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THE BENEFITS OF AIR AND WATER POLLUTION CONTROL: A REVIEW AND SYNTHESIS OF RECENT ESTIMATES

by

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DISCLAIMER

CHAPTER 1

INTRODUCTION

There is increasing concern about the possible overregulation of the U.S. economy. This concern focuses not only on regulation of price, quantity, and service in transportation, communications, and other supposedly non-competitive sectors of the economy, but also on regulations dealing with those problems that economists have analyzed under the heading of external diseconomies: the impacts of air and water pollution on health, recreation, productivity, and amenities; the risk to workers of accident and ill health due to occupational hazards; and so forth. This concern with overregulation manifests itself primarily in the assertion that the costs of regulation outweigh the benefits. Most of the legislation authorizing regulation in the areas of environmental quality, occupational safety and health, and product safety has not placed major emphasis on the relationship between benefits and costs as a criterion in regulatory decisionmaking; and in fact consideration of benefit-cost relationships is ruled out in some cases, for example the establishment of national primary air quality standards. Thus regulatory agencies have had little incentive, until recently, to develop comprehensive and well documented measures of benefits and costs. And they have found themselves not well prepared to deal with the current criticisms alleging overregulation.

There has been a number of efforts to estimate various aspects of the damages due to air and water pollution or the benefits that might be expected to accrue from controlling pollution discharges to air and water. Some of these studies have been based on new primary data sources, while others have attempted to draw inferences from secondary sources. Some of these studies have had good data but used poor theory, others have used good theory but have had poor data; and it is difficult. to compare the conclusions of these studies because of noncomparability in the data base, methodological framework, assumptions as to baseline conditions, etc.

The major objectives of this report are first, to provide a survey and critical review of the existing literature giving estimates of national benefits or damages; second, where possible, to place all the estimates in a common framework with respect to such factors as postulated change in pollution, year and price level, and so forth, so as to facilitate comparisons, and third, to draw upon this body of past research to obtain defensible and consistent estimates of the benefits of controlling air and water pollution. Although there is no effort to do so here, these estimates could then be compared with cost measures to inform the ongoing debate about overregulation.

It is not our objective, nor could it be given the existing state of knowledge, to determine whether the existing set of environmental regulations is optimal in the sense of equating marginal benefits and marginal costs across all control activities for all substances. We cannot hope to provide such detailed information to guide individual decisions, either at a national or regional level. Rather, here we are dealing with national aggregate measures of pollution and pollution effects; and typically the data are for aggregates of polluting substances. Finally, the best we can hope for is to provide estimates of the benefits of non-marginal changes in pollution, not accurate point estimates of marginal benefits.

We turn now to a discussion of the concept of benefits, and the analytical framework within which benefit measures can be formulated and interpreted. First, it is important to draw a distinction between the concept of <u>benefits</u> and the <u>effects</u> of environmental pollution. Pollutants can have effects on people directly or indirectly through a variety of channels. For example, pollutants can affect human health as measured by morbidity and mortality rates. Pollutants can affect individuals' activities such as water-based recreation. Or they can affect the availability of goods and services through their influence on agricultural productivity or the rates of deterioration of materials. Each of these types of effects has an economic aspect. Controlling pollution means reducing the magnitude of these effects.

The values that people place on reducing the adverse effects of pollution constitute our measure of benefits. The basis for determining these values is taken to be individuals' preferences. The benefit of an environmental improvement is the sum of the monetary values assigned to the effects of that improvement by all individuals directly or indirectly affected by that action. These monetary values can be defined in terms of the individuals' willingness to pay to obtain the effects of the environmental improvement, or in terms of the sums individuals would have to receive as compensation in order to induce them to accept voluntarily the adverse effects of pollution. As a practical matter there is likely to be little difference between the two conceptual approaches.[1] They will be assumed to be equivalent in this report.

Second, we must make clear the distinction between <u>benefits</u> and <u>damages</u>. Benefits are the value of some improvement from a "dirty" status quo, while damages represent what is lost in money terms because the emission of pollutants has degraded the environment from some natural "clean" state. Benefits and damages are mirror images of each other. The benefits of eliminating all pollutants are equal in magnitude to the damages incurred because of the existing level of pollutants. The benefit terminology is preferred because it is more applicable to policy decisions regarding potential environmental improvements. However many of the studies reviewed in this report have used the damage terminology

The benefits that stem from pollution control policy are the product of three sets of functional relationships.

1. Changes in the rates of discharge and the time and place of discharges of residuals into the environment lead to changes in various measures of ambient environmental quality.

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[1]See Willig (1976) and Freeman (1979a), especially Chapter 3, and page 128.

2. Changes in ambient environmental quality lead to changes in the flows of environmental services to individuals. These changes may in turn be reflected in changes in the way individuals use the environment.

3. Changes in environmental services lead to changes in economic welfare or benefits.

The first functional relationship is essential for linking benefit measures with costs, since costs are usually defined in terms of the impact of control activities on the rate of discharge of residuals. In this study we will not take account of this set of relationships; rather, we will take as given a postulated change in ambient environmental quality.

