5.2	3.3	5.9	3.5	7.5	7.6	6.4	3.3	5.3
21	4.9	4.4	3.6	4.9	5.9	2.1	6.7	6.3	4.9
22	1.4	3.9	3.6	3.0	2.5	2.1	6.7	6.3	3.7
23	5.4	6.7	5.4	4.9	2.5	5.8	9.3	6.3	5.8

*Unemployment rates taken from Table 5, Geographic Profile of Employment and Unemployment, published by the Bureau of Labor Statistics, for years shown above.

Table 4. Mountain region unemployment rates by industrial sectors for the years 1981, 1983, 1985-1990*

Sector	1990	1989	1988	1987	1986	1985	1983	1981	Average
24	5.4	5.1	6.0	6.1	7.2	7.6	11.0	6.8	6.9
25	5.2	4.9	6.1	6.6	7.7	7.5	8.8	5.5	6.5
26	2.7	4.8	2.7	3.9	2.8	2.6	5.7	4.0	3.7
27	4.1	3.6	4.6	5.4	5.4	5.2	7.2	4.0	4.9
28	3.3	4.5	5.3	4.7	4.8	4.7	6.5	7.6	5.2
29	6.1	6.6	7.5	8.6	8.4	7.7	9.3	8.4	7.8
30	3.2	3.9	3.0	3.8	5.3	4.5	4.9	3.5	4.0
31	3.2	3.9	3.0	3.8	5.3	4.5	4.9	3.5	4.0
32	3.2	3.9	3.0	3.8	5.3	4.5	4.9	3.5	4.0
33	5.3	5.2	5.3	6.6	6.4	6.0	7.6	5.1	5.9
34	5.3	5.2	5.3	6.6	6.4	6.0	7.6	5.1	5.9
35	3.7	3.5	3.9	4.5	4.4	3.9	5.2	3.6	4.1
36	5.3	5.2	5.3	6.6	6.4	6.0	7.6	5.1	5.9
37	3.4	3.6	3.7	4.1	4.1	4.5	4.9	5.1	4.2
38	5.3	5.2	5.3	6.6	6.4	6.0	7.6	5.1	5.9

*Unemployment rates taken from Table 5, Geographic Profile of Employment and Unemployment, published by the Bureau of Labor Statistics, for years shown above.

Unemployment rates are based on rates for the East South Central and Mountain regions as a whole. Although the generation facilities are to be located within Tennessee and New Mexico respectively, the appropriate labor markets include portions of each entire region.

The year 1989 is selected as a modern estimate of a full employment period with low inflation.

A region may have persistently higher rates of unemployment than the national average in many sectors of the economy. Consequently employment benefits are likely to occur even when the nation is "fully employed." The high unemployment rate in the region would be dampened inappropriately by using base year (1989 or 1987) levels of unemployment for the region, so we use base year levels for the entire U.S.

The unemployment rates that we utilize pertain to the experienced civilian labor force. This will tend to exclude new entrants to the labor market. Also, unemployment numbers are reported by industry rather than by occupation. One occupation may appear in several industries; however, we were not working at a sufficient level of detail to enable us to map from occupation to industry. The existence or direction of a bias in the estimates from this limitation is unclear.

The construction and operation of the facilities in Tennessee and New Mexico will directly stimulate employment in two industries. One is the New Construction industry (6). We assume the construction phase will entail a $682 million "overnight" construction cost for a 500 MW coal facility in both the Southeast and Southwest Reference environments.[11] The concept of overnight construction cost excludes finance costs associated with regulatory policy governing cost recovery and the time profile of construction, and is based on 1990 dollars. To associate construction expenditures (and associated potential employment benefits) with the flow of electricity output from each facility they must be annualized over all the kilowatt-hours of electricity that are generated by the project. We do so using constant 1990 dollars and a real interest rate of 5%, resulting in an annual expense of $39.77 million in the New Construction industry.

Expenditures on the operation and maintenance of the facility have been divided according to their fuel component and all other expenditures.

[11]Construction, operation and maintenance costs are based on Tables A.10 and A.11, respectively, in *Annual Outlook for U.S. Electric Power 1991.*

Expenditures on the operation and maintenance of the facility are attributed to the Electricity, Gas, Water and Sanitary Services industry (27). We assume this industry will be stimulated with $35.45 million of production annually throughout the 40-year operation of the plant. The fuel expense is attributed to the Mining industry (3), and will entail about $46 million per year for each facility.[12]

Expenditures in these three industries constitute direct (or primary) demand for final goods and services. In addition, secondary demand is created in industries that supply inputs to these primary industries. Finally, induced demand is created in all other industries that create goods and services that will be sought by workers in the primary and secondary industries.[13]

Earnings multipliers for the U.S. reported in Table 5 reflect the increase in the total annual primary, secondary and induced earnings (wages) that are received by workers in all 38 industries for each dollar of expenditure in the three primary industries. These multipliers are taken from input-output tables provided by the Bureau of Economic Analysis of the U.S. Department of Commerce.[14]

The possible existence of employment benefits – that is, a difference between the opportunity cost of labor and its market wage – depends on the possibility that newly employed workers are drawn from the pool of workers who were previously involuntarily unemployed. A mathematical expression for

[12]The fuel costs per year at the two plants differ slightly, totaling $46.21 million at the Tennessee facility and $46.12 million at the New Mexico facility. New Mexico coal is assumed to be $25/ton and Kentucky coal is assumed to be $35/ton, but the New Mexico coal has a lower heating value. Thus more coal needs to be mined to attain the same energy production (1.9 million tons for the New Mexico site as compared to 1.365 million tons for the Tennessee site).

[13]A possibility for error results from the assignment of expenditures in the primary industries. Expenditures for the ongoing operation of the facility have been broken into two parts, a fuel component and an operations and maintenance component, corresponding to two different categories of expenditures. The fuel component is viewed as an expenditure in the Crude Petroleum and Natural Gas industry, while the operations and maintenance is viewed as an expenditure in an industry that includes electric, gas, water and sanitary services. Some part of an expenditure in the latter industry includes expenditures for fuel, although it will be an average of a variety of fuels from a variety of locations.

This double-counting is expected to be minimal because the primary expenditure is not counted twice, but rather how parts of the primary expenditure are spent may be counted twice while other parts are under-counted. On net, it is unclear whether this error will serve to inflate or deflate the estimate of employment benefits.

[14]See Regional Multipliers: A User Handbook for the Regional Input-Output Modeling System (RIMS II), May 1986. Data for unemployment in households are not relevant and they are not excluded. More detailed input-output tables disaggregated for 531 industries may be obtained from the Bureau of Economic Analysis.

this function for the midpoint estimate, and for a low and high estimate, is presented in Table 6. There actually are a family of functions behind each estimate, one for each industry. The function depends on the difference between the average unemployment rate for each industry and the base year 1989 unemployment rate (the natural rate of unemployment).[15] The values for this function for each industry in the United States and the East, South, Central and Mountain regions are presented in Table 7.

In each case, if some percentage of the newly employed workers will be drawn from the pool of previously idle workers, the market wage will be an overestimate of the social opportunity cost of employment. This difference is the net new employment benefit that we seek to measure. A preliminary estimate of the employment benefits associated with each expenditure in a primary industry is obtained by multiplication of the total earnings using the earnings multipliers by the probability that workers are drawn from the pool of previously unemployed workers, and summing for all thirty-seven industries.[16] Finally, this estimate must be adjusted to reflect the opportunity cost of time for unemployed workers. We return to this issue below.

... if some percentage of the newly employed workers will be drawn from the pool of previously idle workers, the market wage will be an overestimate of the social opportunity cost of employment. This difference is the net new employment benefit that we seek to measure.

An important feature that differentiates regional earnings multipliers and multipliers for the United States is referred to as the "leakage effect." Some portion of economic activity that is stimulated by the demand for goods and services in a region leaks over that region's borders and stimulates secondary and induced activity in other states. Consequently, multipliers for an individual State will necessarily be smaller than multipliers for a region, which will be smaller in turn than for the entire United States.

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[15]When this difference is negative, it is set equal to zero.

[16]Krutilla and Haveman (1968, p. 75) cite Marglin (1962) on this point. "[The] appropriate shadow wage rate is the marginal opportunity cost of the force actually drawn from alternative employment [the market wage rate] multiplied by the percentage which this force forms of the total labor employed in this category..." (p. 51).

Table 5. United States input-output coefficients (multipliers) for earnings to industrial sectors*

Sector No.	(3) Coal Mining	(6) New Construction	(27) Elect. Services
1	0.0077	0.0108	0.0050
2	0.0002	0.0004	0.0001
3	0.3116	0.0032	0.0278
4	0.0055	0.0069	0.0229
5	0.0013	0.0039	0.0008
6	0.0000	0.3027	0.0000
7	0.0167	0.0164	0.0329
8	0.0090	0.0121	0.0058
9	0.0026	0.0044	0.0016
10	0.0043	0.0058	0.0027
11	0.0038	0.0062	0.0027
12	0.0071	0.0105	0.0052
13	0.0126	0.0167	0.0111
14	0.0070	0.0074	0.0033
15	0.0033	0.0167	0.0021
16	0.0027	0.0179	0.0019
17	0.0071	0.0202	0.0035
18	0.0094	0.0287	0.0054
19	0.0305	0.0148	0.0080
20	0.0078	0.0174	0.0048

*These are the earnings multipliers extracted from the three columns of interest in a 39x39 input-output table provided by the Bureau of Economic Analysis of the U.S. Department of Commerce.

Table 5. United States input-output coefficients (multipliers) for earnings to industrial sectors* (Cont'd)

Sector No.	(3) Coal Mining	(6) New Construction	(27) Elect. Services
21	0.0029	0.0038	0.0021
22	0.0015	0.0019	0.0013
23	0.0015	0.0029	0.0015
24	0.0015	0.0023	0.0010
25	0.0250	0.0399	0.0310
26	0.0093	0.0139	0.0065
27	0.0083	0.0088	0.1117
28	0.0405	0.0559	0.0228
29	0.0381	0.0662	0.0251
30	0.0186	0.0225	0.0155
31	0.0130	0.0172	0.0118
32	0.0038	0.0042	0.0028
33	0.0059	0.0084	0.0039
34	0.0061	0.0084	0.0048
35	0.0407	0.1044	0.0302
36	0.0138	0.0195	0.0093
37	0.0360	0.0479	0.0227
38	0.0182	0.0253	0.0116
39	0.0022	0.0030	0.0014
TOTAL	0.73770	0.9796	0.4649

*These are the earnings multipliers extracted from the three columns of interest in a 39x39 input-output table provided by the Bureau of Economic Analysis of the U.S. Department of Commerce.

Table 6. Functional forms to describe the probability that a newly employed person was drawn from the pool of previously unemployed

$$\textit{Low}: \quad P = 1 - \sin\left\{\frac{\pi}{2}(1-\beta_i)\right\}$$

$$\textit{Mid}: \quad P = 0.5\left[\sin\left\{\pi\beta_i - \frac{\pi}{2}\right\} + 1\right]$$

$$\textit{High}: \quad P = \sin\left\{\frac{\pi}{2}\beta_i\right\}$$

where

$$\beta_i = \frac{v_i - h_i}{h^* - h_i}$$

and where

P = probability

i = pertains to industry i

h_i = natural rate of employment in industry i

h^* = 0.25 for all industries. At this rate of unemployment it is assumed that a new job directly or indirectly draws a person from the pool of previously unemployed with probability one

v_i = unemployment rate for industry i

Table 7. Percent probability that newly employed person was drawn from the pool of previously unemployed (base year 1989).*

	United States			East South Central Region			Mountain Region		
Sector No.	low	mid	high	low	mid	high	low	mid	high
3	4.1	8.0	28.3	15.6	28.7	53.6	1.4	2.9	16.9
4	4.1	8.0	28.3	15.6	28.7	53.6	1.4	2.9	16.9
5	4.1	8.0	28.3	15.6	28.7	53.6	1.4	2.9	16.9
6	4.7	9.2	30.3	20.2	36.3	60.3	8.9	17.0	41.3
7	4.7	9.2	30.3	20.2	36.3	60.3	8.9	17.0	41.3
8	1.4	2.8	16.8	3.4	6.7	25.8	1.9	3.8	19.5
9	2.2	4.3	20.7	4.0	7.9	28.0	2.7	5.4	23.2
10	1.4	2.9	17.0	2.3	4.6	21.5	0.0	0.0	0.0
11	0.2	0.5	6.8	0.1	0.2	4.6	4.0	7.9	28.0
12	0.2	0.5	6.7	0.3	0.5	7.2	0.3	0.6	7.9
13	0.5	1.0	9.8	0.4	0.9	9.3	4.9	9.6	31.0
14	0.7	1.3	11.5	0.0	0.0	0.2	1.0	2.0	14.2
15	4.9	9.6	31.0	6.7	12.9	35.9	6.2	12.0	34.6
16	1.5	2.9	17.2	4.9	9.6	31.0	0.1	0.3	5.2
17	6.2	12.1	34.8	4.8	9.4	30.6	1.0	2.0	14.0
18	1.2	2.4	15.4	1.5	3.1	17.5	0.0	0.0	0.0
19	1.6	3.1	17.6	3.4	6.8	26.0	0.8	1.7	13.0

*United States unemployment rates for 1989 were taken to represent the natural unemployment rates for these industries.

Table 7. Percent probability that newly employed person was drawn from the pool of previously unemployed (base year 1989).* (Cont'd)

Sector No.	United States			East South Central Region			Mountain Region		
	low	mid	high	low	mid	high	low	mid	high
20	0.6	1.3	11.3	3.5	6.9	26.3	0.1	0.2	4.2
21	2.5	4.9	22.1	4.1	8.0	28.3	0.0	0.0	0.0
22	0.6	1.3	11.3	8.1	15.6	39.5	0.0	0.0	0.0
23	0.3	0.5	7.4	8.5	16.2	40.3	1.5	3.0	17.3
24	1.2	2.4	15.4	3.9	7.6	27.7	1.0	2.0	14.2
25	0.3	0.7	8.3	0.8	1.6	12.7	0.7	1.5	12.0
26	0.0	0.1	2.3	0.0	0.0	2.2	0.5	0.9	9.6
27	0.0	0.1	2.2	0.0	0.0	2.2	0.5	0.9	9.6
28	0.3	0.6	7.8	1.3	2.5	15.8	0.4	0.8	8.8
29	0.6	1.2	11.0	4.2	8.3	28.8	0.6	1.3	11.2
30	0.0	0.0	2.2	0.0	0.1	3.0	0.2	0.4	6.5
31	0.0	0.0	2.2	0.0	0.1	3.0	0.2	0.4	6.5
32	0.0	0.0	2.2	0.0	0.1	3.0	0.2	0.4	6.5
33	0.7	1.5	12.2	0.0	0.0	0.0	0.0	0.0	0.0
34	0.7	1.5	12.2	0.0	0.0	0.0	0.0	0.0	0.0
35	0.2	0.4	6.6	0.7	1.5	12.1	0.3	0.6	7.8
36	0.7	1.5	12.2	0.0	0.0	0.0	0.0	0.0	0.0
37	0.2	0.4	6.6	1.0	2.0	14.2	0.4	0.7	8.4
38	0.7	1.5	12.2	0.0	0.0	0.0	0.0	0.0	0.0

*United States unemployment rates for 1989 were taken to represent the natural unemployment rates for these industries.

The preferred analysis should match earnings multipliers with unemployment rates for the relevant geographic labor market to estimate net new employment benefits. Typically, State boundaries do not correspond well with relevant labor markets, which are regional in nature. However, earnings multipliers for specific regions were not available to us given available resources. Therefore we rely on earnings multipliers for the entire United States. We examine the sensitivity of this assumption by looking also at estimates based on multipliers derived for the State level.

The approach we use combines unemployment rates for a region with national multipliers. This will result in an over-estimate of true benefits if the region has higher than average unemployment, because some of the earnings will accrue in other areas where unemployment may be lower and hence, net employment benefits would be lower. Conversely, this approach will under-estimate true benefits if the region has lower than average unemployment.

The earnings multipliers presented in Table 5 are for total earnings and capture the sum of primary, secondary, and induced effects. These multipliers are an overestimate for our purposes because they reflect the induced effects for all the primary earnings that result from this investment. To a significant degree these "respending" effects would have occurred anyway as a result of activity that would engage many workers in the primary industries in their next best employment opportunity. A proper accounting should only include the induced effects that result from net new earnings in the primary and secondary industries. To correct for this problem, we reduce all earnings that occur outside the three primary industries by the ratio of net new earnings over total new earnings that occur in the primary industries, which represents that portion of induced earnings that are new.[17]

The estimates that are obtained by the model described thus far are an intermediate step in the final estimate because they do not account for the social opportunity cost of workers previously unemployed. Estimates of the opportunity cost of time for unemployed workers are usually expressed as a fraction of their previous market wage. Unfortunately, existing estimates must be viewed as unreliable for our purposes. Estimates depend on a number of factors including the possibility of flexible working hours (most jobs have a fixed work schedule), whether the worker can maintain a subsistence lifestyle during periods of unemployment, costs of searching for employment, the activities of the worker during periods of unemployment, and the location of

[17]This approach underestimates secondary effects. Secondary industries are more closely linked to primary industries and are stimulated by the demand for new output in the primary industries, rather than by just the net new respending effects.

residence.[18] Since a worker typically cannot make marginal adjustments to working hours, the true opportunity cost of the worker's time may include an element of consumer surplus flowing from activities during unemployment.

We examined three alternative estimates of the opportunity cost of an unemployed worker's time in order to illustrate the range of plausible estimates.[19] The most conservative approach is to value time for unemployed workers at after-tax wages. Conversion of before-tax to after-tax rates requires information about the average rates applied to new earnings for each household, which is unavailable. Instead, the 1987 average tax rate on individual income (Federal and State income taxes and FICA) of 20.8 percent was used.[20] This estimate provides an extreme lower bound on employment benefits, which are equivalent to net new contributions to the public treasury. This estimate is unreliable for at least one important reason. Assuming that the market wage, a worker's reservation wage, and the opportunity cost of time are equal is logically inconsistent with observed persistent unemployment and apparent labor market disequilibrium.

... value time for unemployed workers at the estimated minimum wage after taxes.

The second approach is to value time for unemployed workers at the estimated minimum wage after taxes. We calculate that the 1990 after-tax minimum wage was about \$3.42 or about 26% of average hourly earnings across all industries in our model.[21] The third approach is to value the opportunity cost of time at zero. This number is also unreliable, but it serves to provide an upper bound on net employment benefits.

[18]For a recent survey of similar issues and the underlying theory, see Shaw (1992).

[19]See Harrington, et. al., 1991, p.106-108.

[20]Calculated from the U.S. Bureau of the Census, 1991, 1991 Statistical Abstract, 111th edition, Washington, DC. The tax rates for 1990 remain virtually unchanged.

[21]Calculated from the U.S. Bureau of the Census, 1991, 1991 Statistical Abstract, 111th edition, Washington, DC. The tax rates for 1990 remain virtually unchanged. One reason a number of this magnitude appears reasonable is that it is similar to the estimate of 30% for the opportunity cost of workers' time in commuting as a function of their hourly earnings that has emerged as focal from a voluminous literature. See for example, Bruzelius 1979. Admittedly, many features vary between the commuting and unemployment contexts, so estimates of the value of a person's time in commuting are poor for direct comparison with unemployment.

4. ESTIMATES OF EMPLOYMENT BENEFITS

Employment benefit estimates are summarized in Table 8 for the most persuasive set of assumptions that we identified. These assumptions include using 1989 as a base year index for the natural rate of unemployment, and using unemployment rates for the East South Central region for the Southeast facility, and unemployment rates for the Mountain region for the Southwest facility. The opportunity cost of a person's time who was previously involuntarily unemployed is taken to be the after-tax minimum wage. The significance of these and other assumptions on the estimates that are obtained will be discussed further below.

Each entry in Table 8 represents the monetary value of net new employment benefits (primary, secondary and induced) for the entire U.S. resulting from expenditures in each primary industry. The first three rows for each reference environment pertain to the three primary industries—Coal Mining, New Construction and Electric Services—and the fourth row is their sum. Benefits are presented as mills per kilowatt-hour of electricity that would be produced at the proposed facility. Within each row, three numbers are presented. These are the low estimate, mid estimate, and high estimate, which vary according to the three specifications of the probability function [see Table 5].

The model projects that the mid estimate of benefits across all industries due to all spending associated with the project is 2.08 mills/kWh for the Southeast Reference environment and 0.58 mills/kWh for the Southwest Reference environment. These numbers are our preferred midpoint estimates of net new employment benefits according to our analysis.

The estimate for the Southeast Reference environment is comparatively large. This is due to persistent high unemployment in the New Construction and Mining industries in the East South Central region relative to other parts of the country.

We have calculated estimates based on alternative assumptions in order to determine the sensitivity of results to each assumption and to provide a judgmental ninety percent confidence interval for this benefit estimate. The assumption about the opportunity cost of an unemployed person's time may be most critical. This opportunity cost is expressed as a function of the market wage, so the employment benefit is changed in direct proportion to the that is made. The estimates we present are based on an opportunity cost estimate that is 26% of the market wage on average. Instead, if one assumes an opportunity

Table 8.
United States employment benefits (mills/kWh)[a]

Sector of output	Low	Mid	High
Southeast facility			
Coal Mining	0.52	0.99	2.13
New Construction	0.58	1.09	2.25
Electric Services	0.00	0.00	0.03
Total	**1.10**	**2.08**	**4.42**
Southwest facility			
Coal Mining	0.05	0.09	0.63
New Construction	0.25	0.48	1.37
Electric Service	0.00	0.01	0.12
Total	**0.30**	**0.58**	**2.12**

[a]1989 Base year natural rate of unemployment and regional unemployment rates.

cost equal to the after-tax wage, then the midpoint estimate of net new earnings is reduced to obtain a comparable point estimate of 0.58 mills/kWh in the Southeast and 0.16 mills/kWh in the Southwest Reference environments. If an opportunity cost of zero is assumed, then net new benefits equal net new earnings, or 2.81 mills/kWh in the Southeast and 0.79 mills/kWh in the Southwest Reference environments.

The next most critical assumption is the choice of 1989 as base year for the natural rate of unemployment. The choice of a base year index is important to the estimates that are obtained because the probability function is nonlinear. A small change in the definition of the natural rate of unemployment for each industry has a relatively large effect on the estimate of employment benefits. Using 1987 as an alternative base year, in the Southeast Reference environment the midpoint estimate of total employment benefits as a result of expenditures summed for all three primary industries is 1.01 mills/kWh, or about one-half of that obtained when 1989 is used as a base year. Using 1987 as the base year in the Southwest Reference environment produces a midpoint estimate of 0.23 mills/kWh, about four-tenths of that obtained when 1989 is used as a base year.[22]

A small change in the definition of the natural rate of unemployment for each industry has a relatively large effect on the estimate of employment benefits.

The use of unemployment rates for the entire U.S. produces an estimate of 0.53 mills/kWh in the Southeast Reference environment, about one-quarter of that obtained with regional unemployment rates. The reason is that higher than average unemployment rates in New Construction and Mining in the East South Central region leads to higher benefit estimates from a regional perspective. In the Southwest an estimate of 0.53 mills/kWh is obtained, about nine-tenths of that obtained with regional unemployment rates.

In order to construct a reasonable confidence interval for the point estimate of employment benefits, one can not in general combine reasonable conservative or generous assumptions for each relevant parameter and feed these

[22]A comparison based on percentage change is somewhat misleading because the absolute change is much greater in the Southeast Reference environment than in the Southwest Reference environment. The percentage change is great in the Southwest because the difference between the projected unemployment rate and the natural rate is already small in the base case. Substituting 1987 unemployment numbers for the natural rate of unemployment effectively moves the relevant value further out on the flat part of the S-function.

into the model. The actual level of confidence that is generated by combinations of assumptions depends in a complicated way on the nature of the underlying probability distributions. Absent the resources to conduct a sophisticated uncertainty analysis using Monte Carlo sampling methods, we have to exercise judgment to identify the endpoints for a 90% confidence interval.

In the Southeast, for the upper bound of 90% confidence interval for employment benefits we select 4.42 mills/kWh, the high point estimate presented in Table 8 under the basic set of assumptions. For the lower bound we select 1.01 mills/kWh, the mid point estimate when 1987 is used as the base year natural rate of unemployment. Note that this is close to the 1.10 mills/kWh obtained as a low estimate under the basic set of assumptions. In the Southwest, for the upper bound we select 2.12 mills/kWh, the high point estimate presented under the basic set of assumptions. For the lower bound we select 0.23 mills/kWh, which is the midpoint estimate when 1987 is used as the base year natural rate of unemployment. Note that this is close to the 0.30 mills/kWh estimate of a lower bound under the basic set of assumptions. These results are presented in Table 9.

Our reasoning for the selection of our preferred midpoint estimate and the 90% confidence interval is the following. The set of assumptions for this example are the most persuasive in our judgment. These include the use of 1989 unemployment rates by industry as the definition of an index of the natural rate of unemployment because this rate is closer to the midpoint of the range provided in the literature. For almost two-thirds of the industries the unemployment rate in 1989 across the U.S. was the lowest observed in the decade, while 1989 was also within a period of low inflationary pressures. The inflation rate in 1989 was the midpoint observed between 1983 and the present. We note that the natural rate of unemployment is not likely to be much less than this index provides. On the other hand, the calculation of projected unemployment rates is fairly conservative. We have truncated the observations that are the basis for this projection at 1981, leaving out many years of relatively high unemployment in the 1970s, and ignoring the period of relatively high unemployment since 1991. In addition, we have assigned greater weight to observations toward the end of the last decade, a period of unusually stable economic performance.

The possibility that the estimate we obtain is an underestimate is captured by the high value for the function under the strongest assumptions we can make about other variables. This bound reflects, in particular, the great uncertainty in this analysis surrounding the actual search process for employment opportunities. It also stands in for the possibility that unemployment could be greater than we have projected, which we think is more likely than for it to be less than we projected.

Table 9. Best estimate and judgmental ninety percent uncertainty range for United States employment benefits (mills/kWh)[a]

	Low	Mid	High
Southeast facility			
Total	1.01	2.08	4.42
Southwest facility			
Total	0.23	0.58	2.12

[a]Judgmental ninety percent uncertainty range is based on a combination of assumptions. See text for explanation.

The possibility that we have obtained an overestimate is captured by the alternative assumption using 1987 as a base year for the natural rate of unemployment. It seems to us that the distribution of true values is somewhat skewed, and although the 1989 index is the expected value there is considerable probability mass around a greater index (the median of this distribution is greater than the mean, in our minds). This estimate also stands in for the possibility that the true opportunity cost of unemployed labor is greater than the after-tax minimum wage. It is noteworthy that these estimates are close to the low point estimate provided under the basic set of assumptions in Table 9. For the lower bound, we have chosen the smaller of the estimates obtained in the low estimate in the base case and the mid estimate when 1987 is used as the base year.

In the Southeast, the range of the confidence interval from 1.01 to 4.42 mills/kWh should be taken as a measure of the uncertainties that are embedded in this analysis. On the other hand, this range and our identified midpoint estimate of 2.08 mills/kWh indicate our confidence that employment benefits are significant. Similarly, in the Southwest the 90% confidence interval between 0.23 and 2.12 mills/kWh, and the midpoint estimate of 0.58 mills/kWh indicate that the true value is greater than zero. The midpoint numbers we obtain are high relative to other fuel cycles, particular for the Southeast Reference environment, due to the relatively high projected rate of unemployment for New Construction and Mining in the East South Central region. The midpoint numbers represent 7.6% of total (primary, secondary, and induced) earnings generated by this project in the Southeast and less than 2.1% in the Southwest. They represent 22.5% of the earnings in just the three primary industries in the Southeast and 6.4% in the Southwest.

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PAPER 18[1]

THE EXTERNAL COSTS OF LOW PROBABILITY-HIGH CONSEQUENCE EVENTS: EX ANTE DAMAGES AND LAY RISKS

1. INTRODUCTION

This paper provides an analytical basis for characterizing key differences between two perspectives on how to estimate the expected damages of low probability - high consequence events. One perspective is the conventional method used in the U.S.-EC fuel cycle reports [e.g., ORNL/RFF (1994a,b)]. This paper articulates another perspective, using economic theory. The paper makes a strong case for considering this approach as an alternative, or at least as a complement, to the conventional approach. This alternative approach is an important area for future research.

Interest has been growing worldwide in embedding the external costs of productive activities, particularly the fuel cycles resulting in electricity generation, into prices. In any attempt to internalize these costs, one must take into account explicitly the remote but real possibilities of accidents and the wide gap between lay perceptions and expert assessments of such risks.

In our fuel cycle analyses, we estimate damages and benefits[2] by simply monetizing expected consequences, based on pollution dispersion models, exposure-response functions, and valuation functions. For accidents, such as mining and transportation accidents, natural gas pipeline accidents, and oil barge

[1]Largely based on an earlier paper by A. J. Krupnick, A. Markandya, and E. Nickell which was prepared for the American Agricultural Economics Associations Meetings, Orlando, Florida, 1993. The earlier version benefitted from valuable comments by A. M. Freeman. The authors thank the following members of the U.S.-EC fuel cycle teams who have commented on various ideas developed in the earlier version: M. Dreicer, G. Michaels, and R. Lee.

[2]By damage and benefit we mean the willingness to pay to avoid the effects (or to pay for the benefits) associated with the new plant (and the rest of the fuel cycle that supports it).

accidents, we use historical data to estimate the rates of these accidents. For extremely severe accidents--such as severe nuclear reactor accidents and catastrophic oil tanker spills--events are extremely rare and they do not offer a sufficient sample size to estimate their probabilities based on past occurrences. In those cases the conventional approach is to rely on expert judgments about both the probability of the consequences and their magnitude.

As an example of standard practice, which we term here an expert expected damage (EED) approach to estimating damages, consider how evacuation costs are estimated in the nuclear fuel cycle report [ORNL/RFF (1994a,b)]. Out-of-pocket costs per day of evacuation from a serious accident at a nuclear power plant ($27/person) are multiplied by the 53,000 people expected to be evacuated for 7 days. This calculation gives the value of this aspect of an accident at about $10 million. Multiplying by the expert-assessed annual probability of that accident (6.2×10^{-5}, the estimate in the U.S. study), the estimate of expected damage from this event is $620, or about one cent per person.

When this approach is followed, the implicit suggestion is that if about 1 cent were offered annually to each of those individuals in the evacuation area for accepting the risk of this accident, they would be fully compensated for the evacuation-cost component of that risk.[3] There are some obvious and not so obvious problems with this reasoning. The most obvious problem is that there may be many more types of effects of an accident than an analyst can track and quantify. To the extent that some of these effects are missing, the money value derived will be too low. The psychic costs associated with the trauma of being evacuated

There are three main limitations from an economics perspective with what we term the expert expected damage (EED) approach when it is applied to low probability-high consequence events: (i) Ignoring risk aversion. (ii) Ignoring the ex ante perspective in individual decision making. (iii) Ignoring lay risk assessments.

[3]Here we are abstracting from health risks, damages to land and capital assets, etc.

and the fear that one might have been fatally dosed (even though one is not) are two examples of damages that are difficult to estimate, but potentially large. This problem, however interesting and important, is not the subject of this paper.

Rather, the problems addressed here may be described as a failure of the EED approach to account for individual preferences and for the context in which these preferences are expressed. In modern economics, the basis for the valuation of any commodity, including complex and non-market ones like health risks, is individual preferences as expressed in or inferred through market behavior, or as inferred through observing other types of behavior. Just as few would question the legitimacy of individual perceptions and preferences in determining the price of fast, sleek cars, there is legitimacy in permitting perceptions and preferences that influence individuals' assessments of risks and the value to them of avoiding such risks.

There are three main limitations from an economics perspective with what we term the expert expected damage (EED) approach when it is applied to low probability-high consequence events:

(i) Ignoring risk aversion. The EED approach assumes that actuarial value and satisfaction, what economists term "utility," are equivalent. In "risky" situations, however, study after study find that people need more money to compensate them for taking risks than the actuarial value of these risks (i.e., more than 1 cent in the case above). The reason is simply that people are adverse to taking risks, particularly of the type we are considering here.

(ii) Ignoring the ex ante perspective in individual decision making. We distinguish here between an ex ante approach and an ex post approach to making decisions when outcomes are uncertain. The ex post approach, which is part of the EED approach, assumes that individuals maximize the expected value of their welfare realized in alternative states[4]. However, economists have found more empirical support for individuals' maximizing expected utilities, which we term the ex ante approach, following Hammond (1981). The term "expected utility" is used because individuals are assumed to maximize the expected value of their utility over a state with, and a state

[4]In fact the valuation actually carried out is even less welfare based than that. Often it is simply the cost of an accident in terms of the direct impacts (value of loss of crops, costs of rehousing etc.) that are taken. These would not be equal to the welfare costs in terms of willingness to pay even if the event were certain.

without, the accident while accounting for the probability of each state occurring. This may be distinguished from the EED approach where one estimates the loss in satisfaction from the consequences of an accident if it occurred with certainty and then multiplies this amount by the probability that the accident will occur.

(iii) Ignoring lay risk assessments. The EED model as described above is based on expert assessment of the probabilities of uncertain events and on the magnitude of both these events and the consequences of the fuel cycle that might be considered relatively certain (e.g., releases of tritium from nuclear power plant operations). Studies have shown that the lay public believes these estimates are far too low. Further, for certain risks, the public has been shown to hold a complex, multi-attribute definition of those risks, encompassing much more than probabilities and consequences -- incorporating trust, controllability, dread and other concerns that are outside of the purview of an expert risk assessment.[5] The distinction between expert and lay assessments of risk does not mean that either is incorrect. However, it suggests that, since damages are estimated based on *individuals'* willingness to pay (WTP) to avoid risks, it is more appropriate to estimate damages based on lay perceptions of risk since their willingness to pay is based on *their* perceptions. This idea is equally applicable to all of the fuel cycles, though assessments of the nuclear fuel cycle are unusually sensitive to this distinction.

In theory, unless these issues are addressed, the sum of money estimated as the damage will not match the amount needed to fully compensate those potentially harmed. Accordingly, this paper offers an alternative paradigm, which we term an expected utility (EU) approach, that incorporates risk aversion, the ex ante perspective (i.e., expected utility maximization), and lay perceptions of risks. The EU model is then used to simulate the consequences for damage estimates of substituting the EU model for the EED model using the "state-dependent utility function approach" alluded to above. In this exercise, we use hypothetical values for the degree of risk aversion, the size of the loss if an accident occurs, the probability of an accident, and other parameters to investigate this relationship. We also simulate the differences between the two models using parameter values drawn from one of the U.S. fuel cycle reports (ORNL/RFF 1994b). Next, an attempt is made to use the appropriate, if very

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[5]Here, there is some overlap between ignoring risk aversion and ignoring lay risks, as some of the "extra-risk" attributes, such as "dread," may properly be classified under risk aversion.

inconclusive, literature to quantify the ratio of expert to lay risks. Finally, this information is used in another simulation analysis with the EU and EED models to investigate the effect of these differences on the damage estimates.

2. SCOPE OF THE ANALYSIS

To provide a specific context for the analyses, we consider the nuclear fuel cycle. As stated previously, the conceptual underpinning of the analysis applies to all fuel cycles. However, we expect the differences in estimates of damages to be markedly different in the nuclear fuel cycle, probably more so than in other fuel cycles[6].

We make an assumption that the public's preferences are additive and separable across stages of the nuclear cycle.[7] Such an assumption permits us to focus on a particular stage of this cycle to illustrate our approach and to compare it to estimates of damage derived from the EED approach, which necessarily views expected consequences at each stage as additive and separable.

For the rest of this paper, we will focus on the power plant operation/accident stage. In the context of the U.S.-EC fuel cycle studies, where the focus is on externalities from a new power plant, the damage valuation problem can be reframed as a siting problem. Conceptually, we ask the question: What is the individual willingness to pay to avoid a new power plant being sited at a given location (in our case, at a particular reference

[6]Why should the nuclear fuel cycle be used as an example? The answer lies in public preferences: "all things nuclear" seem to be held in the public mind with a particular dread, mystery, and awe that is lacking with "all things coal," or with the other fuel cycles. As will be seen below, differences between lay and expert risks associated with the operation of coal-fired power plants are small relative to such differences with respect to nuclear plants, and insofar as we can learn, no one is seriously concerned about a "severe" accident at a coal-fired power station to anywhere the degree that they are about nuclear accidents. Thus, public risk aversion to probabilistic events does not appear to arise for the coal or other fuel cycles to the extent that it does for the nuclear fuel cycle. This is not to say that the damages from a coal-fired power plant are less than those from a nuclear plant. Rather we are saying that risk aversion and the divergence between lay and expert risk assessments are not much of an issue for the coal cycle; hence, there is little payoff to using an EED model and an EU model to estimate damages in that fuel cycle.

[7]To estimate the damages and benefits associated with the entire nuclear cycle using the EU approach is a daunting, perhaps even a hopeless, task. It can be argued that the public views nuclear power risks holistically, not distinguishing in their assessments of risks between the various stages of the nuclear cycle, from mining to storage, and even including concern over nuclear proliferation and nuclear terrorism. If this is true, one should not apply the EU (or EED) model to each stage of the nuclear cycle and sum the resulting damage estimates. Rather, if nuclear power risks are viewed holistically, the EU model should be defined over all stages of the nuclear cycle simultaneously to derive a holistic damage estimate.

environment)? This willingness to pay would be conditioned on the individual's risk aversion, his or her knowledge about and understanding of the probabilities and magnitude of accidents, the effects of operational releases of radiation and other pollutants, other elements that would form this individual's assessment of risks, and the potential employment and economic benefits to be realized from the new plant.[8]

3. AN EXPECTED UTILITY MODEL OF POWER PLANT OPERATIONS AND ACCIDENT RISKS

3.1 DISCUSSION OF THE CONCEPT

The measure of interest for the EU approach is the WTP of all individuals (including managers, stockholders of affected businesses, etc.) to avoid a small risk of a nuclear accident that would be posed by the siting and operation of a nuclear power plant in a particular area. This is a siting problem, which can be portrayed as a choice between accepting a plant with compensation and rejecting it.[9]

Three papers provide models on this subject: Freeman (1989), Smith (1992), and Kunreuther and Easterling (1992). Each uses basically the same formulation, but Freeman's is the most straightforward. Following Freeman, we posit an EU model where individuals maximize expected utility over two states that are identical except that a power plant accident occurs in one state, but not in the other. The utility function in either state is assumed to be the same, but the preferences for goods differ, depending on whether an accident has occurred. The model abstracts from any time dimension. This relationship may be described as follows:

$$\max_{X}: E(U) = qU(X, A^*) + (1- q)U(X, 0)$$

subject to the budget constraint

$$M = pX$$

[8]We leave aside here the question of whether and to what extent employment and other economic effects may be considered externalities.

[9] There is a literature, discussed below, which concludes that individuals will not accept compensation in return for their approval for a waste repository located near them, at least until certain minimum standards of trust and public participation are met. Assume for now that such conditions are met.

where A^* are the adverse physical consequences of the event (which takes a value of zero if the event does not occur), X represents all other goods, $U(\cdot)$ is the utility function, q is the probability of the adverse event, M is (certain) income, and p represents a vector of (certain) prices over goods X. Given the utility function and budget constraints, there is an expression - called an indirect utility function $V(M,p,A)$ - that gives the maximum utility attainable given M, p, and A.

Because we are not at this stage dealing with the third limitation of EED -- that it ignores the issue of expert vs. lay risks - we assume that somehow individuals know q and A^*. Also assume that they have no opportunities for self protection or self insurance. If there were such opportunities, individuals could be made indifferent to these risks through market opportunities.

We address the first limitation of the typical EED approach - that it ignores risk aversion - by postulating utility functions that build in risk aversion. More income is assumed to result in higher utility levels but at a diminishing rate. Risk aversion is measured by the curvature of the utility function with respect to income. If the utility function were drawn as a straight line, this would mean that the individual is neither risk averse nor risk loving, since he would be indifferent to an equal chance of income increasing or decreasing by the same amount. This is precisely what is being assumed by the EED approach, where money and utility are being implicitly equated. With a downward curvature, risk aversion is implied - the individual loses more utility with a decrease in income than he gains with an increase in income.

We address the first limitation of the typical EED approach - that it ignores risk aversion - by postulating utility functions that build in risk aversion.