Although measuring benefits involves the use of economic theory and technique, the second functional relationship makes it clear that benefit estimates must be built on a foundation of other types of knowledge. For example, estimates of the health benefits from air pollution control must be based on scientific knowledge of the relationship between pollutant concentrations and human health. Estimates of recreation and fishery benefits stemming from water pollution control require knowledge of the relationship between pollutant levels and biological productivity. Approaches to determining the changes in physical effects, environmental uses, or flows of environmental services must be specific to each type of environmental pollutant and environmental use. These will be discussed as appropriate below. Lack of knowledge of these relationships may, in some instances, be the major barrier to empirical estimates or benefits.

There are three basic approaches to determining the values that individuals place on improvements in environmental quality. One is to simply ask individuals, through surveys and direct questioning. The second is to place proposals for alternative levels of improvement in environmental quality to referendum vote. Under certain circumstances, the outcome of the voting process will be consistent with, and therefore reveal information about, the underlying demand for environment improvement. The third approach involves analyzing data from market transactions in goods and services related to environmental quality. Under certain circumstances, the willingness to pay for environmental improvements can be estimated from market data about the demands for goods and services that have substitute or complementary relationships with environmental quality. All of the empirical techniques actually used in an effort to obtain quantitative estimates of benefits involve some variation on one of these basic approaches. For further elaboration and discussion of empirical techniques based on these three approaches, see Freeman (1979).

Since benefits have been defined as the dollar value of the change in environmental quality, some choice must be made about the postulated range over which environmental quality has changed. The conceptually correct measure of benefits would compare environmental quality levels <u>with</u> and <u>without</u> some specified degree of control, holding all other things equal, including the patterns of production, technology, and demand which determine the generation of pollutants. Thus one should compare an actual observed outcome resulting from the policy with a hypothetical or counterfactual position reflecting the same underlying economic conditions and only differing because of the impact of environmental policy.

A major issue today is the impact of already adopted environmental policies. To measure the benefits of existing policy one should compare the actual environmental quality in the appropriate year, say 1978, with that environmental quality which would have been experienced in 1978 in the absence of any legislation restricting or controlling pollution, all other things being equal. To implement this measure, we would have to have some way of predicting what the levels of economic activity and discharges to the environment would have been in the absence of any controlling legislation. Economic models can be developed for this purpose; but it is beyond the scope of this project to do so.[2]

An alternative measure would be to compare environmental qualities levels in, say, 1978 with actual environmental quality levels in the year in which the policy was adopted. This measure has the virtue of being based upon actual observations of environmental quality levels before and after the adoption of the policies. But in a growing economy, it is likely to lead to an underestimate of the conceptually correct or ideal measure. This is because in the absence of a control policy

[2] For an example of this type of analysis, see U.S. Council on Environmental Quality (1978), pp. 419-421, and Waddell (1978).

population growth and increases in the level of economic activity would have resulted in higher pollution levels in later years.

The two measures described above look at the actual or realized impact of environmental policies at a particular point in time. An alternative measure would look at the benefits to be realized if and when policies actually achieved stated targets such as national ambient air quality standards. Again, the conceptually correct measure would compare the hypothetical alternative of no policy with the world in which environmental quality targets, had been met other things held equal. Alternatively one could compare the actual environmental quality levels at the time of adoption of the policy with the stated targets or standards on the implicit assumption that the targets would be met immediately.

Within the resources available for this project, it will not always be possible to implement these measures. Where the data permit, we will attempt to assess the actual benefits realized in 1978 resulting from reductions in pollution since 1970 and those to be expected from the successful attainment of stated environmental targets. However in both cases this will lead to and underestimate of the conceptually correct measure since the comparison will be with pre-policy actual environmental quality levels rather than the hypothetical "no policy but other things equal" levels.

Even the most careful estimate of benefits will contain inaccuracies due to errors in measurement and statistical estimation. And it will often be necessary to make some assumptions regarding unknown parameters. These sources of error and uncertainty are especially important here given the present state of the art and knowledge about damage and benefit relationships. The preferred approach to dealing with uncertainty is to adopt the framework and language of probability theory.[3] This framework can be utilized to incorporate information on the error properties of statistical estimates and subjective probability statements in the determination of expected values or most likely estimates and confidence intervals or ranges within which the true value is thought to lie with some probability.

Some of the benefit estimates reviewed here have used a formal or informal probability framework to establish most likely values and lower and upper bounds. Others have simply stated point estimates as if they were certain. The synthesis estimates I present below will be stated in terms of a range with lower and upper bounds. I will also state a most reasonable point estimate. Neither the range nor the most reasonable point estimate will be derived from a formal probability model, but rather will be based on subjective judgments and estimates. The bases for these judgments will be described for each estimate. The most reasonable point estimate should not be cited or discussed independently of the range of upper and lower bounds, since to do so would convey an entirely false sense of accuracy or certainty about inherently inaccurate and uncertain values.