The second limitation of the EED approach is addressed by deriving expressions for damage using an ex ante perspective and an ex post perspective and performing simulations to ascertain how large these differences in damages are. With an ex ante approach, the value of reducing the consequences of an event to zero (the siting case) (i.e. A=0) is given as the maximum the individual would be willing to pay (W^*) before he or she knows whether the event will occur to leave the individual indifferent to the adverse event occurring (with probability q) or not (with probability 1-q). In equation form, this condition is:

$$qV(M, A^*) + (1-q)V(M,0) = qV(M-W^*,0) + (1-q)V(M-W^*,0) = V(M-W^*,0).^{10} \quad (1)$$

The ex post value on the other hand is given by qW', where

$$V(M, A) = V(M-W', 0) \quad (2)$$

In Fig. 1, a state-dependent utility function is drawn, with the relationship between V and M being plotted for $A = 0$ and $A = A^*$. The ex-ante and ex-post values are shown. Given the functional form for V it is possible to compare the values of qW' and W^*. In the paper by Freeman, an estimate of the ex post and ex ante values is also made but not by making a global comparison between the two, as given in equations (1) and (2). Instead he looks at the marginal WTP for a small change in A and compares the ex ante and ex post values associated with such a change. We would argue that the non-marginal approach is better, given the discrete nature of A. Hence we report below some sample results of parameter values that are realistic for the nuclear plant siting problem referred to above.

3.2 SIMULATIONS

Simulations using the Southeast Reference environment for a nuclear power plant serve as examples of the differences between ex ante and ex post estimates of marginal damages. Table 1 provides the parameters for the simulations. Table 2 presents the ratio of these estimates for two different hypothetical utility functions, each with a range of risk aversion built in.

Columns 2 and 3 of Table 2 apply to a particular utility function. In the first case, the elasticity of the marginal utility of income is constant, as is the coefficient of risk aversion *when the value of A (damages from the event) is set at zero*. However, if A is positive, the coefficient of relative risk aversion decreases with income, which is not consistent with the observed evidence. The second utility function has increased relative risk aversion, with the level of risk aversion depending on the level of income. Hence this function is more consistent with the evidence on risk aversion and income.

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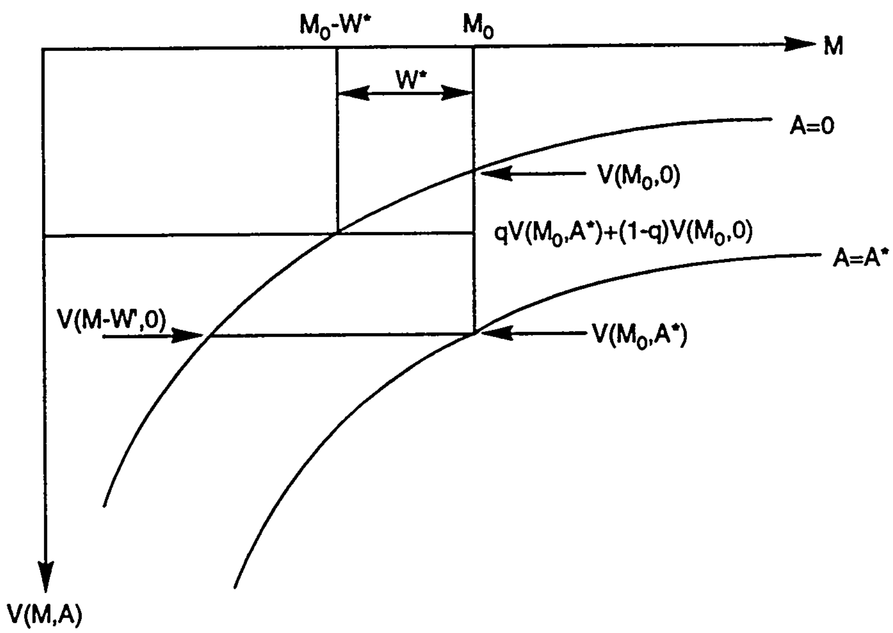
[10] Freeman shows that the rightmost expression is identical for the case of reducing the probability of the adverse event to zero.



W* is ex-ante WTP

qW' is ex-post WTP

A corresponding ex post WTP can be calculated.

Fig. 1. Ex ante and ex post WTP with a state dependent utility function

Table 1. Parameters for simulations of ex ante to ex post damage ratios, by utility function

Parameters	Function I	Function II
Utility Function	$V = (M(1-W))^b$	$V = -e^{-bM(1-W)}$
b	0.5	0.0001, 0.0005
r (risk aversion)	0.5	2, 10
M (income)	$24,807	
W (losses as % of income)	0.1%; 10%; 50% and 14% (see the ORNL/RFF Nuclear Fuel Cycle Report)	
q_e	0.001; 0.1; 6.2 x 10^{-5} (see the ORNL/RFF Nuclear Fuel Cycle Report)	
q_r	See Text: 0.0; 0.1; 0.5	
$\mu/(\mu + \beta)$	See Text: 0.01; 0.05; 0.25; 0.5; 1	

Each of the two sections in Table 2 provides the ratio of ex ante to ex post damage measures for an accident occurring with an expert probability defined in the first column. These probabilities are 0.5, 0.10, 0.001 and $6.2x10^{-5}$ [the latter being the probability of a serious reactor accident as estimated in the

Table 2. Ratios of ex ante to ex post marginal values of damages

Utility Function I V = (M(1-W))^b

Expert Probability	b	Relative Risk Aversion (r)	Ratio of Ex Ante to Ex Post Damages			
			W = 0.1%	W = 10%	W = 50%	W = 14%
0.500	0.50	0.50	1.000125	1.013167	1.085786	1.018843
0.100	0.50	0.50	1.000225	1.023700	1.154415	1.033919
6.20E-05	**0.50**	**0.50**	**1.000250**	**1.026332**	**1.171562**	**1.037685**

Utility Function II V = -EXP[-BM(1-W)]

Expert Probability	b	Relative Risk Aversion (r)	Ration of Ex Ante to Ex Post Damages			
			W = 0.1%	W = 10%	W = 50%	W = 14%
0.500	0.0001	2.50	1.000620	1.061859	1.292036	1.086391
0.100	0.0001	2.50	1.001117	1.119276	1.771140	1.171468
0.001	0.0001	2.50	1.001240	1.134800	1.978320	1.195377
6.20E-05	**0.0001**	**2.50**	**1.001241**	**1.134950**	**1.980599**	**1.195610**
0.500	0.0005	10.00	1.003100	1.292036	1.777119	1.388523
0.100	0.0005	10.00	1.005600	1.771140	6.316335	2.209759
0.001	0.0005	10.00	1.006221	1.978320	64.58300	2.687303
6.20E-05	**0.0005**	**10.00**	**1.006227**	**1.980599**	**78.24230**	**2.693193**

W is potential loss of income if an accident occurs expressed as a percent of total income.

For Utility Function II the ratio is dependent on the level of income as is the relative risk aversion.

Median Income is $24,807

U.S. Nuclear Fuel Cycle report ORNL/RFF (1994b)]. After simulations with a variety of parameter values for the utility functions, those presented under column b seem reasonably representative, with results being fairly insensitive to this parameter. The column labeled r shows the degree of risk aversion implied. Each simulation assumes a loss from the event as a percentage of income, using the same ranges as in the first set of simulations for three cases plus a fourth case for the loss estimated in (ORNL/RFF 1994b). The (median) income for a Tennessee household in 1990, $24,807, is used in the simulations and in computing the percentage loss for the fourth case.

Referring to the first three columns associated with each function, the basic story is that ex ante damages exceed ex post damages for the cases shown, and that this ratio is larger the greater the risk aversion (function II), the smaller the probability of the event and the greater the loss if the event occurs. The most extreme result is with an expert probability of 0.001, risk aversion of 10 and a loss of 50% of one's income if an accident occurs. Using the ex post measure of loss as opposed to the ex ante measure results in ex ante damages being 65 times greater than ex post damages. A more subtle result of utility function II is that the relationship of this ratio to the probability of the event is strongly dependent on the degree of risk aversion. When risk aversion is small (say 2.5), the ratio rises much more slowly with falling probabilities than it does when risk aversion is large (10). Another result, which is more intuitive, is that when the losses are small, the choice of utility function and parameter values is not very important, there being little difference between ex ante and ex post damages.

Turning to the simulations based on the U.S. Nuclear Fuel Cycle report, the probabilities of a serious accident is $6.2x10^{-5}$, divided into an LCF (limited containment failure) and an MCF (massive containment failure), with probabilities of $4.6x10^{-5}$ and $1.6x10^{-5}$, respectively. Losses conditional on occurrence of the event are computed by multiplying the impacts by the unit values discussed in that report. We focus these simulations on just the evacuation area, for purposes of illustration, and because the damages per household are largest. Health damage from an LCF to this population is $6 million and for an MCF it is $52 million, while the direct evacuation costs of an LCF are $0.2 million and of an MCF $14 million. Total damages are $72.2 million. Spread over 53,000 people, or (with 2.56 persons per household) 20,700 households, conditional damages total $3,488 per household. With median household income of $24,807, the loss represents 14 percent of income, the value in Table 1 for W. The relevant results for this scenario are set in bold in Table 2.

For the "plausible" parameters in the nuclear fuel cycle (with the potential expected loss of income being 14% of total income), the two utility functions provide very different estimates of the ratio of ex ante to ex post damages. These ratios range from 1.02 to 2.69, with greater risk aversion resulting in a larger ratio for function II. The shaded results can be used for a sensitivity analysis. The ratio rises rapidly for larger losses. Referring to function II, with risk aversion of 2.5, the ratio is 1.006 with a loss of 0.1% of income, rising to 1.98 for a 50% loss. This suggests that overestimating consequences contributes to less error than underestimating consequences when using the EED approach instead of the EU approach. Another interesting result is that the ratio is not particularly sensitive to the estimate of the probability of the event, irrespective of risk aversion and utility function assumed. Notice how the ratio of ex ante to ex post damage is virtually the same for a probability of 0.001 and a probability of 0.000062. This result suggests that, with respect to the comparison of the EED and EU approaches, complex probabilistic risk assessments (PRAs) that estimate probabilities in great detail are not very important with respect to the ratio of ex ante to ex post damage.

These simulations are quite similar to those of Freeman (1989) noted above. In the cases where the marginal value of reducing A increases with income (which holds for both utility functions given in Table 2) Freeman finds that the results depend significantly on the specific utility function assumed. Specifically, for catastrophic, low probability events, such as severe reactor accidents some utility functions lead to ex ante damages exceeding ex post damages by large percentages; others yield the reverse conclusion. Freeman concludes that without more information on preferences, we cannot predict the sign and magnitude of the difference between ex ante and ex post damages.

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4. EXPERT VERSUS LAY RISKS

There are always differences between experts' and lay people's perceptions of risks, whether those risks are of a dam break, natural gas pipeline explosion, or oil spill. Nowhere are these differences more crucial to the viability of a technology than in the nuclear fuel cycle. Probably the most salient element of the debate over the very *future* of nuclear power is the "disjuncture" (Kasperson 1992) between the risks of nuclear power as considered by experts engaged in the analysis of these risks and public perceptions of nuclear risks. The community of nuclear analysts, after highly detailed analyses of accident scenarios (called probabilistic risk assessments), have consistently found risks from nuclear power plant accidents and from other components of the fuel cycle to be exceedingly small. In contrast, the public has been shown to liken nuclear power plant accident risks to risks of nuclear war (Weart 1992), has appeared to ignore the expert estimates, and has consistently fought for diminished reliance on nuclear power and against the location of any nuclear activity in "their backyard," irrespective of compensation offered.

In this section we hope to lay out these different perspectives, explore the literature for quantifying differences between expert and lay risk estimates, and use the model developed in the previous section to examine the effect of these differences on damage estimates using the EED and the EU perspectives.

4.1 DEFINITIONAL ISSUES

This disjuncture between the experts and the public goes by a wide variety of descriptions, each with slightly different connotations, but each revealing important aspects of the problem. Kasperson (1992) refers to the disjuncture between "technical" and "social or perceptual" analysis of hazards (p. 155), characterizing the former as focusing on the probability of events and the magnitude of their consequences, with "risk" defined as the multiplication of the two. The latter term then refers to the "qualitative properties (newness, involuntary nature, catastrophic potential) of hazards that shape social experience and reactions," leading some analysts to call for modification of the technical calculations. However, Kasperson notes that no coherent framework for integrating technical and social aspects of risks has come forward.

Bradbury (1989) refers to "technical" risks as "objective" risks, dividing "social" risks into "psychometric risks" (which includes values and judgments of individuals) and "cultural" risks, which emphasizes the social construction of risks. Others simply use the dichotomy of objective versus subjective risks, or actual versus perceived risks. In the risk assessment (RA) debates at the Environmental Protection Agency, the same dichotomy is termed the "hard" path and the "soft" path to risk assessment.

Each of these designations conveys the impression that when the experts finish their analyses, there is a reasonable degree of certainty over the probability and consequences of various accidents while, on the other hand, these is almost no logic or rationale to assessments of risks by the public. These designations leave an impression that one perspective is correct and the other one not. This paper eschews making that judgment.

Evidence suggests that experts are not entirely objective, and that the public is not entirely irrational and subjective in their assessments of risks. For example, at the technical end, in developing probabilities of accidents, human error and human-machine interactions as well as materials deterioration are relatively new areas of study and not completely understood. In estimating consequences, the effects of radiation on health are subject to much uncertainty, given recent interpretations of cancer data from Japan. In addition, variability of weather patterns, uncertainties about the speed and scope of evacuations, and uncertainties about the food contamination pathways for radiation make an expert, "technical" risk assessment dependent on numerous assumptions based on the analyst's judgment.

Regarding this judgment, when it comes to risk perceptions, technical people may be just as vulnerable to the influence of "extraneous factors" in their judgments as any other group. For example, in a survey on how close different groups would be willing to live to a nuclear power plant, a coal-fired power plant, and various other types of facilities, Lindell and Earle (1983) found that nuclear engineers were willing to live closer to a nuclear facility than chemical engineers but that chemical engineers were willing to live closer to a coal plant than nuclear engineers! This finding suggests that mere familiarity diminishes risks even for technicians. Fischhoff's (1989) position is that while there are actual risks, no one knows exactly what they are, and that so-called objective risk estimates issued by experts contain much judgment.

At the same time, it is not at all clear that the public cannot be quite "objective" about environmental hazards, as long as they are relatively familiar ones. Morgan (1993) notes that lay people are capable of ordering a set of well-known hazards in terms of annual numbers of deaths. But, when the same question is asked in terms of the riskiness of less familiar hazards, they produce a different order.

This difference lies at the heart of the separation between expert and lay risk assessment. Researchers in this area assert that these differences arise because lay people, unlike experts, do not define risks solely in terms of expected number of deaths. As Slovic and others have shown in numerous surveys,[11] other attributes of risks are important components of public perceptions, including:

- expert assessments of impacts conditional on an event occurring;

[11]See Slovic [1992] for summaries and reflections about this work.

- some assessment of probability of the events that itself may be influenced by the way in which the public uses rules of thumb and various cognitive processes to interpret small probabilities (Kahneman and Tversky 1979);

- an assessment of risk attributes such as controllability, dread, voluntariness, equity (particularly intergenerational), and trust in the government or utility to deal with problems should they arise.

Given that the public's definition of risk is multifaceted, simply educating the public about expert risks will not have much effect on their overall assessment of risks unless the other components of public perceptions of risks are also addressed.

Other researchers (Kunreuther, Desvousges, and Slovic 1988) describe this phenomenon a bit differently. In their surveys in Nevada and elsewhere about individuals' willingness to accept (WTA) a nuclear waste repository at various sites, the researchers find that individuals who were willing to accept some compensation for accepting the facility (24% of the sample) were insensitive to the amount, which varied from $1,000 to $5,000 per year for 20 years as a credit on their income taxes. The authors conclude that the facility must meet some minimum standard of safety before people will consider trade-offs in terms of employment benefits or other compensation. Thus, looked at in this way, the public is willing to accept expert risk assessments to a degree, but only after a set of other conditions is met that have nothing to do with assessments of probabilities and consequences.

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Regardless of the way in which the public may view risks, it is often argued by the technical community that such risks should not be given weight in policy decisions because the informed public uses irrational rules of thumb in interpreting the low probabilities that experts assign to nuclear risks and that most of the public lacks adequate information to assess risks.

Concerning the first point, Kahneman and Tversky (1979) have abundantly documented a host of cognitive processes influencing lay risk assessment. These include framing, the overweighing of low probability events, etc. The question is whether the risk estimates that arise from these processes should then be adjusted to purge the effects of such cognitive processes. Economic theory is clear on this point. To the extent that perceptions affect behavior, perceptions are what matter.[12] One need only look at the huge advertising budgets of cigarette or automobile manufacturers to justify the powerful effect of perceptions about sexiness, power, or other amorphous product attributes on willingness to pay. No less is this the case with respect to nuclear power plants or hazardous waste sites, both of which have, according to the experts, negligible risks to the public but can have large effects on property values. Considering the time commitments implied by the public opposition to siting these facilities and the attempts to close existing facilities, WTP based on the perception of nuclear power's risks is large indeed.

On the second point, there is a lively debate in economics and other disciplines about the appropriate amount of information and understanding required of the public before their preferences are accepted as legitimate input into public policy decisions. The positions run the gamut from legitimizing the given state of lay knowledge irrespective of its depth or accuracy[13] to relying entirely on expert opinion in cases where the good being valued is unfamiliar or complex. As an intermediate position, there may be some benefit to weighing more heavily preferences of individuals already residing in areas hosting nuclear facilities, since "familiarity" with the hazard appears to be so salient in influencing perceptions.[14] One problem with this approach is the difficulty of separating risk assessments from the influence of the economic benefits (and losses) that such facilities may bring.

Considering the time commitments implied by the public opposition to siting these facilities and the attempts to close existing facilities, WTP based on the perception of nuclear power's risks is large indeed.

Based on the foregoing discussion, we view the disjuncture described above as a conflict between two sets of risk perceptions: those of the expert and

[12]One may seek to change preferences, but this lies outside of economic analysis and the welfare-theoretic paradigm. See Kopp and Sagoff (1993).

[13]A corollary to this position is that if it is believed the state of knowledge is too low, one institutes policies to raise it, but that such activities are separable from assessments of risks based on given preferences. With nuclear risks, however, there is some evidence that communication and information raise, rather than lower risks.

[14]Lindell and Earle have examined the preferences of this group relative to those of nuclear engineers, chemical engineers, urban residents, etc. and find that the distance they would be willing to reside from a nuclear power plant is intermediate between that of the nuclear engineers and urban residents.

those of the layman. In this light, the use of the objective/subjective dichotomy is pejorative, as is the hard/soft dichotomy. Therefore, we use the terms expert estimates of risks and lay perceptions of risks -- or expert risks and lay risks.

4.2 LITERATURE ON EXPERT ESTIMATES VS. LAY PERCEPTIONS OF RISK

This section briefly reviews the literature on the factors that explain the various elements of lay perceptions of risk. The purpose is to see what quantitative estimate of the differences between lay perceptions and expert estimates of risks can be made, and then to use them to calculate the costs of nuclear accidents, as an example, on an ex ante and ex post basis using the simulation framework of the previous section.[15]

A. Qualitative Representations of Risk

There is a substantial literature that identifies the main components that explain subjective risk (see Slovic 1987; Fischhoff 1989; and others). The findings are of considerable interest in understanding this concept but do not provide any direct advice on how expert estimates and lay perceptions of risks are related in quantitative terms. Hence we do not discuss them further in this paper.

Another strand of research in the qualitative literature examines the signaling effect of accidents. Kasperson's (1992) concept of the social amplification of risk applied to the TMI accident shows how the social costs of that accident were vastly larger than the direct costs, because of tighter regulation of nuclear plant operations, stricter construction standards, forced reliance on more expensive energy sources after the closure of some plants that did not measure up, and the canceling of plans to build new plants in the face of vastly increased public opposition to things nuclear. Building on Kasperson's observations, we suggest that another major accident at a nuclear power plant could very well result in the shutdown of all existing power plants or plants of similar design. In this case, a measure of the costs of a future nuclear accident is the probability of an accident times the incremental cost of substituting power from coal-fired power plants for that of the shutdown nuclear plants. At this stage, however, we have not been able to estimate this incremental cost, although we believe it would make a substantial difference to the overall external costs of the nuclear fuel cycles.