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[3] See Freeman (1979), pages 30-32.

References to Chapter 1

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CHAPTER 2
AIR POLLUTION CONTROL BENEFITS

Introduction

The Clean Air Act Amendments of 1970 established a major federal policy effort to control stationary and mobile sources of air pollution. In this section we review a number of previous efforts to estimate the benefits of air pollution control in various categories such as health and materials. Then, where possible, we adjust these estimates to a common set of assumptions to make comparisons possible and utilize these figures to estimate the range of possible benefits and the most reasonable or "best guess" value. We provide estimates of benefits to human health, reduced soiling and cleaning costs, reduced materials damage, and improved agricultural productivity and the broad category of asthetics and amenities. We also examine estimates of damages as measured by the effects of air pollution on property values.

We take as our point of reference air quality levels in 1970 and estimate the annual benefits realized by the 1978 population due to air quality improvements between that date and 1978. It must be recognized that this may be only a portion of the total benefits of the air pollution control policies implemented since 1970. The benefits due to prevention of further degradation, if known, should be added to the abatement benefits estimated here. We also attempt estimates of the damages to the 1978 population which would be experienced if 1970 air quality levels were experienced in 1978.

Health Benefits

The procedure used here for estimating human health benefits involves three steps. The first is to determine the relationship between exposures to different levels of air quality and human health as measured by mortality and morbidity rates. The second step is to use this relationship to predict the changes in mortality and morbidity associated with some specified change in air quality and exposure to pollutants. The third step is to use monetary measures of willingness to pay to assign values to the predicted changes in mortality and morbidity. We will now describe each step in more detail.

Getting better info&nation on the relationship between air quality and human health is itself a major and difficult research task.[1] One approach is to use multivariate statistical techniques to analyze data on mortality or morbidity and exposures across different population groups. As an alternative to large scale statistical studies, analysts might attempt to derive dose-response functions extrapolated from laboratory studies of animals and of clinical effects on humans.[2] Given an

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[1]See Freeman (1979b) and references therein for further discussion. This topic is also discussed in Lave and Seskin (1977) Chapters 1 and 2. For reviews of the literature on health effects of air pollution, see Lave and Seskin (1977) Appendix A, and American Lung Association (1977) and (1978).

[2]Extrapolation from animal studies has been used for estimating the effects of some toxic chemicals and food additives, but has not been used as a basis for estimating air pollution health effects. Clinical studies of humans usually show only acute health effects and at exposures higher than typical ambient levels. Thus they do not provide a basis for estimating the effects of exposure to ambient pollution concentrations.

empirically or judgmentally derived dose-response function, the next step is to predict the change in mortality or morbidity conditional upon the expected change in air quality, holding all other variables constant. The third step is to determine the monetary value to be assigned to each death avoided or day of ill health prevented.

A number of approaches to assigning monetary values to ill health or the loss of life have been proposed and/or utilized in the literature on the economics of health and safety. These approaches can be broadly categorized as determining values according either to individual preferences (willingness to pay), or to resource or opportunity costs. What follows is a brief evaluation of these approaches.[3]

what about looking to see what the gov't has revealed itself to be willing to pay?

Valuing Health. The most common approach to the valuation of premature loss of life or increased morbidity is the so-called productivity or human capital technique. This approach values each life lost at the present value of the expected stream of future earnings for that individual had that individual's premature death been avoided. Morbidity is valued according to the loss of output as measured by earnings or imputed earnings. This is a form of resource or opportunity cost approach to assigning values. It is based on the assumption that

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[3] For more extensive discussions of alternative concepts of the value of life or safety, see Schelling (1968), Mishan (1971), Acton (1973), and Jones-Lee (1976). Raiffa, Schwartz, and Weinstein (1977) present an excellent discussion of concepts and applications in the context of public sector decisionmaking.

earnings reflect the individual's marginal productivity, i-e., the individual's contribution to total economic output. With the death or illness of the individual, that output is lost.

There are three criticisms of this approach. First, the productivity measure of benefits has no necessary relationship to the theoretically correct measure, individual willingness to pay. Second, the productivity approach allows no role for the probabilistic nature of death and death avoidance in the health and safety areas or for differing individual attitudes or preferences toward risk and risk avoidance. An individual could pay no more than the present value of his or her earnings stream plus accumulated wealth to avoid certain death. But the statistical value of life based upon the person's willingness to pay for small probability changes could be several times this amount.[4] Third, the implicit judgment underlying the productivity/human capital approach is that an individual is worth what he or she does, i.e., that output is a measure of worth. According to this approach, preventing the premature death of a retired, unemployed, or disabled person would have no value. This is clearly an unacceptable ethical implication of the productivity concept.