[15] This paper uses the terms "lay perceptions" and "expert estimates" in an operational sense, but there is no intrinsic superiority in one over the other.

A final strand of this literature attempts to explain either hypothesized or actual individual decisions with respect to some environmental hazard, using variables such as the probability of an accident, the degree of trust in the government, etc., to explain these decisions. Research like this helps to explain the extent to which the various attributes of risk affect decisions. See, for example, Kunreuther, Easterling, Desvousges, and Slovic (1990) who evaluated decisions on potential sites for a nuclear repository in Nevada. They find preferences between sites were influenced by the perceived estimates of the benefits and costs to individuals, and that perceptions of risks to future generations played a large part in determining the perceived benefits.

B. Quantification of Lay Risk

Numerous authors have remarked that there has as yet been no attempt to weigh and combine lay perceptions and expert estimates of risks in an actual risk assessment. The simulations by Smith (1992) (discussed below) represent a step in this direction. But, outside of simulation, this statement appears to be true. According to Slovic (1992), suggestions for performing this task have included: (i) giving more than proportional weight to preventing accidents with catastrophic consequences (Wilson 1975, Griesemeyer and Okrent 1981); (ii) adjusting risk estimates to account for risk perception characteristics (Rowe 1977; Litai, Lanning, and Rasmussen 1983); (iii) using multiattribute utility theory to combine "technical" estimates and lay perceptions of risks (Fischhoff, Watson, and Hope 1984); and (iv) using a cost-effectiveness paradigm to allocate more money to risk reduction for hazards that lack controllability, are poorly understood, and create involuntary risks (Bohnenblust and Schneider 1984).

While no risk assessments have incorporated these suggestions, there have been some quantitative analyses of lay risks in the literature, although they are not necessarily targeted to nuclear power plant risks. Indeed, there are several studies of nuclear repository risks and risks posed by landfills or hazardous waste sites. While we would prefer to rely on better targeted studies, analyses by Slovic and others touched on above, show that the public does not distinguish much between many of these types of hazards.[16] For example Slovic (1987) finds that the public views risks from a repository as larger than those from a nuclear power plant. And as we shall see below, the public appears to view hazardous waste sites and nuclear power plants as interchangeable in risks.

[16]Slovic, Layman, and Flynn (1991) discuss survey results before and after TMI which show that the public perceives the consequences of a nuclear reactor accident to be very similar to that of a nuclear war. Kunreuther, Desvousges and Slovic (1988) found that the perceived risks of the repository were at least as great as those from a nuclear reactor or a weapons test site.

In any event, it should be emphasized that ***all of the quantitative explorations in the rest of this paper are based on very strong assumptions to make up for the lack of a close fit between the available literature and the questions being asked here. Accordingly, they are merely suggestive of possible relationships.***

C. Use of Risk Ladders

A risk ladder is a graphical representation of risks (usually on a log scale) usually ranging from certainty down to de minimus risks, with the actual risks of a variety of hazards located on the ladder to help anchor lay risk perceptions about the hazard being examined in the study. If experts are asked to rank the hazard (or to estimate the risk based on a engineering analysis) and the lay public is asked to rank the hazard, then the ratio of the two risks (lay/expert) can be used to adjust expert risks through a simple multiplication. One study is particularly informative in this regard.

The paper by McClelland, Schulze, and Hurd (1990) is useful in bridging the gap between expert and perceived risks because it uses a risk ladder to quantify the relative risks people perceive. A mail survey was conducted of residents in a development located near a municipal waste landfill nominated for the Superfund Priorities List located in Los Angeles, whose only "health" problem appears to be related to release of methane gas which causes noxious odors. No elevated levels of any chemicals have been found. The 768 residents were asked about their risk perceptions both before and after closure of the site and their responses were explained by a variety of variables, with women, families with young children, distance from site (strongly correlated with odor perception), and age (younger) being strongly correlated with perceived riskiness.

The distribution of health risk perceptions from the usable responses (which may be from those most concerned) is bi-modal, expressed as deaths per 1,000,000 persons exposed. Before closure of the site, risks were clustered at 10^{-2} and from 10^{-4} to 10^{-6}, with an average risk of 0.022.

To calculate a ratio of lay perceptions of risk to expert risk estimates, we need an estimate of expert risk. We can assume that it is a "de minimus" risk of 10^{-6}, and calculate the average lay risk perception relative to this.[17] The lay risk estimate apparently ignores responses of risks=1 (a sizable percentage) as being from people who did not understand the concept. Thus, the ratio of lay perception of risks to expert estimate of risks would be $2.2*10^4$ ($=0.022/10^{-6}$).

[17]Assuming that expert risks would be 10^{-6} in this situation is obviously a strong and critical assumption.

This ratio would be used to multiply the expert-based estimate of expected deaths and injuries. Such a procedure would be upwardly biased because we are applying this percentage to a situation where death risks are larger than with a landfill. However, applying these results from a landfill context to a nuclear power plant context would underestimate risks to the extent that lay people perceive power plant risks as being far higher than risks from municipal landfills, *ceteris paribus*. On the other hand, to the extent that odor from the landfill causes elevated concern about health risks (a mechanism suggested by the literature), some of this bias may be reduced.

Strictly speaking, the survey results only apply to assessment of risks by the population within the affected area, which in the landfill case, was about 4,100 homes. Transferring such results to the nuclear power plant case is difficult because the affected area could be hundreds of miles or more.[18] One option might be to define the evacuation zone as the "affected area" and use the distance variable results to define lay risks beyond this point.[19] But, such a transfer of information from the landfill to the nuclear power case is obviously problematic.

D. Distance to Acceptance

A number of studies have asked respondents to provide the distance they would be willing to live from a nuclear or other type of facility (including those that may be considered "riskless," such as office buildings). One study even asks this of nuclear engineers and a variety of other groups. We use the relative distance provided by lay people versus nuclear engineers as a proxy for relative differences in risk perceptions.

Mitchell (1980) conducted such a survey of the willingness of people to accept a new facility, using the distance of the facility from their home as a measure of preference intensity. In 1980 (i.e., after the TMI accident), nearly 2,000 people nationwide were polled. The actual question was: "How close to your home could the facility be sited before you would want to move to another place or to actively protest, or wouldn't it matter to you one way or the other how close the facility was?" Respondents were told that each of the facilities would be operated according to government environmental and safety regulations.

[18]ORNL/RFF (1994b) calculate health effects out to 1,000 miles.

[19]This variable shows a very fast drop off of risk judgments with distance: -0.45/2 steps per block, or (with 26 steps in total on a log scale) about $2.2*10^{-3}$ deaths per block from the site, evaluated at the mean distance from the site (3.68 blocks).

Mitchell asks for responses to a coal facility and a nuclear facility, among other types of facilities (e.g., a 10-story office building, a large factory, and a disposal site for hazardous chemicals). Having responses to both a coal and nuclear power plant is particularly advantageous because it permits a very crude estimation of net risk differentials. However, as separate polling was not done to distinguish experts from lay people, we assume that all responses are from the lay public. The locational preference of experts is from the Lindell and Earle study (see below).

Because the decision problem is one of siting, which takes place as a result of "voting," we use the locational preferences of the median respondent. The results show that the median respondent would not locate closer than about 80 miles from the nuclear plant, and would not locate closer than 6-9 miles from the coal plant (say 7.5). It is also interesting to note that 29% of the respondents do not want the nuclear facility at any distance, and that about 13% feel this way about the coal-fired facility (Fig. 2).

Lindell and Earle (1983) take the same approach as Mitchell. They analyze responses from two mail surveys where specific groups (nuclear engineers, chemical engineers, environmentalists, urbanites, science writers, and people living near hazardous facilities) were asked about how close they would be willing to work or live from each of eight facilities, including a nuclear and a coal power plant. The first survey was conducted in 1978 involving 229 individuals nationwide. The second survey was conducted in mid-1980 with 396 individuals. They find, in the aggregate, distance gradients of perceived risks that are very similar to those of Mitchell. This survey is superior to Mitchell's for our purposes, however, in that distance gradients are estimated by group and we can, therefore, designate the gradient for the nuclear engineers as the "expert" gradient, using other groups for the lay gradient. Lindell and Earle find that 38% of the nuclear engineers are willing to live or work one mile from the nuclear facility, while only 7% of the urban residents feel this way. Comparing responses to the coal plant, 60% of urban residents would be willing to locate 10 miles from the facility while 84% of the *chemical engineers* would do so. An interesting finding is that only 47% of the *nuclear engineers* (i.e., *a smaller percentage than that for urban residents*) would be willing to locate 10 miles from the coal facility! Particularly interesting is the finding that nuclear engineers would locate close to a nuclear facility than chemical engineers, while chemical engineers would locate closer to a coal facility.

A problem in using the results of this research is the sparse reporting of results in the paper. The results for the nuclear facility are fully described, but not for the coal facility or other facilities, where only the percentage of respondents by group who would be willing to live 10 miles from each type of facility is given. As shown in Table 3, the median nuclear engineer would be

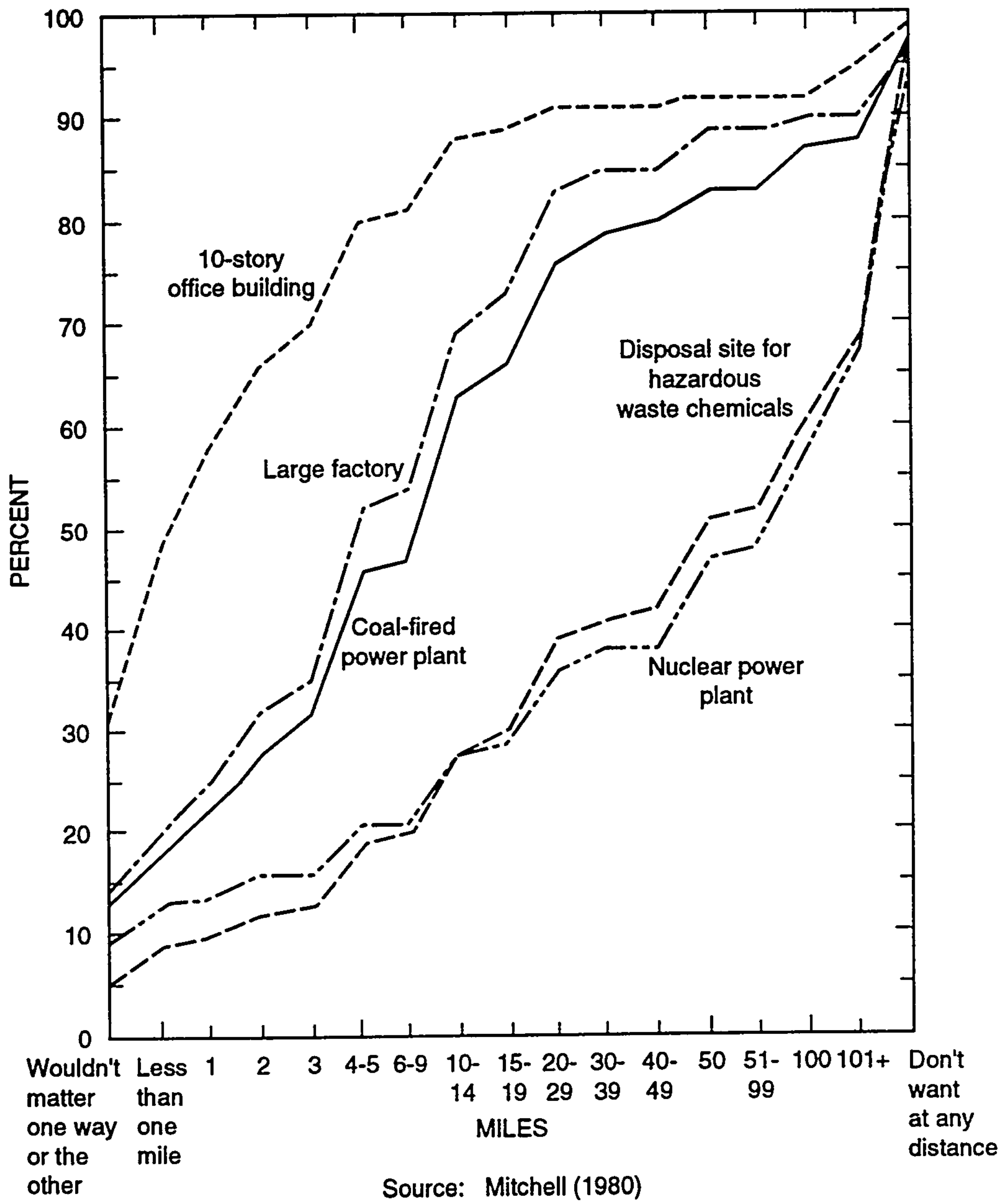

Fig. 2. Cumulative percentage of people willing to accept new industrial installations at various distances from their homes

willing to live 3 miles from the nuclear facility while the median respondent in all of the other groups would be willing to live between 30 and 100 miles from the facility. If the group living near a hazardous facility is designated as the "lay" group, the median distance would be far closer to 30 miles than if the urbanites were designated as the "lay" group.

For comparison to Mitchell's findings about acceptable living distances from a coal plant, 59% of the entire sample in Lindell and Earle's surveys was found to be willing to live within 10 miles, with the median respondent in nearly every specific group (except environmentalists and nuclear engineers) willing to live within 10 miles of the coal plant. This finding accords well with Mitchell's finding that the median individual would be willing to live 6-9 miles from the coal plant.[20]

To use the results of the above two papers to estimate the ratio of lay to expert risk preferences requires making some strong assumptions. First, we need to assume that the location decision reflects only health risk preferences; not potential employment benefits, visibility concerns, or any other factors. With this assumption, we then can make with more confidence a second assumption: that the distance response in the surveys reflects the point of perceived zero risk. This is still a strong assumption if individuals are expecting declining rent gradients for housing as one moves closer to the facility. If this is the case (and it may not be, as such an issue was not mentioned in either of the surveys), then individuals may be making implicit risk-money trade-offs rather than making a location decision solely on the basis of risk. Third, we need to assume that an individual who is willing to live closer to a plant than another individual perceives risks to be lower at every distance from the plant than the other individual. Then, for convenience, we can make the fourth assumption that perceived risks fall linearly with distance. Finally, we must make the strong assumption that risk preferences with respect to a nuclear waste site are similar to those for hazardous waste sites and nuclear power plants.

With these assumptions we have a situation as depicted in Fig. 3. The line labeled A represents the perceived risks over distance of the median nuclear engineer (the "expert"), who, in this hypothetical example, would be willing to live 10 miles from the plant (as a point of zero risk, by assumption). Maximum risk received by the engineer is represented by Ra. Lines B, B1, and B2

[20]Smith and Desvousges (1986) repeated the Mitchell analysis for a sample of 609 suburban Boston residents in 1984. They found the same ordinal preference ordering as Mitchell and Lindell and Earle, with the most risky facility being nuclear power, followed by hazardous landfills, and coal-fired power plants. But, the median respondent would live only 22 miles from the nuclear plant and 10 miles from the hazardous waste site, far closer than the others found. The authors offer that familiarity resulting from so many hazardous sites in the Boston area may be the cause.

Table 3.

Percentage of respondents willing to live or work within 10 miles of facility by group and facility

Facility	GROUP						
	Environmentalists	Urban Residents	Hazardous Facility Communities	Science Writers	Nuclear Engineers	Chemical Engineers	Total
Natural Gas Power Plant	73	67.5	79.1	83.7	85.7	100	78.3
Oil Power Plant	65.8	56.4	68	71.1	66.7	91.7	67.3
Coal Power Plant	46.5	55.9	68.2	57.9	47.6	83.5	59.1
Oil Refinery	48.3	45.9	54.9	49.9	40.5	77.8	51.7
LNG Storage Area	33.8	46.4	53.6	49.9	52.4	72.3	49
Nuclear Power Plant	15.2	21.5	35.4	31.6	88.1	61.1	35.3
Toxic Chemical Disposal Facility	24.4	24.1	27.8	39.4	23.8	55.5	29.5
Nuclear Waste Disposal Facility	14	14.7	18.6	32.4	59.6	47.3	25.1
Facility Mean	40.1	41.6	50.7	52	58.1	73.7	49.4

Source: Lindell and Earle (1983)

represent the perceived risks over distance of the median urbanite (or other representative of a "lay" person), who is assumed to be willing to live 20 miles from the plant. Maximum lay risk is represented by Rb. With the additional assumption that the median individual in both groups has the same perception about the rate at which perceived risks fall with distance, then line B and line A are parallel.

Now, as a thought experiment, assume that the median expert and the median lay person are given the task of assessing the risks to the population. The two individuals use their own assessment of risks to themselves over distance, adjusting this risk for the number of people living at various distances from the prospective power plant. In effect, this involves taking the area under their risk-distance curves but weighing each risk at a given distance by the population living at that distance. Assume, for simplicity, that population is homogeneously distributed around the plant.[21] Then, in the example above, where the expert would be willing to live no less than 10 miles from the plant and the lay person would live at least 20 miles from the plant, simple geometry shows that the total perceived lay risks would be four times that of the expert. If, as we observe in the Lindell and Earle study, the median nuclear engineer would be willing to live 3 miles from the plant while the median lay person would live 60 miles from the plant (picking a round number between 30 and 100 miles), then total lay risks would be 400 times that of the experts.

The assumption that the lay risk line is linear and parallel to the expert risk line is a strong assumption, made for convenience only. Lines such as B1 or B2 are also plausible. However, a line like B1, with a slope less than that of A, seems more plausible than B2. The assumption that Rb>Ra seems unobjectionable, with no prior knowledge on the magnitude of this difference. However, it is more plausible that an average lay person would perceive risks falling less rapidly with distance relative to the experts, than the reverse. In this case, the above estimate of a 400 times difference would be an overestimate of the difference between lay perceptions and expert estimates of risks.

4.3 SIMULATION OF EXPERT VERSUS LAY RISKS

In this section, we add the distinction between expert estimates and lay perceptions of risks to the state-dependent utility model developed above. For modeling lay risks, we take the position of Smith (1992), who posits that "risk

[21]If population were distributed heterogeneously around the prospective plant site, the ratio of the risks for the two groups might not be affected significantly, although the total population risks could be quite different.

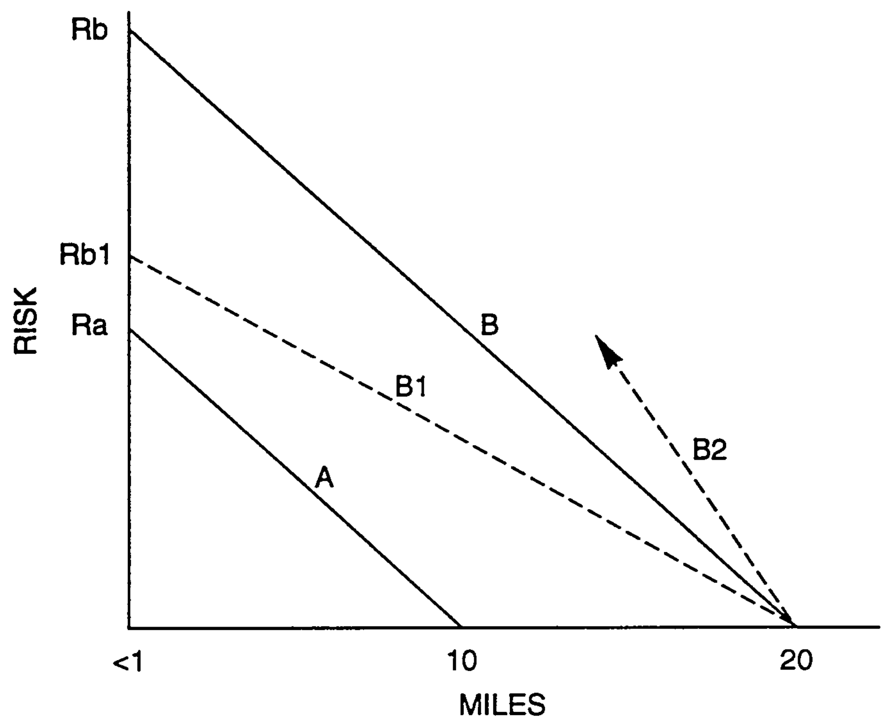

Fig. 3. Risk-distance gradients

perceptions reflect objective information about the events at risk and each individual's beliefs about the process" (p. 43). Translated into the terminology used here, this statement implies that lay risk perceptions are formed in part from what the experts say and in part from one's own preferences, experiences, as well as knowledge acquired elsewhere.