Because of variation in patterns of earnings over the life cycle and differences in labor market experience among individuals, including discrimination, values derived from the productivity approach depend crucially on the age, sex, and race

[4] For some evidence on this point, see below.

of the individuals involved. For example, using a 6% discount rate and 1972 patterns of earnings, saving the life of a white male between 30 and 34 years of age would prevent the loss of about $180,000 in discounted lifetime earnings (in 1972 dollars) while for a nonwhite woman of the same age, the earnings loss prevented would be only about $70,000 (Cooper and Rice, 1976). Some benefit-cost analyses using productivity values have attempted to adjust for the age, race, and sex composition of the population at risk. Others have used averages for the population as a whole.

Another component of the resource or opportunity cost of illness is expenditures on medical care, i.e., doctors, hospitalization, drugs, etc. Rice (1966) and Cooper and Rice (1976) have estimated the total direct costs of illness in the United States. This total includes those expenditures associated both with temporary illness and with illnesses ending in death. Some of the studies discussed below have estimated the fraction of this total that can be attributed to air pollution induced illness.

The major alternative to the productivity measure is conceptually more attractive because it is consistent with the basic theory of welfare economics. It is to value increases in longevity or reductions in the probability of death due to accident or illness according to what affected individuals are willing to pay to achieve them. Individuals in a variety of actions act as if life expectancy were like any other economic good, that is, they are willing to trade-off life expectancy for other goods and services which they value more highly or vice versa. They make decisions which involve reductions in life expectancy or increased probability of death in return for increases in income or other goods and services, revealing thereby that they perceive themselves to be better off having made these choices. Some people accept risky or hazardous jobs because of higher wages. Some people make their trips between cities by airplane rather than by bus or train because it is quicker and more convenient, even though the chance of accidental death is greater.

As these examples make clear, the question is no~~w~~t how much would a specific individual be willing to pay to avoid certain death tomorrow. Rather, it is how much would the individual be willing to pay to achieve a small change in the probability of death during a given period. An individual's willingness to pay for changes in the probability of his death can be translated into a more convenient figure for evaluating strategies, namely, the value of <u>statistical life</u> or the value of a <u>statistical death</u> avoided. Suppose, in a group of 1000 similar individuals, each individual has a willingness to pay of, say, $1000 for a policy which would reduce the probability of his or her death, by, say, 0.01. This policy is a form of collective good for the individuals involved; The benefit to the group is found by adding across all individuals. The aggregate willingness to pay is one million dollars (1000 x $1000), and the expected number of deaths avoided is 10. The group's aggregate willingness to pay to avoid one death is then $100,000 ($1 million ÷ 10). This we refer to as the statistical value per life.

Two approaches to the empirical measurement of willingness to pay and the statistical value of life have been used in the literature. One is to observe market transactions where individuals actually purchase or sell changes in their risk levels. For example, if wage differentials among occupations are related to differences in occupational risk levels, these differences may be interpreted as reflecting, at the margin, individuals' trade-offs between certain kinds of risks and money. The other approach is to conduct surveys, asking individuals a series of questions about hypothetical situations involving safety/money trade-offs. If the questions are carefully designed, and if individuals are capable of predicting accurately how they would act if placed actually-in these hypothetical situations, their answers may reveal the money values they attach to reductions in risk.[5]

There have been three major studies using wage rates to estimate willingness to pay for risk changes. Although the first two studies were based on the same data on wages, they

[5]Neither approach captures the willingness to pay of relatives or close friends. Needleman (1976) has offered evidence that, including others' willingnesses to pay could increase the statistical value of life by 25-100%.

reached quite different results. This is apparently due to differences in the data on occupational risk. Thaler and Rosen (1976) concluded in their study that the statistical value of life lies between $273,000 and $508,000, with the best estimate being $391,300.[6] These figures are in sharp contrast to the results obtained by Robert Smith (1976), whose estimates ranged between $2.2 million and $5.1 million. Finally Viscusi (1978) found values for blue-collar workers between $1.8 million and $2.7 million using a different set of wage and risk data. His paper also included a careful discussion of the possible explanations for the differences in values cited here.

There are similar differences in the results of two efforts to use the survey approach to obtaining values for statistical life. Acton (1973) obtained 36 responses from a stratified random sample of residents in the Boston area. He asked people to state their willingness to pay for a community program of emergency coronary care which would reduce each individual's probability of death by heart attack. Two different forms of the question implied values for statistical life of $47,000 and $72,000. Jones-Lee (1976) asked a similar small sample of individuals several questions about their willingness to accept higher airplane fares to travel on lines with lower probabilities of experiencing a fatal crash. The value of statistical life implied by the respondents was about $6.1 million.

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[6] All figures in the next two paragraphs have been converted to 1978 dollars through the Consumer Price Index.

These disparities among the estimates suggest that substantially more research is required before reliable estimates of the value of life can be obtained by these techniques. But the results do suggest order-of-magnitude bounds on the appropriate figures.

The published estimates of air pollution health benefits discussed below have relied primarily on the productivity approach. One exception is Crocker, et al. (1979) who used estimates of willingness to pay derived from two of the wage differential studies cited above. In the "synthesis" estimate of health benefits to be derived below, I explicitly reject the productivity approach. I value mortality reductions according to a best estimate value based upon these willingness to pay studies. Except for Acton's study of the willingness to pay for emergency coronary care, the estimates cited above range from approximately $275,000 to $6 million. Although these figures span a range of over one order of magnitude, it seems highly likely that the "true" value for a representative individual in 1978 falls within this range. In the estimates below, I will use a value of $1 million per death avoided. However, the basis for the calculations will be provided so that the reader might easily substitute another value and compute alternative measures of benefits accordingly.

Reductions in morbidity should also be valued according to willingness to pay. However it is even more difficult to provide a firm empirical basis for the value of morbidity reduction. I will value reductions in morbidity alternatively at $20 and $40 per day of illness or restricted activity. The $40 figure is roughly the average daily earnings for those in the labor force. However not all instances of morbidity result in lost earnings to individuals. They may work at reduced capacity, or take paid sick leave; and morbidity may occur on other than work days. On the other hand, the earnings measure does not reflect pain, discomfort, and anxiety. Again, the reader may substitute other values in computing his or her own estimate of benefits.

In addition to these imputed values for reduction in mortality and days of morbidity, I will include an estimate of the savings in direct medical expenditures. An individual's willingness to pay to avoid the onset of an illness which might lead to death would, in principle, include the willingness to pay to avoid the burden of the direct expenditures on doctors, hospitalization etc. Thus adding direct expenditures to estimates of willingness to pay would seem to involve double counting. But there are two reasons why that is probably not the case here.

First, the estimates of the willingness to pay to reduce the probability of death are derived primarily from data where accidental injury is the principal cause of death.[7] In such cases, the associated medical costs are likely to be small, and the associated pain, suffering, and anxiety would be of short duration. Thus the willingness to pay to reduce the probability of

[7]The exception is Thaler and Rosen (1976) who used data on total mortality rates by occupation.

accidental death may be a substantial underestimate of the willingness to pay to avoid death by some other causes, for example cancer. In the latter case, both medical expenses and pain and suffering may be substantially greater, as would the willingness to pay to avoid them.

Second, for institutional reasons such as medical insurance, individuals do not directly bear all of the medical costs of their illnesses. These costs would not influence the individual's willingness to pay to reduce the risk of incurring these illnesses. Yet they are opportunity costs to society. For these reasons direct medical expenditures are added to willingness to pay measures in computing. overall health benefits due to reduced mortality and morbidity.

<u>Stationary Source Air Pollution - Review:</u> For purposes of policy design, it would be most helpful to have separate estimates of the benefits of controlling each of the major criteria air pollutants--suspended particulates, sulfur compounds, carbon monoxide, nitrogen oxides, ~~and~~ photochemical oxidants and lead. However we do not presently have the knowledge of dose-response functions required to provide disaggregated estimates of benefits. One reason for this is the high degree of correlation in the levels of, and exposures to, some pairs of pollutants such as suspended particulates and sulfur compounds. These two compounds are emitted primarily from stationary sources rather than from mobile sources such as the automobile. Most of the studies to be reviewed in this section have considered both particulates and sulfur compounds together. The

general conclusion is that together they have an impact on human health, but because of the high degree of correlation between them, it is not possible statistically to separate their effects using the conventional tools of regression analysis. In this section we examine efforts to estimate the benefits of controlling particulates and sulfur compounds together and take them as representing, to a first ~~approximation~~, the benefits of controlling all stationary source air pollutants.[8]

There is now a substantial number of aggregate epidemiological studies dealing with the effects of suspended particulates, sulfur compounds, or both on human health. These studies have used a variety of cross sectional and in some cases longitudinal data on mortality and/or morbidity and pollution levels for various population samples, including groupings by SMSA, city, county, and, in one case, census tracts within one urban county. We will first review those studies dealing with mortality, and the estimates of health benefits (sometimes including morbidity) which have been based upon them. Then we will turn to the much smaller number of studies examining the relationship between morbidity and air pollution.

In 1970 Lave and Seskin published the first report of their ongoing research on the health effects of air pollution. They

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[8]More than half of all nitrogen oxides emissions come from stationary sources. However the major health problem associated with nitrogen oxides is thought to stem from its role as a precursor, with hydrocarbon, of photochemical oxidants. Control policies for these substances has focused on mobile sources. Benefits of controlling these substances will be discussed in a later section.

presented a thorough review of the existing literature on health effects and reported the results of their own regression analyses of data reported in this literature. They also reported the first results of their own cross section analysis of 1960 mortality and pollution data for 114 SMSA's. This analysis was based on a simple linear regression model of the following forms:

$$M = a + bP + cS + e$$

where M is the mortality rate, P is some measure of pollution, and S is a vector of socioeconomic variables such as percent non-white, percent over 65 years of age, and percent poor.