In Smith's formulation, which is based on Viscusi's (1989) prospective reference theory approach, the probability of an accident (q) is replaced by a perceived probability function

$$(q) = (\mu q_e + ßq_r)/(\mu + ß) \quad (3)$$

where q_r is the individual's initial risk assessment and q_e is the expert's risk assessment which updates the individual's assessment. The parameters μ and ß are weights describing the informational content associated with expert risk assessments versus each person's initial risk level. So, if an individual didn't believe the experts, μ would have a low value, and if an individual totally believed the experts, ß would take a zero value. This approach has the advantage of decomposing the expression into two expected utility functions, corresponding to the individual's *a priori* risk assessment and the expert's risk assessment.

Now, as was done in the previous section, Smith's formulation is reworked for the parameter values and functions relevant to our problem. This calculation involves estimating the ratio of damages using the lay-expert perspective developed above compared to the expert perspective. These ratios are computed separately from an ex post perspective and from an ex ante perspective.

The scenarios are specified as they were in the first set of simulations (see Table 1), with some new probability parameters added. They cover the same two indirect utility functions examined in the ex ante/ex post simulations above, with most of the same parameters, reflecting the same degree of risk aversion, and the same median income. Alternative parameter values are given for the expert estimate of risk, the initial lay perception of risk, and the weight assigned to the expert estimate of risk.

The results of the simulations for various parameter values and indirect utility functions are provided in Table 4, which has two parts, one part for each utility function. As for Table 2, this table also contains some illustrative estimates of parameters based on the literature review (in bold and shaded). The table provides the ratio of damages when accounting for lay perceptions of risks (which are influenced by expert estimates of risks) to damages when accounting only for expert estimates of risks. This ratio is computed from both an ex post perspective and an ex ante perspective. Thus, referring to the upper leftmost portion in the first part of Table 4 when experts' probabilities of the accident occurring are 0.1, the initial lay probability is 0.5, and the weight assigned to the experts' estimate is 0.5, the ex post (XP) damage using all of this information is 3.00 times the damage using only the information from the expert. Similarly, for these same parameter values, but calculating ex ante estimates, the damages with lay perceptions admitted are 2.98 times those with expert estimates admitted, for the case of a loss of 10% of income and risk aversion of 0.5.

Referring only to the unshaded sections of the table, the following general results can be seen:

(i) As expected, the lay-based to expert-based damage ratio is proportional to the ratio of the weighted average of lay to experts' probabilities for the XP case. In the case noted above, the weighted average lay person's probability estimate is 3 times that of the expert.[22]

(ii) More generally from the ex post perspective, the greater the divergence between the lay people's and experts' probabilities (e.g., because the initial lay probability is higher, or the weight given to the experts' probability is lower), the greater the ratio.

(iii) From an ex ante perspective, the above conclusions also hold;, i.e., in general, the ratio of lay-based to expert-based damages is very sensitive to the divergence between expert and lay probabilities, although not strictly as a weighted average of the expert and initial lay probabilities (as was the case for the ex post perspective).

(iv) Beyond this, the conclusions are much more clouded from the ex ante perspective. The lay-based to expert-based damage ratio may or may not be sensitive to the size of the loss and the degree of risk aversion assumed, depending on the particular utility function chosen. For function I, the ratio is insensitive to the size of the loss. For example, scanning the first row, we find the ratio varying from 2.81 to 2.9 over very large differences in losses and risk aversion. Yet for function II, variation is far larger, particularly for combinations of high risk aversion and high losses.

[22]((0.1*0.5) + 0.5*0.5))/0.1 = 0.3/0.1 = 3.

Table 4. Expert vs. lay damages form ex post (XP) and ex ante (XA) perspectives

Utility Function I $V = (M(1-W))^b$

Expert Probability	Initial Lay Probability	Lay Weight for Expert	Ratio of Lay to Expert XP	Ratio of Lay to Expert XA (1)			
				r=0.5	r=0.5	r=0.1	r=0.5
				W=10%	W=50%	W=10%	W=14%
0.10	0.000	1.00	1.00	1.00	1.00	1.00	1.00
0.10	0.500	0.50	3.00	2.98	2.91	2.97	2.81
0.10	0.500	0.01	4.96	4.91	4.67	4.85	4.33
0.001	0.000	1.00	1.00	1.00	1.00	1.00	1.00
0.001	0.100	0.50	50.50	50.44	50.13	50.38	49.72
0.001	0.100	0.25	75.25	75.11	74.43	74.97	73.50
0.001	0.100	0.05	95.05	94.82	93.74	94.60	92.25
0.001	0.100	0.01	99.01	98.76	97.59	98.53	95.97
6.20E-05	**0.000**	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**
6.20E-05	**0.012**	**0.00**	**193.55**	**193.49**	**193.21**	**193.44**	**192.83**
6.20E-05	**0.025**	**0.00**	**403.23**	**402.97**	**401.75**	**402.73**	**400.08**
6.20E-05	**6.20E-02**	**0.00**	**1000.00**	**998.41**	**990.93**	**996.94**	**980.62**

W is potential loss of income if an accident occurs expressed as a percent of total income.

For Utility Function II the ratio is dependent on the level of income as is the relative risk aversion, which is given as r. For cases (i) b=0.0005. For case (ii) b=0.00009.

Median Income is $24,807

Table 4. (cont'd)

Utility Function II V = -EXP[-BM(1-W)]

Expert Probability	Initial Lay Probability	Lay Weight for Expert	Ratio of Lay to Expert XP	Ratio of Lay to Expert XA (1)			
				r=11.2 (i)	r=6.2 (i)	r=2.0 (ii)	r=10.7 (i)
				W=10%	W=50%	W=10%	W=14%
0.10	0.000	1.00	1.00	1.00	1.00	1.00	1.00
0.10	0.500	0.50	3.00	2.51	1.28	2.93	2.29
0.10	0.500	0.01	4.96	3.63	1.40	4.73	3.13
0.001	0.000	1.00	1.00	1.00	1.00	1.00	1.00
0.001	0.100	0.50	50.50	47.66	8.12	50.19	45.44
0.001	0.100	0.25	75.25	69.13	9.09	74.56	64.62
0.001	0.100	0.05	95.05	85.53	9.66	93.95	78.82
0.001	0.100	0.01	99.01	88.73	9.76	97.82	81.55
6.20E-05	**0.000**	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**
6.20E-05	**0.012**	**0.00**	**193.55**	**190.76**	**64.26**	**193.26**	**188.34**
6.20E-05	**0.025**	**0.00**	**403.23**	**391.36**	**86.05**	**401.97**	**381.40**
6.20E-05	**6.20E-02**	**0.00**	**1000.00**	**930.86**	**114.72**	**992.33**	**878.20**

W is potential loss of income if an accident occurs expressed as a percent of total income.

For Utility Function II the ratio is dependent on the level of income as is the relative risk aversion, which is given as r. For cases (i) b=0.0005. For case (ii) b=0.00009.

Median Income is $24,807

We are now in a position to offer some illustrative estimates, with parameter values based on the analysis in the U.S. Nuclear Fuel Cycle Report, of the differences introduced by taking an ex post rather than an ex ante perspective and an expert-based rather than a lay-based risk perspective.[23] The key parameter values are: an average loss of 14% of income, an expert-assessed probability of the event occurring of 6.2×10^{-5}, and a lay assessment of this probability that is 400 times that of the expert.[24] For sensitivity analysis, we also use lay/expert probability ratios of 1,000 and 200. As discussed earlier, the choice of the factor of 400 is illustrative of the ratio of lay to expert *risks* associated with several studies discussed above (Mitchell 1980; Lindell and Earle 1983; see also Smith and Desvousges 1986).

The results are shown in the shaded portions of Table 4 for the lay versus expert risk comparisons and Table 5 for the combined comparison of damage estimates computed with respect to ex ante/ex post perspectives and lay/expert risks. The key results are in bold.

Referring to Table 4, we note first that the lay-based to expert-based damage ratios computed from an ex post perspective (in column four for each utility function) equal the assumed ratios of lay perceptions to expert estimates of risks. This result follows because the probabilities of an accident are not embedded in the utility function but simply multiple terms derived from it to estimate willingness to pay. When the expert probability is set at 6.25×10^{-05}, the corresponding ratio of lay to expert risks probabilities are 0, 192, 403, and 1,000 respectively. Second, with losses of 14% of income, the results from an ex ante perspective (in bold) are quite close to those from an ex post perspective, independent of utility function and risk aversion. Thus, a rule of thumb for scaling up expert ex post damage estimates to account for lay perceptions of risks approximates an approach of scaling up an expert-based ex ante damage estimate to account for lay perceptions of risks. (See the discussion of Table 5 below for a direct numerical comparison of ex ante and ex post estimates to make this point clearer.)

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[23]Note that this analysis is in "utils," thus, without specifying the marginal utility of money, we cannot convert these estimates to a monetary value, other than by assuming that the loss is the ex post estimate and noting that with risk aversion, this loss will be underestimated.

[24]For ease of display, we provide the expert probability, an initial lay probability that is 400 times that of the expert, and assign the expert's probability a zero weight, making the initial and the final lay perception of the probability identical.

Table 5. Ratios of ex ante to ex post damages based on adjusted probabilities

Utility Function I V = (M(1-W))^b

Expert Probability	Initial Lay Probability	Lay Weight for Expert	Ratio of Ex Ante to Ex Post Damages					
			r=0.5	r=0.5	r=0.5	r=0.5	r=0.1	r=0.1
			W=0.1%	W=10%	W=50%	W=14%	W=50%	W=14%
0.10	0.000	1.00	1.0023	1.02370	1.15442	1.03392	1.02867	1.00668
0.10	0.500	0.50	1.00018	1.01843	1.12010	1.02638	1.02303	1.00523
0.10	0.500	0.01	1.00013	1.01327	108647	1.01899	1.01716	1.00380
0.001	0.000	1.00	1.00025	1.02631	1.17140	1.03765	1.03134	1.00738
0.001	0.100	0.50	1.00024	1.02500	1.16291	1.03578	1.03001	1.00703
6.20E-05	**0.000**	**1.00**	**1.00025**	**1.02633**	**1.17156**	**1.03768**	**1.03136**	**1.00739**
6.20E-05	**0.012**	**0.00**	**1.00025**	**1.02602**	**1.16951**	**1.03724**	**1.03104**	**1.00730**
6.20E-05	**0.025**	**0.00**	**1.00024**	**1.02568**	**1.16728**	**1.03675**	**1.03070**	**1.00721**
6.20E-05	**6.20E-02**	**0.00**	**1.00023**	**1.02470**	**1.16094**	**1.03535**	**1.02970**	**1.00695**

W is potential loss of income if an accident occurs expressed as a percent of total income.

For Utility Function II the ratio is dependent on the level of income as is the relative risk aversion, which is given as r. For cases (i) b=0.0005. For cases (ii) b=0.00009.

Median Income is $24,807

Table 5. (cont'd)

Utility Function II   V = -EXP[-bM(1-W)]

Expert Probability	Initial Lay Probability	Lay Weight for Expert	Ratio of Lay to Expert XA (1)					
			r=12.4 (i)	r=11.2 (i)	r=6.2 (i)	r=10.7 (i)	r=1.1 (ii)	r=1.9 (ii)
			W=0.1%	W=10%	W=50%	W=14%	W=50%	W=14%
0.10	0.000	1.00	1.00560	1.77114	6.31634	2.20976	1.67315	1.15290
0.10	0.500	0.50	1.00435	1.48395	2.68875	1.68310	1.43329	1.11369
0.10	0.500	0.01	1.00313	1.29527	1.78885	1.39320	1.26856	1.07851
0.001	0.000	1.00	1.00622	1.97832	64.58301	2.68730	1.83772	1.17371
0.001	0.100	0.50	1.00591	1.86718	10.38784	2.41808	1.75034	1.16318
0.001	0.100	0.01	1.00561	1.77294	6.36361	2.21348	1.67461	1.15310
6.20E-05	**0.000**	**1.00**	**1.00623**	**1.98060**	**78.24230**	**2.69319**	**1.83949**	**1.17391**
6.20E-05	**0.012**	**0.00**	**1.00615**	**1.95211**	**25.97606**	**2.62070**	**1.81731**	**1.17135**
6.20E-05	**0.025**	**0.00**	**1.00607**	**1.92230**	**16.69796**	**2.54739**	**1.79394**	**1.16857**
6.20E-05	**6.20E-02**	**0.00**	**1.00584**	**1.84365**	**8.97592**	**2.36517**	**1.73157**	**1.16077**

W is potential loss of income if an accident occurs expressed as a percent of total income.

For Utility Function II the ratio is dependent on the level of income as is the relative risk aversion, which is given as r. For cases (i) b=0.0005. For cases (ii) b=0.00009.

Median Income is $24,807

Referring to the shaded portions of Table 4 for sensitivity analysis, we note that for combinations of small losses and low risk aversion, the lay-based damage estimates from an ex ante perspective are fairly close to those from an ex post perspective, irrespective of utility function assumed. This result does not hold, however, when risk aversion and losses are large, except for Function I. For Function II, the ratio of lay-based to expert-based ex ante damage estimates is actually smaller for the large risk aversion/large loss cases.

5. SIMULATIONS OF EU VS EED PERSPECTIVES

The final set of simulations are performed on a model that combines both the ex ante/ex post perspective with the expert-lay perspective. Table 5 presents the simulation results. Referring again to the unshaded portions of the table, we can make the following generalizations:

(i) Differences in expert and lay probability assessments have very little effect on the ratio of ex ante to ex post damages because these differences appear in both the numerator and the denominator of the expressions underlying this ratio, i.e., this divergence affects damage estimates from both perspectives;

(ii) The greater the lay perceptions of risks over the expert estimates of risks the *smaller* the ex ante/ex post damage ratio;

(iii) The variance of the damage ratio caused by differences in lay-based and expert-based probabilities rises as the loss and risk aversion increase. Referring to function II in Table 5, we observe that with a 0.1% income loss the ratio is nearly identical for each lay-expert risk combination, while this ratio differs greatly for the same combinations when the loss is 50% of income;

(iv) When the losses are large, the risk aversion large, and the probabilities small (what might be regarded as the classic set of conditions for public perception of nuclear power risks), the ratio of ex ante to ex post damage is very sensitive to the lay-expert probability differential. This result is seen, for instance, in Table 5, Function II. Giving the expert-based probability of 0.001 a 50% weight and the initial lay-based probability of 0.01 a 50% weight gives an ex ante to ex post damage ratio of 10.4 when losses are 50% of income, but this ratio falls to 6.4 if the expert weight is 0.01 and rises to 64 if the expert is given all the weight.

When the losses are large, the risk aversion large, and the probabilities small ... the ratio of ex ante to ex post damage is very sensitive to the lay-expert probability differential.

Turning to the bold-set portions of the table, if utility functions were actually of the form of function I, we could generally ignore all of the issues discussed in this paper, as the ratio of ex ante to ex post damage is very close to 1 for almost all parameter values assumed in the sensitivity cases. The exception is the case where the potential loss of income reaches 50%. For other utility functions, the ratios of ex ante to ex post damages appear to be more influenced by risk aversion assumptions than by the form of the utility function, as the ratios are also around 2 for relatively low risk aversion (2-3) but increase significantly for risk aversion over 10. However, the levels of probabilities and the difference between lay-based and expert-based probabilities are not particularly significant factors in affecting the ex ante to ex post damage ratio.

6. CONCLUSIONS

Current practice in estimating the damages from low probability-high consequence events appears to ignore significant features of the problem. Individual aversion to risk, the ex ante perspective in decision making under uncertainty, and lay risk perception all appear to affect individuals' willingness to pay to avoid the risks of those types of accidents. These factors accord with observed behavior and with the theory that underlies it, but are not part of conventional estimates of damages.

The difference between lay and expert risk perception is a particularly important element conceptually. While no one wishes to see public policy decisions made on the basis of an uninformed, irrational public, the idea is advanced here that the public can make rational and replicable assessments about risks and that the concept of risk held by the public is *broader* than that of the experts. Neither concept is right nor wrong, but to ignore such differences in risk perceptions means ignoring the real economic consequences of such differences, whether in depressed property values near sites with undesirable activities or in the inability of utilities to build and operate certain types of power plants.

Beyond explaining how these conceptual elements can be addressed, this paper has sought to give some empirical content to these issues through a series of simulations. These simulations use several different portrayals of individual preferences (i.e., two different state-dependent utility functions). In this effort, we have built on and extended work by Freeman (1989) and Smith (1992). To make the simulations as "realistic" as possible, we have drawn on the scant literature available for parameter values and, in particular, have used baseline data and results from a conventional damage assessment of nuclear power plant accidents performed as part of the U.S.-EC Fuel Cycle Study.

The results are in the form of the ratio of the damages estimated by taking the above elements into account (what we term the expected utility approach, EU) to the damages estimated without these elements (the expert expected damage approach, EED). We find, for a model that takes expert estimates of risks as given, that this ratio is greater the greater the risk aversion, the smaller the probability of the event, and the greater the loss if the event occurs. This is precisely the case one might expect to be applicable to estimating damages from a nuclear power plant accident. Nevertheless, with the parameter values taken from the traditional (EED) damage estimate, our results are more equivocable. We find that the utility function chosen makes a big difference in the estimates of damages and that for one of the functions examined, the difference is trivial. In addition, we find that the results are not very sensitive to the estimate of the probability of the event.

The scant literature on differences between expert and lay risk perceptions provides a poor basis for judging how wide this gap is. More research in this area that seeks to quantify these differences rather than simply describe them is desperately needed. Using an illustrative ratio from this literature of lay to expert risks, and a model that bases lay risk perceptions on expert risk estimates and a weighing factor for how much the expert is believed, we find, not surprisingly, that damages under the EED approach are proportionally higher when lay risks are used rather than expert risks. We also find, however, that this difference is *not* proportional when the EU approach is used and, in fact, that the EU approach gives somewhat smaller estimates of damage than the EED approach (using lay risks). How different the estimates are depends very much on the utility function assumed.

In the simulations putting all the elements together, we explore results in terms of the ratio of damages using the EU approach with lay risks versus expert risks to that using the EED approach with lay versus expert risks. Using lay risks instead of expert risks makes a big difference in the level of damage, but not to the ratio, because either lay risks or expert risks appear in *both* the numerator and the denominator. For parameters from the U.S.-DOE Nuclear Fuel Cycle Study, the other issues discussed in this paper could be ignored if Utility Function I appropriately describe preferences. But, this function is probably less consistent with the evidence on individual behavior than Utility

Function II. Results using the latter function are quite sensitive to the degree of risk aversion assumed, less so to the probability and magnitude of the event.

There is a compelling conceptual and empirical case for specifying a model for estimating damage from low probability-high consequence accidents that accords with economic theory. There are big differences between experts' and lay people's estimates of the externalities associated with the risks of those accidents. Such a model provides a bridge to understand those differences better. While the empirical issues are clouded by a lack of information on the appropriate utility function and degree of risk aversion, our simulations indicate the likely differences between estimates based on the EED probabilistic risk assessment approach and those based on the EU "risk valuation" approach.[25] We would argue that, until the presently sparse literature on the risk valuation method is significantly expanded, particularly in the direction of assessing the willingness to pay to reduce or eliminate power plant accident risks, little progress can be made in estimating such damages using that approach directly. However, this paper shows that the ratio of expert-based estimates of damages to lay-based estimates can be approximated (for the ex post valuation case) by the ratio of expert to lay perceptions of risks.

There is no easy reconciliation of the differences between these perceptions. It can only happen from communication of information and views, which is a two-way process in which both parties are educating, and being educated, by the other.

There is no easy reconciliation of the differences between these perceptions. It can only happen from communication of information and views, which is a two-way process in which *both* parties are educating, and being educated, by the other. Two important messages of this paper are that lay people's perceptions of risk are as equally "valid" as expert estimates of risk, and that the former should be given greater emphasis than they currently receive in estimating the expected damages from low probability — high consequence events.

[25] The EED approach is based on a number of defensible technical methods that have been developed over a number of years (they are not discussed in this paper).

Finally, we note that while the context for this paper is a study of externalities, the analysis in the paper is directed at estimates of *damages.*[26] These damages may not be externalities. Whether they are depends on the policy and regulatory setting.

[26] Damages are as defined on p. 18-1.