It will be most convenient to report their results, and others to follow, in the form of elasticities. An elasticity measure gives the percentage change in mortality associated with a 1% change in a pollution variable. In linear equations, these elasticities are computed at the mean values of the independent and dependent variables. The elasticities for Lave and Seskin's own regressions on U.S. data on mortality from all causes were as follows:

Total Death Rate		
Particulates	.05	
Sulfates	.04	
Sum		.09
Infant Death Rate		
Particulates	.07	
Sulfates	.03	
Sum		.10

Neo-natal Death Rate

Particulates	.06	
Sulfates	.04	
Sum		.10

Fetal Death Rate

Particulates	.09	
Sulfate	.05	
Sum		.14

These elasticities mean, for example, that a 1% (50%) reduction in pollution would reduce total mortality by .09% (4.5%). They also presented "best guess" estimates of elasticities by causes for all pollutants combined derived from their review of the literature and re-analysis of the data:

Bronchitis Mortality and Morbidity	.5-1.0
Lung Cancer Mortality	.5
Respiratory Disease Mortality and Morbidity	.5
Cardiovascular Mortality and Morbidity	.2
Cancer Mortality	.3

Lave and Seskin based their estimate of monetary benefits on the resource or opportunity cost concept of the value of health. They utilized data by Rice (1966) on the forgone earnings due to mortality and morbidity and direct expenditures on medical care by disease category. Using their best estimates of elasticities by category, and after some unspecified minor adjustment, they estimated the benefits of reducing air pollution by 50% to be approximately $2.1 billion dollars per year in 1963 dollars. Using the consumer price index for a crude adjustment to 1978 dollars would place the figure at approximately $4.5 billion.

There are three comments to be made about this estimate. First the elasticities from which it is derived are their consensus "best guess" estimates rather than those derived from their regression analysis. Subsequent estimates by Lave and Seskin have been based entirely on their own regression equations. Second, the postulated 50% reduction in pollution is not based on any specific policy. Thus it is difficult to draw policy conclusions from this figure. Third, the dollar valuation is based on the inappropriate resource or opportunity cost approach. Using a willingness to pay approach would result in a substantially higher figure.

A subsequent report by Lave and Seskin (1973) has been the basis for four different estimates of air pollution control, each of which are discussed below. In this study Lave and Seskin expanded their sample to 117 SMSA's. All data were for 1960. In their "best" regression equation on total mortality, they controlled for the percent of the population non-white, population density, and the percent of population over age 65. They were not able to control for diet, smoking, exposure to other environmental hazards, or other socioeconomic variables. Pollution values were taken from one or at best a few monitoring stations in each urban area. They are not necessarily representative of the current exposure of any one individual or the exposure history of the population. Lave and Seskin are quite aware of the limitations of their data set and the difficulties and pitfalls in the analysis. For a full discussion, see Lave and Seskin (1977).

Lave and Seskin found their best statistical results using the arithmetic mean of suspended particulate readings over the year and the minimum bi-weekly sulfate reading in the year. Coefficients on both pollution variables were positive and highly significant. The elasticities were .053 for particulates, .037 for sulfates. The combined elasticity for both pollutants was .09.

Waddell was the first to use Lave and Seskin's results as the basis for estimating national health benefits from air pollution control. Waddell estimated that total suspended particulate levels in 1970 would have to be reduced by 26% in order to reach the national primary air quality standard. Assuming the same percentage reduction in sulfates, Waddell used the combined elasticity of .09 to predict a 2.34% reduction in total mortality and associated health costs. The average annual growth rates in private health expenditures and wage and salary incomes were used to inflate the direct and indirect health costs from Rice (1966) from 1963 to 1970 dollars. Rice did not allocate direct medical costs between morbidity and mortality. Therefore a 2.34% reduction in 1970 total health costs must be interpreted as an estimate of mortality and morbidity benefits combined. The resulting estimate was $3.73 billion per year in 1970 dollars. This estimate embodies the assumption that the relationship between direct and indirect morbidity costs and pollution levels is the same as that observed between mortality rates and pollution.

Waddell then used a separate data set from the EPA CHESS study (Environmental Protection Agency, 1974) to derive a separate estimate of morbidity caused by respiratory diseases. The CHESS data do not lend themselves to regression analysis. Waddell apparently developed subjective estimates of dose-response functions for various types of respiratory diseases, used them to predict the change in the number of cases of morbidity, and applied imputed values to estimate benefits. The estimates are not well documented in the report. On the basis of the CHESS studies, he estimated respiratory related morbidity benefits of between $.9 and $3.2 billion per year with a mid point of about $2.0 billion. These cannot be added to the Lave-Seskin based estimate. Rather Waddell deducted an estimate of direct and indirect morbidity costs due to respiratory diseases from the Lave-Seskin total, and added his own CHESS based estimate.

Finally, arguing that the Lave-Seskin regression equation applies only to an urban population exposed to high levels of air pollution while the Rice morbidity and mortality cost data applies to the U.S. population as a whole, Waddell adjusted the Lave-Seskin estimate for the percentage of the total population living in urban areas (73.5%). This gives a revised estimate for Lave-Seskin of $2.58 billion, which when added to the CHESS morbidity benefits yields a total of $4.6 billion per year as a best estimate of the health benefits of attaining the national primary standards. The range is $1.6-7.6 billion per year.

where does this range come from? did [illegible] establish it?

The main criticisms of this estimate concern the use of the CHESS data for the morbidity component of disease costs and the use of the resource or opportunity cost basis for valuing health benefits. The CHESS study itself has been subjected to a number of criticisms concerning its choice of methodology and data. And as indicated above, Waddell's use of the study to determine the incidence and value of morbidity costs is not well documented.

Kenneth Small (1977) used Lave and Seskin's 1973 regression equation and estimated elasticity to compute the benefits of a 50% reduction in air pollution, compared to the 26% reduction assumed by Waddell. There were three other principal differences between Small's and Waddell's estimates. Small did not use the CHESS data on respiratory disease morbidity. Rather he followed Lave and Seskin in assuming proportionality between mortality and morbidity costs. This makes Small's estimate smaller than Waddell's, since the CHESS costs imputed by Waddell were greater than ~~the~~ his downward adjustment to Lave and Seskin's total cost figure. Second, Small's estimate is based on 1963 dollars. That is, he did not inflate the mortality and morbidity costs of Rice to 1970 levels. Third, and most substantive, Small used a different adjustment factor to account for the urban component of total health costs as estimated by Rice.

Small argued that only the portion of the population classified as living in urbanized areas was subject to air pollution. This percentage was about 55% in 1963. Small argued that only 55% of the total cost of ill health estimated by Rice could be attributed to people living in areas affected by air pollution. However, it should be recalled that Lave and Seskin's data were drawn from SMSA's. The total SMSA population includes not only those living in urbanized areas, but also others living in non-rural areas within the geographic boundaries of the SMSA's. If the non-urbanized portion of the SMSA population is not affected by air pollution, Lave and Seskin's elasticities underestimate the impact of air pollution on the health of the urbanized portion of those populations. The appropriate counter to this bias would be to apply the Lave and Seskin elasticities to the total SMSA population, not just the urbanized portion. Thus, I conclude that Small's method of adjustment is inappropriate, and his estimate is biased downward.[9] (His figure is $4.21 billion in 1963 dollars or $7 billion in 1970 dollars.)

The final estimate based on the Lave and Seskin 1973 report port was prepared by Heintz, Hershaft, and Horak (1976) for the Environmental Protection Agency. They started with Waddell's

[9]There is another factor which should in principle be taken into account in adjusting for the urban component of the population. That is the greater incidence of ill health and associated health costs in urban areas compared to rural areas, i.e., the so-called "urban factor". If mortality and morbidity rates are greater in urban areas, for reasons other than--or in addition to--air pollution, then using the SMSA population to compute SMSA health costs will lead to an underestimate. And since this figure is the basis for calculating health benefits, benefit estimates would be biased downward as well.

estimate in 1970 dollars and used a detailed adjustment to Rice's 1963 total health costs to make them applicable to the 1973 population, prices and incomes. These adjustments yielded an estimate of $5.7 billion(in 1973 dollars)for the benefits of a 26% improvement in particulants and sulfur compounds.

In 1977 Lave and Seskin published the capstone of their long-term research effort (Lave and Seskin, 1977). They used mulivariate regression techniques to investigate the relationships between a variety of air quality indicators and mortality rates. They estimated cross-sectional, time series, and combination cross-section-time series models. They investigated the impact of pollution levels on total mortality rates and mortality rates disaggregated by age, sex, race, and disease specific rates. They examined a variety of functional forms and model specifications, and undertook an extensive search for threshold effects. Air quality variables were chosen to represent long-term or chronic exposures for typical residents of urban areas. Air quality indicators for each SMSA were used along with other possible explanatory variables in regression equations to explain mortality rates by SMSA. In general the results of their analysis support the hypothesis that higher exposure to air pollutants leads to higher mortality rates. Moreover, they found no evidence for thresholds for sulfates, particulates, or sulfur dioxide. And they concluded that the linear model gives the best fit to the data.[10]

Table 1 provides a summary of their major results in the form of elasticities calculated at the means of the relevant linear regression equations. The estimates of elasticities across different data sets, model specifications, and degrees of disaggregation are substantially similar. These elasticities provided the basis for Lave and Seskin's estimate of the health benefits of controlling air pollution in urban areas.

To be conservative, Lave and Seskin chose the lowest elasticity measure they obtained, that for their original 1960 equation for unadjusted mortality rates. They cited an EPA estimate that successful implementation of the Clean Air Act Amendments of 1970 would reduce sulfur oxide emissions by 88% and particulate emissions by 58% from 1971 levels by 1979. They assumed that this would result in an equivalent percentage reduction in air pollution levels. In other words they used a "rollback" model. Using separate elasticities for sulfates and particulates (not reported in Table 1) they predicted a 7% reduction in mortality rates by 1979.