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PAPER 19[1]

THE USE OF EXTERNALITY ESTIMATES IN THE CALCULATION OF ADDERS BY STATE PUC REGULATORS

1. INTRODUCTION

The primary focus of the U.S.-EC study is the development and illustration of methodologies for the estimation of marginal damages and associated externalities that result from the addition of electricity generating capacity in a specific reference environment. This paper describes how this information can be used to guide resource planning by electric utilities and State public utility commissions (PUCs). First, we discuss the "second-best" policy environment in which PUCs must operate. We then discuss the use of "adders" which are a policy tool that many PUCs are currently considering. Then, we introduce and estimate a formal model to calibrate these adders, based on estimates of externalities, in order to promote economic efficiency in resource planning and investment decisions.

1.1 THE "SECOND-BEST" POLICY ENVIRONMENT FOR PUBLIC UTILITY REGULATION

Economic efficiency (Pareto optimality) requires that the levels of production and consumption of all things that matter directly (or indirectly through their effects on firms and profits) be set so that the marginal social value of the thing is equal to its marginal social cost, where social value (cost) is defined as the sum of private and external value (cost). In a "first-best" world, a PUC would cooperate with the relevant local, State and Federal environmental agencies and legislators to achieve a Pareto optimal allocation of goods and services in society, preferably through market-based mechanisms. In principle, this could be achieved with policies designed to internalize external costs throughout society by replacing our patchwork of command and control policies with economic incentive

[1]Based largely on a working paper by D. Burtraw, K. Palmer, and A. Krupnick, 1993. "Electric Utility Planning in the Presence of Externalities," Resources for the Future.

policies that inherently force internalization. However, this comprehensive turn of events is unlikely to happen in the near future, and in any case it is beyond the scope of this investigation.

This paper describes how ... PUCs ... can seek cost-effective choices of electricity investment and generation, given the policies and regulations in place, ...

We proceed on the assumption that making optimal adjustments to the wide array of existing environmental policies is beyond the power of the PUC. However, the PUC does have the opportunity to evaluate alternative supply options taking existing pollution control policy as given. This evaluation can seek cost-effective choices of electricity investment and generation, *given* the policies and regulations in place, where cost-effectiveness here means choosing technologies with the lowest social cost to meet new electricity demand. This is the problem of "second-best" policy making.

Once one makes a distinction between optimal regulation and enters the second-best world of social costing, much of the confusion in the social costing debate dissolves. For instance, economists who feel that social costing is a bad idea justify this position in a way that mischaracterizes the objectives of social costing as a first-best policy. For instance, Joskow says, "their (PUCs) reliance on numerical "adders" to reflect environment impacts will not achieve the goal of improving the environment at the lowest reasonable cost to society" (Joskow 1992 p.1) [emphasis added]. And "States such as Massachusetts, Nevada, New York, and California have decided they can 'fix' what ails existing environmental regulations by appending various externality "adders" to the private costs of new utility resource options" (Joskow 1992 p.12) [emphasis added].

We believe that in many instances PUCs will not have the authority or the expertise required to set environmental policies and standards. And for those externalities regulated by environmental agencies, PUCs have no business trying to second guess or correct the perceived errors of these other agencies. However, internalizing externalities in electricity system planning is consistent with a much more limited objective, an objective which is entirely appropriate to PUCs. This objective is to minimize the increment to social cost associated with meeting a given increase in the demand for electricity.

We wish to emphasize three points about this objective:

(i) In pursuing this objective, PUCs must take existing environmental policy as given. Therefore, pursuit of this objective does not involve second guessing environmental agencies or attempts to "fix" existing environmental policy. But it does require a concern for the environmental impacts of the electricity supply and demand side management (DSM) choices made by the PUC.

(ii) This objective can be interpreted as a straightforward extension of the traditional rationale for PUC regulation of electricity prices, namely concern for the impact of prices on consumer welfare. A low price for electricity is not a fundamental goal of policy; rather it is one of several instruments to be used to promote the fundamental objective of improving consumer welfare, PUCs should be aware of the effects of their decisions on environmental quality. Choosing the electricity supply option with the lowest private cost helps to keep the price of electricity low. But a PUC does consumers no favor by choosing the alternative with the lowest private cost if this choice imposes a hidden "tax" on consumers through its environmental damages.

(iii) The PUC that wants to promote consumer welfare must take account of both the private costs and the external costs of alternatives in its decision making. This will often require some form of "adder".

However one feels about the objectives of social costing, there is universal agreement about the existence of a "piecemeal problem" - i.e. the possible losses of efficiency associated with singling out the electricity sector for social costing. (The likely application of this approach in some, but not all, States may also be a kind of piecemeal problem, to be dealt with in the next section).

These efficiency losses could arise because of distortions introduced in relative prices. Limiting use of social cost adders to the utility industry puts the utility's electricity at a competitive disadvantage relative to self-generation or purchasing power directly from an independent power producer (IPP). Some IPPs may find it advantageous to deal directly with customers rather than go through the PUC review process of bids to connect to the grid. This is the so-called "by-pass" problem. If these potential sources have uninternalized environmental costs associated with them, social welfare could be reduced by requiring externality adders only in the utility system. Electricity, in general, may be inappropriately disadvantaged next to other forms of energy.

The two problems of incorrect electricity price signals and by-pass arise not because social costing is wrong, but rather because it is incomplete in its coverage. PUCs do not presently have jurisdiction over electricity sources that do not connect to the grid. They may not have the authority to set the price of electricity above its private cost in an effort to "pass through" uninternalized environmental costs to electricity consumers. The solution to these problems is to develop a comprehensive system for internalizing environmental costs not only on utilities but on other sources of these emissions as well.

1.2 THE USE OF SOCIAL COST ADDERS BY STATE PUCS

One policy tool has gained the widest interest at the State level. It is the use of adders in the resource planning context to account for externalities in investment decisions. The merits of this approach have been debated widely. The purpose of this discussion is not to evaluate these merits further. Rather, assuming that a PUC is interested in this approach, this discussion provides guidance about how to calculate the "optimal adder," given information about externalities of the type developed in the U.S.-EC study.

The primary consideration in the calibration of adders is to take into account the important institutional features of the electricity industry and the particular characteristics and limitations of adders as a policy instrument. Economic regulation in both the investor-owned and public power segments of the electricity industry is characterized by an obligation to serve customer demand at a regulated price. The electricity industry is forced to compete in the provision of energy services with regulated and unregulated alternatives. Unfortunately, policies that are available to State utility regulators, to attempt to correct for inefficiencies associated with the external social costs of energy use, can be applied only on a piecemeal basis to just the regulated segments of the energy industry.

Adders are somewhat similar to taxes, but they are not actually charged and no revenue is exchanged.

Adders are somewhat similar to taxes, but they are not actually charged and no revenue is exchanged. Instead they serve as place-holders intended to influence the choice of technology to meet an increment in demand, and to promote decisions on the basis of least *social* cost rather than least private cost, which typically reflects only a utility's financial costs. However, if these adders have any effect on the outcome of the resource planning process, they will indirectly affect private costs and consequently price, due to the reordering of resource options.[2] This price effect may cause some customers to consume less electricity and to consume more of an alternative. Hence, the application of an adder as a policy instrument to correct for social costs could have unintended economic and environmental consequences that policy makers should try to anticipate.

[2]The use of adders in *dispatch* of existing resources has yet to appear for serious consideration on PUC agendas, but it may be on the horizon.

One manifestation of second-best policy making is the so-called "piecemeal" problem. State utility regulators can direct policy only toward regulated segments of the energy industry. The increase in prices in these segments of the industry that may result indirectly from social costing (noted above) will have a direct effect on individual choices about energy consumption. For instance, a residential customer may decide to avoid the use of regulated energy sources at the margin and heat his or her home with wood, which would have adverse environmental effects of its own that are not reflected in the estimates of externality associated with a new source of electricity. Alternatively, a large industrial firm may decide to reallocate production away from a facility that falls under the domain of social costing of electricity and toward an area with less expensive electricity prices. In order to promote economic efficiency, policy makers should be made aware of these inadvertent responses resulting from the piecemeal application of social costing just to regulated firms.

Another manifestation of second-best policy making is the fact that regulated electricity prices do not typically reflect marginal costs. Although the deviation from marginal cost pricing is itself a policy response to natural monopoly characteristics of the electricity industry, the fact that prices do not reflect marginal cost is a source of economic inefficiency. The application of adders may exacerbate this inefficiency, or alternatively be corrective, depending on the specific circumstances in an individual service territory.

Furthermore, if there exist idle resources in a utility's service territory, the potential response by some firms to reduce activity or potentially to leave the service territory due to the indirect effect on prices could have a negative effect on macroeconomic activity and economic welfare.

For these reasons, the optimal adder may not be equivalent to the measure of externality as it has been developed in the U.S.-EC study. The social costing of electricity fits soundly in the realm of what economists term "second-best" policy. This means that the use of adders as a policy instrument must co-exist with other aspects of the regulatory problem that impose their own form of economic inefficiency, or with other policies that attempt to correct other forms of inefficiency.

... this paper outlines a formal model of social costing that takes into account ... the bypass problem and the regulated pricing of electricity.

The following section of this paper outlines a formal model of social costing that takes into account the first two issues mentioned above—the bypass problem and the regulated pricing of electricity. This model is intended to help policy makers take into account unintended consequences in these other dimensions in the implementation of social costing policies. The conclusion is a formula for setting the optimal adder, based on estimates of externalities, in order to restore economic efficiency in resource planning.

2. THE ECONOMIC MODEL

Burtraw et al. (1993) analyze social costing within a simple but fairly general formal model under the assumptions that the goal of the PUC is to maximize social welfare, and that environmental policy (as established by statute and environmental agencies) is not subject to consideration.[3] The utility is assumed to comply with all relevant regulations and attempts to minimize its costs in fulfilling its obligation to serve demand at a regulated price. The utility regulator (PUC) is assumed to set price equal to average cost (including a reasonable return on investment) in order to ensure both the solvency of the firm and that no excess profits are earned. This assumption can be altered to reflect the particular pricing policies in any individual service territory. The model considers multiple technology options for the regulated utility, and an option for utility customers to bypass the grid and substitute alternative sources of energy, which may have their own pollution not subject to social costing.[4] In addition, the model considers the welfare implications of deviations from marginal cost pricing in a regulated industry, and the influence that social costing may have on this issue.

... the optimal adder for a technology equals its externality, adjusted by a multiplicative factor.

The result of this theoretical exercise is represented in the following formula, which relates the optimal adder (a^x) for a given technology (x) on the left hand side of the equation to the externalities associated with a kilowatt-hour of output generated with technology (D'_x), which is the second term on the right hand side of the equation. According to the formula, the optimal

[3]Burtraw, Dallas, Winston Harrington, A. Myrick Freeman III, and Alan J. Krupnick (1993). "Some Simple Analytics of Social Costing in a Regulated Industry," Discussion Paper QE93-13, Washington DC: Resources for the Future.

[4]In the natural gas industry, the term "bypass" has a particular connotation regarding purchase by large industrial customers from interstate pipeline companies, effectively bypassing the local distribution system and State level regulation. We use the term more loosely, to describe reductions in demand that might result from fuel switching or possibly the shift of industrial production to facilities outside the utility's service territory.

adder for a technology equals its externality, adjusted by a multiplicative factor. As the notation suggests, we simplify the exposition by assuming that the proper measure of externality is marginal damage. If this is not the case, then marginal damage estimates should be adjusted in order to arrive at an estimate of externality. The cumbersome first term on the right hand side simplifiels to a simple coefficient, or number, that modifies the marginal damage estimate. This coefficient involves only terms that are in principle readily available to the utility. The same coefficient applies for each technology under consideration, modifying estimates of externalities to account for the second-best policy context.

$$a^{x} = \left(\frac{A + 1}{B + C + 1}\right)(D'_{x}) \tag{1}$$

The full definition for this formula is provided in a footnote.[5] The

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[5]The terms in the equation are defined as follows:

$A = \varepsilon_{w}\left(1 - \frac{K'}{p}\right)$ representing the deviation between price and marginal cost

$B = \frac{\varepsilon_{u,p} u}{w\ p}\left(\frac{\partial F}{\partial e^{u}} s^{u}\right)$ representing the effects of bypass

$C = \frac{\varepsilon_{w}}{p}\left(\frac{\partial F}{\partial e^{y}} s^{y}\right)$ representing the reference technology

$D'_{x} = \frac{\partial F}{\partial e^{x}} s^{x}$

where:

a^{x} = the optimal adder (\$ / kWh)

ε_{w} = price elasticity of demand for utility supplied electricity

K' = marginal cost of a reference technology y (\$)

p = the price of utility supplied electricity (\$)

$\varepsilon_{u,p}$ = cross price elasticity of demand for unregulated supplies with respect to the price of electricity

u = quantity of unregulated supply currently consumed (kWh equivalent)

(continued...)

coefficient adjusting the estimate of externality involves information about a reference technology represented here as technology y, although the choice of a reference technology is arbitrary. One possibility would be to designate existing capacity as a reference technology. What is important is that the formula for each technology involves the same reference technology and that the adders be considered in tandem.[6] The term A in the numerator of the expression reflects the benefits of moving toward marginal cost pricing.[7] The term A could be positive if price is less than the marginal cost of the reference technology, or it could be negative if price is greater than the marginal cost of the reference technology. If prices are close to marginal cost, the term A is small and the numerator equals approximately 1. The numerator will almost always be positive, unless price is much greater than marginal cost, and even then demand would have to be very inelastic.

In the B term in the denominator, appear the cross-elasticity of demand between regulated and unregulated supplies of energy services, and the externalities associated with substitution toward the unregulated supply. Under usual assumptions this externality estimate is positive (the effects are undesirable). If the cross-elasticity is large and the externalities from unregulated supplies are also large, then B will be large. This will tend to reduce the coefficient and the specified adder. The reason is that the likelihood of driving customers away from

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[5](...continued)

w = quantity of utility supplied electricity consumed (kWh)

$F = F(e^i)$ = damage function ($)

e^i = emissions or other effects associated with technology i (potentially vector valued)

$\frac{\partial F}{\partial e^i}$ = marginal damage associated with technology i ($ / emission or effect)

s^i = emission rate per kWh for technology i (emission or effect / kWh).

[6]The reference technology in this formulation is denoted y. The reason a reference technology is necessary is that since the adders are not actually paid, it is their relative magnitude rather than their absolute value that is significant. Consequently the formula that is reported is a sufficient condition, but not a necessary condition, for the optimal adder. A necessary condition is that the *difference* between the adder for technology x and the reference technology y be equal to the given coefficient times the *difference* in the marginal damage terms (the second terms on the right hand side) for the two technologies.

[7]This is most transparent when technology y is "clean" meaning there are no positive or negative externalities associated with its production, then the marginal private cost of production equals marginal social cost, and the expression in parentheses depends directly on the deviation between price and marginal social cost.

regulated supplies of electricity is large and the social costs are large, so this unintended consequence of social costing mitigates against its application. Also in the denominator is a term representing the externalities for the reference technology (*C*). In all plausible cases we find the denominator is positive. Hence, the greater the measure of externality associated with the technology (the ·term), the greater would be the optimal adder, as intuition would suggest.

If the cross-elasticity is large and the externalities from unregulated supplies are also large, ... This will tend to reduce the coefficient and the specified adder.

To calibrate and apply this model requires specific information about the customer service territory in the reference environment under study. Hence, this approach reinforces the assertion which has been made at several points in this document and for several reasons—social costing policies must be location specific due to the regulatory environment as well as due to the geographical aspect of externality estimates. In the remainder of this discussion we apply this model to a service territory that approximates that which would apply for the Southeast Reference environment in Tennessee. We discuss specific assumptions about the customer service territory for our Southeast Reference environment in Chapter 6. In the application to specific fuel cycles the model is adapted to include multiple customer classes, multiple pollutants associated with specific technologies, and multiple technologies for self-generation.[8]

3. AN APPLICATION TO THE SOUTHEAST REFERENCE ENVIRONMENT

The southeast reference environment is located in Tennessee, an area that is served primarily by a large number of publicly owned and operated utilities. The bulk of electricity supply to utilities in Tennessee is provided by the Tennessee Valley Authority (TVA), which operates an integrated transmission system throughout Tennessee and some neighboring areas. We characterize the relevant service territory to be served by a generation facility in the Southeast Reference environment as the State of Tennessee.

Transforming estimates of externality to obtain an optimal adder requires several key pieces of data for the electric utility and for its customers. The

[8]In addition it can be readily adapted to reflect various descriptions of the pricing policies of the utility regulator.

required utility data include the price of electricity by customer class and the marginal cost of delivered electricity generated with the reference technology. Customer data include the own-price elasticity of demand for electricity, cross-price elasticities of demand for other fuels relative to the price of electricity, and quantities of electricity and other fuels consumed by customer class. In addition, the formula requires estimates of the marginal emission rates of pollutants (external effects) associated with generating electricity using the reference technology and with using substitute fuels. Also required are estimates of externalities for these various external impacts. The following paragraphs describe how each of these data items was calculated and the sources used for inputs to these calculations.

Average prices for this region for residential and commercial/industrial classes of customers are presented in Table 1. Prices were calculated by dividing total operating revenues of all Tennessee municipal utilities and direct sales to customers by TVA for a particular customer class by total kilowatt-hour sales across all Tennessee municipal utilities and TVA to customers in that class.[9]

Table 1. Electricity price by customer class

Customer class	Average price (cents/kWh)
Residential	5.61
Commercial/Industrial	5.35
Aggregate	5.46

3.1 RESIDENTIAL ENERGY CONSUMPTION AND EXPENDITURE SHARES

Residential energy consumption for fuel oil, electricity, natural gas and gasoline are based on residential energy expenditure estimates and energy prices for the residential sector.[10] We divided total expenditures for each type of energy by the relevant energy price to find total quantity of each type of energy consumed. These data were available for both the United States as a whole and for Tennessee. Residential expenditure shares of each of the four energy types are calculated using expenditure data from the State Price and Expenditure Report (1990). We used total United States Household Consumer Expenditures from the Bureau of Labor Statistics (BLS) to calculate energy expenditures as a percentage

[9]These calculations were preformed using data for 1991 from Tables 40 and 46 of Financial Statistics of Major Publicly Owned Utilities published by the Energy Information Agency.

[10]Energy Information Administration (1992).

of total expenditures. The estimates of energy expenditures as a percentage of total expenditures for Tennessee are based on per household consumer expenditures for the Southeastern region (from BLS) multiplied by the number of households in Tennessee.

3.2 COMMERCIAL/INDUSTRIAL ENERGY CONSUMPTION

The framework that we use to calculate energy consumption by customer class for commercial/industrial customers and associated own- and cross-price elasticities for 35 industrial sectors is that provided in Jorgenson and Wilcoxen (1990). We used several sources to calculate the quantities of each type of energy consumed by each of these sectors. Consumption of each fuel type by each of the sectors for the United States and Tennessee is estimated for 1989 and is expressed in trillions of Btu's.

... calculate energy consumption by customer class for commercial/industrial customers and associated own- and cross-price elasticities ... provided in Jorgenson and Wilcoxen

Energy consumed by the manufacturing sectors was calculated based on energy consumption data from the Manufacturing Energy Consumption Survey.[11] These data which are national in geographic scope are combined with Gross Product data by industry for the U.S. to calculate energy input shares by energy type for each industry which are then applied to State level Gross Product data for Tennessee. Energy consumed by the mining sectors was calculated based on energy consumption data for mining activity in Tennessee from the 1987 Census of Mineral Industries.[12] Since neither of these sources presented the energy consumption for 1989, we used Gross State Product[13] data broken down by sector of the economy for the years for which the data were reported to calculate a ratio of energy consumed to dollars of output. This ratio was then multiplied by the Gross State Product for each of the sectors in 1989 to estimate total energy consumed in that year.

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[11]Energy Information Administration (1991).

[12]Bureau of the Census, U.S. Department of Commerce (1990).

[13]Bureau of Economic Analysis, U.S. Department of Commerce, released in the Survey of Current Business, December 1991.

We used a different method to calculate energy consumed by the non-manufacturing and non-mining sectors of the economy. We relied on Input/Output (I/O) tables from the Bureau of Economic Analysis for the bulk of these calculations. The 1987 I/O tables present data aggregated to 85 sectors and the 1982 I/O tables present data at a disaggregated 527 sector level. We aggregated the 85 sectors into the 35 specified in the Jorgenson/Wilcoxen model and identified energy sectors of the economy in the 1987 tables. This gave us a rough estimate of types and quantities (by knowing energy prices for each sector) for the 35 sectors.

We used the disaggregated 1982 tables to refine this estimate. The 85 sector table (1987) was not detailed enough to differentiate between energy and other output within aggregate energy producing industries. The 527 sector tables allowed us to calculate the percentage of the total expenditures that could be attributed to energy expenditures for each of the energy sectors. We multiplied this percentage by the expenditures in 1987 for each of the energy sectors to get a more accurate estimate of energy expenditures for each of the 35 sectors. The energy expenditures were then translated into Btu's consumed using sector-specific energy prices.[14]

We then used the Gross State Product data for the years corresponding to the calculations to develop ratios of energy consumed to dollars of output. These ratios were multiplied by 1989 Gross State Product data to estimate total Btu's consumed in 1989. We used the ratios of energy consumed to dollars of output at a national level to estimate energy consumed by the 35 sectors in Tennessee. We applied these ratios to 1989 Tennessee Gross State Output to calculate types and quantities of energy consumed by each of the sectors.

3.3 RESIDENTIAL ELASTICITIES

Residential own- and cross-price elasticities of demand for electricity and other fuels were calculated using the parameter estimates from a system of household demand equations estimated by Jorgenson, Slesnick and Stoker (JSS), (1988), combined with information on energy expenditure shares for households in Tennessee. The demand equations in the JSS model are derived from a indirect translog utility function. This model uses a hierarchical structure that assumes that energy consumption forms a weakly separable subset of the arguments in the household utility function. In this hierarchy the uppermost system of equations corresponds to the household demands for 1) food and clothing, 2) other nondurable goods, 3) capital services, 4) consumer services and 5) energy. In a

[14]Energy Information Administration (1992).

second level of the hierarchy, energy consumption is divided into: i) electricity, ii) natural gas, iii) fuel oil and iv) gasoline.

This system of equations was estimated using combined cross-sectional data for households to capture the effects of demographic variables and aggregate time-series data to obtain variation in prices. The census region where the household was located was among the many demographic variables included in the model and therefore the price coefficients were estimated subject to controls for regional variations in household expenditure patterns. For purposes of our calculations we combine data for estimated energy expenditure shares for Tennessee households in 1989 with the appropriate price coefficients from the estimated model to calculate net Allen own- and cross-price elasticities of demand for each type of energy.[15] The elasticities estimates we obtain are extremely long-run and assume that all capital stocks including the stock of housing and household appliances have been optimally adjusted to prevailing energy prices. The elasticities used in this analysis are reported in Table 2.

Table 2. Household energy demand elasticities (derived from Jorgenson, Stoker and Slesnick household demand model)

Type of energy	Net elasticity of household demand with respect to price of electricity in Tennessee
Electricity	-2.24
Natural gas	-0.66
Fuel oil	1.89
Gasoline*	0.91

* We did not include gasoline as a substitute energy form in our analysis.

Since the estimate of the own-price elasticity of demand for electricity derived using this model (-2.24) is quite high relative to other estimates found in the literature, we also performed the analysis using a lower own-price elasticity of demand for electricity of -0.4.[16] We used the same cross-price elasticities in both cases.

[15]These estimated expenditure shares are based on actual data and are not the expenditure shares that the model might predict for the typical Tennessee household.

[16]Bohi and Zimmerman (1984). Unfortunately these authors provide no information about cross-price elasticities. In principle, cross-price elasticities would change if own price elasticities change.

3.4 COMMERCIAL/INDUSTRIAL ELASTICITIES

Commercial and industrial own- and cross-price elasticities of demand for electricity come from a tiered model of industrial input demands by sector of the economy for 35 sectors developed by Jorgenson and Wilcoxen. As with the residential model, this tiered model structure assumes that demand for individual energy inputs is weakly separable from demand for other non-energy inputs. The first tier of inputs include capital, labor, materials and energy. Within the energy tier, the model breaks out demand for coal, crude petroleum, refined petroleum, electricity and natural gas. The input demand equations are derived from a translog cost function. The elasticities are calculated by combining estimated price coefficients and data on actual national expenditure shares.[17] Peter Wilcoxen kindly provided us with the coefficient estimates for each of the 35 input share equations and with national cost share information for all relevant inputs for each of the 35 sectors in 1990. The elasticities used in this exercise are net Allen elasticities of demand which means that total output of each energy-consuming sector is held fixed, but consumption of total energy is allowed to adjust in response to a change in the price of electricity. The equations for the commercial/industrial elasticities are similar to those for the residential section.

The formula for adjusting externalities to find the optimal adder involves the substitution activities of a particular customer class as defined by the rate class. Due to a lack of disaggregate rate data, all industrial and commercial customers are lumped into one customer class. However, own- and cross-price elasticities of demand are defined separately for each of the 35 sectors in the economy. Emissions rates are also defined separately for commercial and industrial customers for some of the substitute fuel technologies. These disaggregate elasticities and associated emissions rates for each of the different fuel options are aggregated in the process of calculating the adjustment factor. The aggregate elasticities are listed in Table 3.

Once again, since -1.37 is higher than many of the own-price elasticity of demand estimates which appear in the literature, we also calculated the adjustment factor using an own-price elasticity of demand for electricity of -0.6 keeping the same cross-price elasticity assumptions.

[17]National expenditure shares were used since most of the Tennessee energy consumption and expenditure data were estimated by assuming that energy intensity in the industry in Tennessee was identical to national energy intensity.

Table 3. Commercial/industrial energy demand elasticities (based on Jorgenson and Wilcoxen industrial sector input demand model)

Type of energy	Net aggregate commercial/industrial elasticity of demand with respect to the price of electricity
Electricity	-1.37
Coal	0.15
Fuel oil	0.06
Natural gas	0.12

3.5 MARGINAL COST OF THE REFERENCE TECHNOLOGY

The marginal cost that we find is the marginal cost of delivered power. This estimate includes the marginal cost of generation and the marginal cost of transmission and distribution. For purposes of this exercise, we define the marginal cost as the average long-run incremental cost of new generation and associated transmission and distribution.

The selected reference technology is a new pulverized coal (PC) plant with wet flue gas desulfurization.[18] The generation costs are based on EPRI (1989) estimates of costs (capital costs, fixed operation and maintenance (O&M) and variable O&M) and performance characteristics for a 500 MW pulverized coal unit. The capital cost of the plant is amortized over 30 years using the selected real interest rate (we consider three possible long-run real interest rates of 3, 4 and 5 percent respectively). The annual capital cost is then converted to capital cost per kilowatt-hour assuming an average annual capacity factor of 65 percent. Fuel costs per kWh are based on the delivered costs of coal at the TVA's Cumberland plant[19] and on the heat rate listed for a 500 MW PF facility in EPRI (1989). The marginal cost also includes the cost of lime needed for scrubbing purposes and the cost of sludge disposal. All generation and scrubbing costs were adjusted for line losses which were assumed to equal 9% of generated electricity.

The transmission and distribution (T&D) O&M expenses were estimated by applying the ratio of T&D O&M to generation O&M for all Tennessee utilities

[18]The technology characterization in this example differs slightly from that used to describe a similar technology in our study of the coal fuel cycle. The distinction is unimportant to the illustration of the methodology in this paper and would have a very small effect on the numerical results.

[19]Energy Information Administration (1993 p. 46).

to the estimate of O&M per kWh for a new scrubbed PC facility.[20] T&D capital expenditures were obtained by applying the average national ratio of T&D investment to generation investment at investor-owned utilities over the past 10 years to the capital cost for a new scrubbed PC facility.[21] These capital costs were also amortized over 30 years. The resulting marginal cost numbers for each of the three interest rate assumptions are reported in Table 4.

3.6 EMISSIONS RATES

The emissions factors that were used in our calculations are listed in Table 5. Most of these emissions factors were taken directly from the Environmental Protection Agency's Compilation of Air Pollutant Emission Factors.[22] We made assumptions concerning various aspects of emission factor equations which are summarized below:

Table 4. Marginal cost of electricity

Interest rate assumption	Long-run marginal cost of electricity (cents/kWh)
3% real	4.93
4% real	5.23
5% real	5.53

3.7 NATURAL GAS COMBUSTION

The industrial and residential values are identical with the exception of oxides of nitrogen. These emission factors come primarily from Caverhill (1991), with the industrial NO_x values supported by EPA's AP-42 (U.S. EPA 1985).

[20]Energy Information Administration (1993b, pp. 185 - 190).

[21]Kahn and Gilbert (1993, p. 15).

[22]U.S. EPA, Office of Air and Radiation (1985). "Compilation of Air Pollutant Emission Factors, Volume 1: Stationary Source and Area Sources, 4th Edition," (September).

3.8 BITUMINOUS COAL COMBUSTION

... emissions factors were taken directly from the Environmental Protection Agency's Compilation of Air Pollutant Emission Factors.

All emissions from coal combustion come from the industrial sector. The emissions factors assume combustion of pulverized coal with wet flue gas desulfurization to achieve 90% sulfur removal and a coal with 2% sulfur by weight.

Table 5. Emission factors by fuel type and customer class (Pounds per million Btu's combusted)

	TSP	NO_x	SO_2
Industrial*			
Natural gas combustion	.0091	.13579	.0006
Bituminous coal combustion	.06155	.60961	.93494
Fuel oil combustion	.02622	.14421	.02048
Residential			
Natural gas combustion	.0091	.095	.0006
Fuel oil combustion	.03277	.12979	.02048

*The emission rates for commercial customers are identical to industrial rates except for those associated with natural gas combustion which are identical to residential emission rates.

TSP = total suspended particulates
NO_x = nitrogen oxides
SO_2 = sulfur dioxide

3.9 FUEL OIL COMBUSTION

All industrial emissions factors assume industrial boilers (gross heat rate = 10×10^6 to 10×10^7 Btu's/hour) and a low sulfur fuel (2% sulfur by weight).

Residential emissions factors assume residential boilers (gross heat rate = $<0.5 \times 10^6$ Btu's/hour) and distillate oil.

3.10 EXTERNALITIES

The three external impacts of electricity generation or alternative energy use included in this exercise are emissions of NO_x, SO_2 and TSP during combustion of fuel to generate electricity or of substitute fuels by the end-user. The estimates of externality (marginal damage) for each of these three pollutants are arbitrary values that are not based on estimates developed in this study.[23] This one set of externalities is applied to emissions from centralized electricity production and to emissions from local use of substitute fuels. This is technically incorrect since emissions of the same pollutant from large centralized facilities with tall stacks and from small decentralized boilers will have different impacts on human health and the environment and therefore should have different externalities. A more complete analysis would differentiate the externalities from different sources for the same pollutant. The externality estimates adopted for purposes of this study are presented in Table 6.

Table 6. Externality estimates for included pollutants

Pollutant	Externality ($1992/lb)
SO_2	$2.03
NO_2	$0.82
TSP	$1.19

3.11 ADJUSTMENT FACTORS TO CALCULATE OPTIMAL ADDERS

We calculated the factor for adjustment of externality estimates to arrive at an optimal adder for using prices and demand elasticities for each of the two customer classes under three different assumptions regarding marginal costs (resulting from different assumptions about real interest rates) and two different own-price demand elasticities for electricity. These factors were calculated using equation (1) under the assumption of a constant emission rate and a linear damage function. These calculations assumed that pulverized coal with wet flue gas desulfurization is the reference technology for a Tennessee utility. The resulting factors are presented in Table 7.

The adjustment factors in this example are all equal to or greater than 1 indicating that the optimal adders will exceed the externality estimates. The factors exceed one because the externalities associated with use of the substitute

[23]The values are the same as those suggested by Ottinger et. al. (1990). The values are arbitrary for the purpose of this study because they fail to account for regional variation in damages.

energy technologies are less than the externalities associated with generating electricity. Note that these factors are greater than one even when the price of electricity is greater than the marginal cost of delivered electricity, as is true for the residential price under all three marginal cost scenarios (see Table 1). The factors tend to increase with the cost of electricity and with the absolute value of the own-price elasticity of demand for electricity.

Table 7. Externality adjustment factors for Tennessee (with scrubbed pulverized coal as a reference technology)

	MC = 4.93 cents (3% interest rate)	MC = 5.23 cents (4% interest rate)	MC = 5.53 cents (5% interest rate)
Based on Residential Price (5.61 cents/kWh)			
own-price elasticity = -2.24	1.152	1.336	1.531
own-price elasticity = -0.4	1.018	1.040	1.064
Based on Commercial/Industrial Price (5.35 cents/kWh)			
own-price elasticity = -1.37	1.129	1.223	1.323
own-price elasticity = -0.6	1.032	1.067	1.105

We emphasize that these adjustment factors are for rough estimates intended to illustrate the use of this methodology. We were not able to obtain precise information for the relevant service territory to facilitate a more complete analysis. In addition, the externality estimates that appear in the denominator of equation (1) include only the marginal damages for three pollutants, rather than the array of externalities that have been identified in this study.

The adjustment factors in this example are all equal to or greater than 1 indicating that the optimal adders will exceed the externality estimates.

This analysis is broad enough to capture the first round of general equilibrium effects that occur from the application of social costing to just one sector of the economy—which is termed "bypass" because customers leave the utility grid through substitution to alternative technologies or to unregulated sources for the production of energy services. An important limitation of this model concerns the broader general equilibrium effects that may occur because of the ability of consumers and producers to substitute away from energy services entirely, toward other services or factors of production such as labor, capital, or the use of other resources. Furthermore this model ignores the irreversibility of capacity investments, a factor that affects resource planning decisions.

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PART VI

SIGNALING UNCERTAINTY AND QUALITY IN INFORMATION

PAPER 20 NUMERICAL UNIT SPREAD ASSESSMENT PEDIGREE (NUSAP)

PAPER 20[1]

NUMERICAL UNIT SPREAD ASSESSMENT PEDIGREE (NUSAP)

1. INTRODUCTION

Data estimates and information require a system for signaling the uncertainty and quality of the entries for users. Few of the entries will be known with certainty, or even generally agreed upon as the prevalent quantity or relation. For example, the ecological or health responses of resources exposed to energy-related pollutants cannot be known with certainty given current knowledge of the relationships. The monetary valuations associated with the imperfectly known impacts are also uncertain and sometimes controversial. To leave entries standing alone without signaling their uncertainty and quality would overstate the precision with which the entries are known. In addition, signaling the uncertainty and quality for entries will indicate areas where further study is needed most.

Uncertainty and quality are signaled through a notational system named NUSAP as an acronym for its categories. NUSAP was developed by Funtowicz and Ravetz to provide a "quality control" of quantitative information.[2] We have adapted the NUSAP system for signaling the uncertainty and quality of quantitative information to be used in estimating the emissions, impacts, and external costs of fuel cycles. Uncertainty refers to the spread of plausible values for a cell entry and the level of confidence placed in a quantitative statement. Quality refers to both an entry's worth as a piece of information and the credibility of the theory, data, and methods used to generate the entry.

NUSAP was developed by Funtowicz and Ravetz to provide a "quality control" of quantitative information.

[1]Based largely on a working paper by A. Schaffhauser Jr.

[2] Funtowicz, S. and Ravetz, J. 1990. *Uncertainty and Quality in Science for Policy.* Kluwer Academic Publishers, Dordrecht, The Netherlands.

The NUSAP system signals uncertainty by stating the spread of values associated with a numerical entry. A number standing alone does not convey the uncertainty about its true value. In fact, it falsely suggests that it is known with certainty. Thus, the uncertainty must be signaled. One may get a better feel of the need to signal uncertainty with numeric entries by contrasting our case of uncertainty to cases where probability measures are used. Probabilities quantify the likelihood or frequency of an occurrence, or quantify the degree to which an hypothesis is believed to be true. Likelihood, frequency, and degree of belief calibrate the lack of certainty of an element (where the element is the set of all possible occurrences or hypotheses). The lack of certainty surrounding the element has already been signaled by the mere existence of more than one possible occurrence or hypothesis. However, a number standing alone does not convey a lack of certainty. Where the element itself is a piece of quantitative information, as with a numerical entry, the uncertainty of the element must be signaled explicitly, in its own numerical terms. This is done by designating an interval of values. The interval of all possible values may be immense, and determining where "possible" begins and "impossible" ends would be perplexing. (After all, isn't *anything* possible?) This makes such an interval of possible values unworkable. Thus, the bounds of the interval are demarcated by designating a level of confidence; this "confidence interval" is called a spread.

The spread of NUSAP is not necessarily generated by statistical analysis. If there does not exist a statistical distribution, then the NUSAP spread will be generated by the subjective judgment of experts. Also, a statistical distribution may not fully capture the uncertainty about a value. Probability distributions are generated from all sorts of activities that are unrelated (or related only by analogy) to the ideals out of which probability theory developed (i.e., the logical structure of games of chance, random variation in repeated controlled experiments, and long-run frequencies of outcomes). Many times distributions are simply assumed. Probability distributions were designed around the ideals, and therefore they may not fully represent the uncertainty in less-than-ideal cases. Thus, a statistical confidence interval may not capture all of the uncertainty involved. If a statistical interval fails to fully represent all significant uncertainty about a value, a spread should be supplied from expert judgement. (A statistical confidence interval's adequacy is dealt with further under the section "Spread of Value" on p. 20-5.)

In addition to signaling uncertainty, NUSAP also signals quality of entries. Quality is the second facet of indeterminacy that is not captured by uncertainty. Quality pertains to the state of knowledge about an element. In evaluating the qualitative aspects of an entry, we address the questions "What do we know?" and "How ignorant are we?" The term "qualitative" here means both "non-numerical" and "goodness." Signaling uncertainty alone does not assess the goodness of an entry. The quality of entries is signaled in the

Assessment and Pedigree categories of NUSAP. The Assessment category evaluates an entry's worth as a piece of information, and the Pedigree category evaluates the source or production process of the piece of information. The NUSAP scheme is presented in the box on p. 20-4.

2. NUMERICAL INFORMATION

The N of NUSAP may represent numerical entries to a data base. These entries may be constants, coefficients, or dependent variables of functions or models. Numerical entries may be, for example, rates of emission of a pollutant, incremental effects of a pollutant on an ecological asset, or a willingness to pay to avoid a health effect.

The N space of NUSAP may also be filled with the notation [LB, UB] for lower bound, upper bound. This notation would be used when an expert feels that the trifling state of knowledge does not warrant designating a specific numerical entry. A spread will exist even though an exact numerical estimate within the spread is uninformative. Thus, the N space of NUSAP will be filled with the label [LB, UB], and S_2 will contain the numbers denoting the upper and lower bound of the spread.[3]

The N space of NUSAP may also be filled with a variable name. A variable name would be used if the entry were an equation or table. An equation would be entered to represent a general-use function or model, that is, one that may be used with the independent variables for many different cases to obtain a number for the dependent variable in each case. N would hold the dependent variable name of the function or model. If the entry were a table, N would hold the name of the output column of the table. The equation denoting the general-purpose function or model or the table which supplies the output column would be included in the data record but not as the N of NUSAP. There will be a separate entry point to collect the equation when data is entered into the data base. The equation or table will be displayed along with the estimation methods part of NUSAP when the data record is retrieved.

A note on practice would fill the N space of NUSAP if the entry were a method for quantifying an accounting framework parameter. For example, a quasi-experimental design for determining an ecological impact would constitute

[3] In the case of complete ignorance, the spread would be the upper and lower bound of all of the possible values (in the extreme case, positive and negative infinity), and the level of confidence would be 100%.

The NUSAP Scheme

Numerical information

Numeral
Notation
Variable Name
Note on Practice

Unit:

U_1: Units of measurement
U_2: Statistic used for value; for example, mean (ME), mode (MD), median
(MN), lower bound (LB), upper bound (UB), expected value (EV),
or no distribution (ND)

Spread of value

S_1: Level of confidence
S_2: Spread lower and upper bound (S[LB, UB])

Assessment of value:

I_1: Informative value based on spread
I_2: Informative value based on application
G: Generalizability to other applications
R: Robustness of value over time

Pedigree:

T: Theoretical basis (and application of theory)
D: Data inputs
E: Estimation methods
M: Estimation metric

Notation:

(N, U_1, U_2): $(S_1, S_2[LB, UB], A[I_1, I_2, G, R])$: $(P[T,D,E,M])$

a quantification practice. Also, quantification practices will be very useful entries if we know how an accounting framework parameter should be produced, but data or knowledge is lacking to complete the quantification process. The note on practice is a brief description of the output of the quantification practice which fills the N space of NUSAP. There will be a separate entry point for the quantification practice, and it will be displayed along with the estimation methods part of NUSAP, as with an equation or table entry.

3. UNIT

U_1 denotes the units of measurement of the numerical entry. For example, U_1 might be kilograms of a pollutant per hour, hectares of crops, or 1990 dollars. U_2 is the statistic used for the numerical entry–for example, any summary statistic of a distribution, no distribution if a statistic is not used, or an expected value of a model.

Because of the variety of entries possible for the N place of NUSAP, some explanation of what will fill the U places is necessary. For coefficients, the units are those of the dependent variable. This makes sense because it is the dependent variable for which a coefficient denotes a rate of change. For those cases where ignorance warrants the "[LB, UB]" label, U would be the units of the undesignated specific numeral. For general-purpose model entries, U would be the units of the dependent variable. For table entries, U would be the units of the output column. For quantification practice entries, U would be the units of the output of the practice.

4. SPREAD OF VALUE

The spread category of NUSAP contains a generalized confidence interval of the entry. The purpose of the spread is to signal uncertainty. S_1 denotes the level of confidence with which the expert believes the true value lies within the interval. The interval is denoted by its lower and upper bound [LB, UB]. The confidence interval used for the spread category of NUSAP is more general in scope than statistical confidence intervals, but statistical confidence intervals may still be spreads.

The spread category of NUSAP contains a generalized confidence interval of the entry.

A NUSAP spread is the same as a statistical confidence interval if (rule 1) there exists a statistical distribution of the value for N, and (rule 2) the expert believes the distribution fully captures all of the uncertainty about the value for N. Thus, NUSAP spreads differ from statistical confidence intervals because they can be produced by subjective judgments of experts. If there is not a probability distribution (failure of rule 1), then the uncertainty must be signaled by the subjective judgement of the expert. Thus, a NUSAP spread may be a confidence interval generated strictly by subjective judgment.

If a statistical confidence interval does not adequately signal the uncertainty of a numeric entry (failure of rule 2), then the expert designates a NUSAP spread based on both the statistical confidence interval and his expert judgement. An example of a statistical interval diverging from a NUSAP spread due to a failure of rule 2 is a practice in the health impacts field. Experts in this field take the upper bound estimate produced by health impact models. In their expert judgment, the upper bound is a better estimate than a central estimate because of a regard for a "margin of safety." Thus, a statistical confidence interval centered on a mean would be attributed a lower subjective confidence than statistical confidence. Thus, the spread at the same degree of confidence as the statistical interval would be centered around the upper bound in order to reflect the "tradecraft" practices of experts in the field. In general, the spread should reflect tradecraft expertise gained through experience in various fields. Thus, experts should apply subjective judgement to statistical confidence intervals.

To standardize the spread of entries, where possible, intervals should be designated for a 90% degree of confidence. That is, the expert has 90% confidence that the true value lies within the range denoted by [LB, UB]. However, separating spread into the S_1 and S_2 components allows the flexibility to designate an interval with lower or higher confidence. This is warranted if a piece of literature states a confidence interval other than the 90% interval (provided it adequately captures the uncertainty). There are also cases where convention or disciplinary practice warrants a confidence level different from 90% confidence.

For general-purpose models or functions, S_2 would be entered as a +,- range (or a x,÷ range) of the model or function's potential output. The +,- range may be +,- some percentage of the as-of-yet uncalculated dependent variable, for example, [+,- X%]. Or, if a distribution is implied, the range could be +,- some number of standard deviations from the dependent variable, for example, [+,- X sd]. This +,- range can be gained from the literature that presented the general-purpose model. Many authors will expectedly indicate a projected range of accuracy of a model they present.

For tables, as with models or functions, S_2 is entered as a +,- range. S_2 would be entered as a +,- range of the table's potential output for which the expert has S_1 confidence that the true value would lie within the range. In other words, S_2 indicates the plausible range surrounding potential table outputs.

However, with general-purpose models or tables, it seems objectionable to designate S_2 until the application is known and the dependent variable or output has been calculated. After all, the spread of the dependent variable or output will depend on the case at hand. Thus, one could design a data base to prompt users to update S_2 when the model, table, or practice is applied. In the meantime, the only information on spread that the user can access is what the expert judges as a +,- range. Given this, even a very rough judgement of S_2 is helpful. One can gain an estimate of the +,- range by imputing typical or plausible values for independent variables or table inputs and noting how the uncertainty about the parameters influences the +,- range of the dependent variable's or output's confidence interval. Where this rough estimate of S_2 would not be helpful, spread should be left blank to be filled in when the model or table is applied.

A spread for a practice would be highly subjective because in many cases where a practice entry is needed, the extent of the uncertainty is itself very uncertain. A useful spread may be designated based on experience with the practice. For example, one may ask how well the practice has worked in the past. The goal is to provide users with a conception of how uncertain the output of the practice is expected to be. Qualifiers such as "the practice's output would be highly speculative at best" or "the practice would provide an order of magnitude estimate" may be more informative than numbers. Experts could also provide notes about the uncertainty issues involved.

5. ASSESSMENT OF VALUE

The entries are assessed in regard to their merit as pieces of information. In other words, an assessment is made of each entry based on its worth for providing information on the emissions, impacts, and external costs of fuel cycles. Entries will be assessed on four aspects: (1) informative value based on spread, (2) informative value based on application, (3) generalizability to other applications or sample spaces, and (4) robustness over time. Each of these aspects will be assessed by rating them low, medium, or high, as indicated in the box "Assessment Indicators" on p. 20-9. Within the indicators, however, each field will have particular aspects that influence the rating. For example, in the valuation of externalities field, the

... an assessment is made of each entry based on its worth for providing information ...

characteristics of the population and the characteristics of the time period are the main features that influence the generalizability of entries.

5.1 INFORMATIVE VALUE BASED ON SPREAD (I_1)

Informative Value Based on Spread provides an assessment of the ignorance that is overcome by the studies, methods, or works that generated the entry. To assess this, one compares the uncertainty about the entry's value now (posterior) to the uncertainty about the entry before the study that produced the entry was conducted (prior). More specifically, the posterior range of plausible values for the entry is compared to the prior range of plausible values for the entry. The extent to which the posterior range is narrower than the prior range indicates the informative value of the entry.[4]

This technique of measuring informative value based on spread is analogous to the traditional method of measuring information content in a probability context. In the probability context, the information provided by an observation is measured by comparing probabilities before and after an observation. The extent to which the posterior probability is different from the prior probability is regarded as the increase in information provided by the observation.[5]

5.2 INFORMATIVE VALUE BASED ON APPLICATION (I_2)

Informative Value Based on Application provides an assessment of how much the uncertainty about the entry affects the quality of the result of analysis using the entry. I_2 is determined from a rough sensitivity analysis of the result for the different plausible values residing in the spread. Thus, I_2 is dependent on the application for which the entry is used. Due to this dependency on application, I_2 will be based on a first guess at the application for the entry. This may consist of merely contemplating the effect of the entry's uncertainty on the accuracy of the entry's likely uses. However, there will be cases where

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[4] The informative value may also be reflected as an increased level of confidence in the spread. However, to standardize the assessment of informative value, we propose that the level of confidence not change. Thus, the plausible range of values before and after is compared, where the same minimum level of confidence required to be "plausible" is used for the prior and posterior range of values.

[5] One advantage of measuring informative value of a spread over that of a probability measure is that it overcomes the problem of increased information without a change in probability. In the probability context, the posterior probability may not be different from the prior, while the observation still provides information. The observation may increase the credibility of the probability without changing its value. In the context of spread, however, the increase in credibility would not be overlooked. It would be reflected as a narrowing of the range of posterior plausible values at the same level of confidence.

the eventual application is known better than in other cases. Thus, as with spread for general-purpose models, I_2 will be a rough estimate until the application is known, at which time it will be revised. Experts are encouraged to assess a tentative I_2 as long as they feel it might be useful to the users of the data base.

5.3 GENERALIZABILITY TO OTHER APPLICATIONS (G)

Generalizability to Other Applications provides an assessment of the usefulness of the entry for other potential applications other than the application for which the value was originally generated. If an entry was produced for a particular purpose or for general purposes, the expert is to assess the validity of the entry for other potential purposes. (A general purpose model will, by definition, score high on this assessment aspect.) If an expert is updating the NUSAP for a general-purpose model upon applying the model to a specific case, the generalizability assessment will convey the aptness of the general-purpose model for the case at hand. (This is actually the opposite of generalizability; it is "specificizability.")

Assessment Indicators

I_1:
- L: Many prior plausible values exist in spread.
- M: Spread is a fair amount narrower than range of prior plausible values.
- H: Spread is much narrower than range of prior plausible values.

I_2: (I_2 here is a first guess; it is to be refined when the particular application is considered.)
- L: The existence of other posterior plausible values (i.e., values in spread) matters for application.
- M: The existence of other posterior plausible values matters marginally.
- H: The existence of other posterior plausible values does not matter.

G: (based on application for which it was originally generated)
- L: does not generalize to other applications
- M: can be generalized with limitations
- H: easily generalized

R:
- L: highly perishable
- M: moderately perishable
- H: time independent

5.4 ROBUSTNESS OF VALUE OVER TIME (R)

Robustness of Value over Time provides an assessment of how valid the value is expected to remain in the future. In other words, how dependent is the value of an entry on temporal changes?

6. PEDIGREE

Pedigree signals the quality of entries by evaluating their sources. In other words, the pedigree exhibits an evaluation of the production mode of the quantitative information. Pedigree contains four categories: theoretical basis, data inputs, estimation methods, and estimation metric. An entry's quality is assessed on these categories by indicating a value between 1 and 5, as is described in the box "Pedigree Indicators" below. The ratings are given based on the expert's subjective judgement of the quality of the production mode on each aspect. For all aspects but theoretical basis, simply indicate a rating from unacceptable to excellent. Different fields will judge the aspects with different considerations because different points will be relevant for different fields. The experience of experts should guide their judgments. A special character will be used to indicate that a category is inapplicable, for example, if this was not an aspect of the entry's production.

Pedigree signals the quality of entries by evaluating their sources.

The estimation metric pedigree category requires some explanation. Estimation metric refers to the thing that is measured in the production of the estimate. Is it the object itself, or a proxy for something immeasurable? If it is a proxy, how good is the proxy?

7. COMMENTS

One of the strengths of the NUSAP system is that uncertainty and quality can be signaled in a brief cryptic and systematic format. However, this will also be a disadvantage when a user wants to know why an entry received a rating. Thus, there should be comment fields for each assessment and pedigree aspect. The comments to be input in these fields serve a purpose similar to those provided under the "explanation" subsection of the examples provided below. However, these comments may be brief since a user may refer back to the literature for more information. The comments will appraise the user of why a particular rating was received. N, U, and S may also need some further explanation.

Pedigree Indicators

T: theoretical basis
- 1: no theory or concepts
- 2: weak theory or concepts, controversial empirical support
- 3: weak theory, good empirical support
- 4: good theory, but one of competing theories
- 5: well-understood and accepted theory

D: data inputs
- 1: unacceptable
- 2: poor
- 3: fair
- 4: good
- 5: excellent

E: estimation methods
- 1: unacceptable
- 2: poor
- 3: fair
- 4: good
- 5: excellent

M: estimation metric (proxy or indicator for what we want to measure)
- 1: unacceptable
- 2: poor
- 3: fair
- 4: good
- 5: excellent

8. EXAMPLES

Example 1. Signal uncertainty and quality of the predicted competitive price of oil in 1983. It was generated from historical data and expert judgement through computer simulation (Funtowicz and Ravetz, p. 147). For illustrative purposes, it is rather detailed.

(N, U_1, U_2): (S_1, S_2[LB, UB], A[I_1,I_2,G,R]): (P[T,D,E,M])

(6, 1983$/bl, mean): (90%, [3, 11], [M, H, M, M]): ([1,2,3,3])

Explanation:

N: The number for predicted competitive price is 6.

U: U_1 It is stated as 1983 U.S. dollars per barrel and
U_2 is the mean of the generated distribution.

S: S_1 90% confidence interval
S_2 is from $3 per barrel to $11 per barrel based on statistical analysis.

A: I_1 The spread is a fair amount narrower than the range of prior plausible values. This is so because the actual 1983 price was $30/barrel, yet this study was needed to determine if this price was significantly lower than the competitive price that would have existed without the OPEC cartel's monopoly power. This means that, prior to the study, the spread of plausible values must have been a fair amount wider. Narrowing the spread to an upper bound of $11 makes it a fair amount narrower.

I_2 If the application is to determine if the cartel has significant market power or not, the number is highly informative. We can tell with a high degree of confidence that the $30/barrel price is not competitive.

G This estimate can be generalized as long as the same "relevant" conditions exist (Funtowicz and Ravetz, p.4).

R The estimate is moderately perishable over time. (It is good for the whole year of 1983.) As the oil market undergoes changes and more recent historical data becomes available, the simulation would have to be redone.

P: T There is no encompassing theory driving this; the simulation operates on historical data and expert judgement.

D The data are poor. The price of oil under certain conditions is measured imprecisely because of the uncertainty surrounding the definition of the conditions and the judgement that the condition exists or doesn't exist. The expert judgments required are of even lower quality because the Bayesian probability distributions of data input are subjective opinions (although based on a review of the literature).

E The estimation methods are average to fair. The estimation of the subjective probability distributions is of poor quality. The literature on subjective probability elicitation indicates that these estimates are not valid or consistent in many cases, as well as being subject to human judgment error and bias. The estimation done by the computer simulation is good provided that one key assumption is true. By putting credence in the simulation, historical data (tempered by expert judgment) can assumedly be used to predict what would have happened in the future. This assumption can not be validated. We do not know the actual 1983 competitive oil price to compare because it never existed. Logically, however, it is under the same influences as the oil price in the past. Thus, the assumption that historical data is valid for predicting what would have happened in the future is plausible. Also, there is peer acceptance of this assumption. However, the subjective judgments are more important than the computer simulation in determining the end result, that is, the prediction of price. This is evident because a less rigorous simulation, say, with a relatively small number of calculations from a hand-held calculator along with some interpolation, would still provide useful results as long as the data are of good quality. However, if the data are worthless, then no matter how sophisticated the computer simulation, the output will be useless. Thus, the method is given a fair rating.

M They want to know what the competitive oil price would have been in 1983. They measured the competitive oil price in the past and subjected the historical data to informed judgement. However, they are not actually measuring the 1983 competitive price. They are measuring means of distributions generated in many simulations of the competitive price fluctuation to predict the price. The assumption that future price depends on the same influences as past price is necessary for this to serve as a proxy. This lowers the quality of the estimate on this aspect. Thus, the metric gets a fair rating.

Example 2. Signal the uncertainty and quality of the general-purpose mathematical model used to predict future competitive prices of commodities. (It is the mathematical model used in the computer simulation in the above example for competitive oil prices.)

$(N,\ U_1,\ U_2)$: $(S_1,\ S_2[LB, UB],\ A[I_1, I_2, G, R])$: $(P[T,D,E,M])$

(P, money, mean): (90%, [x,÷ 2], [-, -, H, H]): ([2,2,4,3])

Explanation:

<u>N</u>: It is a price (P).

U: U_1 The price will be stated in monetary terms.

U_2 It generates a distribution, and the general practice in this field of economics is to take the mean.

S: S_1 The subjective level of confidence is 90% that the true value is within 2 x,÷ the value generated by the model.

S_2 Ideally, this interval should be given in terms of the distribution, that is, standard deviations. However, since the statistics of the distribution were not supplied by Funtowicz and Ravetz, a x,÷ factor is used. The interval is given based on the spread in the original study and subjective judgment about the accuracy of the model's output.

A: I_1 Because the application is not known, I_1 cannot be assessed. For different applications, there is too much variation in the prior knowledge of plausible prices for a useful assessment of I_1 to be supplied.

I_2 The application is unknown, so we do not know how varying the value by as much as a factor of 2 will affect the result at this point.

G The model is easily generalized to predict competitive prices for different commodities.

R The model is time independent.

P: T The model is based on economic theory. There is a long history of expertise in the field of predicting future commodity prices. The model is a standard form for translating experts' guesses into mathematical form. However, there must be many assumptions made in applying the model, and the predicted prices often differ greatly from the actual prices realized. Thus, the theoretical basis gets a poor rating.

D The model requires historical data and subjective judgments as input. The historical data is likely to be of good or even excellent quality. However, the subjective judgments are essential inputs to the model. Because the subjective probability distributions are of questionable validity and there is a high potential for bias with this data input, a rating of poor is assigned to the data inputs.

E The estimation done by the mathematical model is excellent provided that the assumptions made are true. We assume, by putting credence in the model, that historical data (tempered by expert judgment) can be used to predict what will happen in the future. This assumption has not been completely validated. It makes sense, however, that the future price is under the same influences as the past price. Thus, the assumption that this mathematical model is valid for predicting what will happen in the future is plausible. Also, there is peer acceptance of this assumption. Thus, it is given a good rating on this aspect.

M The model predicts a future competitive price of a commodity. The mathematical model parameterizes the factors believed to influence the competitive price. However, they are not actually measuring the future competitive price. Many assumptions are necessary for this to serve as a proxy. Thus, the metric is assigned a fair rating.

9. NUSAP DATA ENTRY FORM: SUMMARY EXPLANATION

N: Enter the number, notation, variable name, or note about practice.

$\underline{U}_1$: Enter the measure for the number, upper and lower bound, or variable (e.g., pounds). Also enter the time period for the entry (e.g., per hour).

$\underline{U}_2$: Enter the statistic which the number or variable is (e.g., mean, median, no distribution).

$\underline{S}_1$: Enter the degree of confidence of the spread. Use 90% whenever possible for standardization.

$\underline{S}_2$: Enter the upper and lower bound or ± % range, ± standard deviations range, x÷ range, or factor of variation of the spread.

A: Enter the assessment ratings for each applicable category (i.e., H, M, or L). Enter N/A for not applicable.

I_1: Assess the informative value based on spread. That is, assess the extent to which the entry narrows the spread of plausible values over what was known before the study that produced the entry was conducted (prior).

- L: Many prior plausible values exist in spread.
- M: Spread is a fair amount narrower than range of prior plausible values.
- H: Spread is much narrower than range of prior plausible values.

I_2: Assess the informative value based on the foreseen application for the entry. That is, how informative are the results of calculations with this entry expected to be given the current persisting (posterior) uncertainty about the entry. (I_2 here is a first guess, to be refined when the particular application is considered.)

L: The existence of other posterior plausible values (i.e., values in spread) matters for application.
M: The existence of other posterior plausible values matters marginally.
H: The existence of other posterior plausible values does not matter.

G: Assess the generalizability of the entry to other applications, locations, or sample spaces different from the application for which it was originally generated.
L: does not generalize to other applications
M: can be generalized with limitations
H: easily generalized

R: Assess the entry's robustness over time.
L: highly perishable
M: moderately perishable
H: time independent

P: Enter the pedigree ratings for the applicable categories (i.e., 1 to 5). Enter N/A for any inapplicable pedigree category.

T: Assess the theoretical basis of the entry and the tenability of the theory's application to produce the entry.
1: no theory or concepts
2: weak theory or concepts, controversial empirical support
3: weak theory, good empirical support
4: good theory, but one of competing theories
5: well-understood and accepted theory

D: Assess the quality of the data inputs used to generate the entry.
1: unacceptable
2: poor
3: fair
4: good
5: excellent

E: Assess the estimation methods used to generate the entry.
1: unacceptable
2: poor
3: fair

4: good
5: excellent

M: Assess the estimation metric (i.e., proxy or indicator for what we want to measure.)
1: unacceptable
2: poor
3: fair
4: good
5: excellent

<u>Comments:</u> Enter any comments about the NUSAP categories. For example, measures may require explanation such as "WLM stands for working level month." Statistics may need explanation, such as "mean from meta analysis." The level of spread may require explanation such as "confidence level corresponds to +/- 2 standard errors corresponding to multiplication or division by a factor of 1.7 for the upper and lower bound." The reasons why assessment ratings and pedigree rating were received can be explained here (and probably should be).

REFERENCES

Funtowicz, S. and J. Ravetz (1990). *Uncertainty and Quality in Science for Policy.* Kluwer Academic Publishers, Dordrecht, The Netherlands.

NOTES