To assign a monetary value to this reduction in mortality, they utilized more recent estimates of mortality and morbidity costs developed by Cooper and Rice (1976). These data were for

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[10] However, for critical reviews of this body of work, see Landau (1978), Viren (1978), and Thibodeau, Reed, and Bishop (forthcoming). See also Lave and Seskin's response to Thibodeau, Reed, and Bishop (forthcoming).

Table 1

Estimates of the Elasticity of Mortality with Respect to Air Pollution--From Lave and Seskin

		Combined Elasticity.	
Model	Air Pollutants (Combined)	Unadjusted Mortality Rate	Age-sex-race adjusted Mortality Rate
1960 Annual Cross-Section-- 117 SMSA's	Sulfates and particulates	.094	.096
1969 Annual Cross-Section--112 SMSA's	Sulfates and particulates	.116	.100
1969 Annual Cross-Section--69 SMSA's	Sulfates and particulates	.106	.096
1969 Annual Cross-Section--69 SMSA's	Sulfur dioxide and particulates	.126	.110
1960-69 Annual Cross-sectional time-series--- 26 SMSA's	Sulfates and particulates	.094	.102
1962-68 Annual Cross-sectional time-series-- 15 SMSA's	Sulfates and particulates	.118	.126
1962-68 Annual Cross-sectional time-series-- 15 SMSA's	Sulfur dioxide and particulates	.106	.114
1963-64 Daily time-series-- Chicago	Sulfur dioxide	.108	

Source: Lave and Seskin (1977) Table 10.1, p. 218.

1972; so Lave and Seskin adjusted them to apply to a 1979 population with its associated real incomes and relative prices of medical care. But the estimates were presented in 1973 dollars.[11] On the assumption that the reduction in air pollution has the same percentage impact on morbidity and mortality, they estimated the reduction in total health costs (i.e. benefits) of an 88% reduction in sulfur oxide pollution along with a 58% reduction in particulates to be $16.1 billion at the 1973 price level.

This estimate of benefits is substantially higher than any of the previously cited estimates. This difference cannot be attributed to differences in the underlying relationship between pollution and health. The elasticities used in this estimate and the earlier ones by Waddell and others are essentially equal. There are three reasons for the differences. First Lave and Seskin's estimate applies to a larger 1979 population with its associated higher real income levels and relative cost of medical care. Second, Lave and Seskin postulated a substantially larger reduction in pollution levels than that utilized by any previous study. Third, unlike Waddell, they did not include an adjustment to the national health costs to exclude costs associated with the non-urban population. Rather, they assumed that control of emissions nationwide would also yield equal percentage decreases in pollution levels in rural areas. Since only about 73% of the national population live in SMSA's, the bene-

[11] See Lave and Seskin (1977), page 225 and pages 348-349 for details of the adjustments.

fits to the urban population are about $11.8 billion per year. This is still substantially higher than any of the earlier estimates.

As part of their review of the Lave and Seskin work, Thibodeau, Reed, and Bishop (forthcoming) replicated ~~their~~ Lave and Seskin's data base and basic cross section equations for total mortality in 1960. They then examined the sensitivity of the results to the treatment Of six SMSA's which were outliers in the sense that one or more of their explanatory variables lay quite outside the range of the remaining SMSA's. When the outliers were deleted there were significant changes in the coefficients of the pollution variables. They concluded that their evidence supports the existence of a positive relationship between air pollution (as measured by particulates and sulfates) and mortality, but that estimates of its magnitude are quite sensitive to model specification.

Viren (1978) was more strongly critical of the Lave-Seskin conclusions. He offered a variety of data from other studies and from his own analysis of the 1960 air pollution and mortality data to support his contention that the association between pollution and mortality may well be spurious. For example, he cited studies that show a geographic pattern of cigarette consumption which results in a positive association between smoking and some air pollutant variables, cited the well-established connection between smoking and mortality, and argued that air pollution could be a proxy for smoking. He also cited evidence of strong regional gradients in mortality and some measures of air pollution and argued that pollution may be a proxy for the true determinants of mortality differences. He also showed that adding variables to the basic Lave-Seskin regressions would some times reduce one or both of the pollution variables to statistical insignificance.

Viren's results show both the presence and consequence of multicollinearity among the variables chosen for various regressions. This is one of his major points. However he did not offer compelling theoretical justifications for some of the variables he chose to include. Nor did he utilize any of the available statistical techniques for coping with multicollinearity. Thus although his study amply illustrates the difficulties and pitfalls in analyzing data in this area and suggests caution in interpreting positive results such as those of Lave and Seskin, it does not disprove the hypothesis that air pollution causes mortality.

The remaining two major efforts to estimate monetary measures of health benefits broke new ground in two respects. First they provided separate estimates of mortality and morbidity; and second, they both eschewed the simple reduced form linear model in favor of more complicated models and estimation techniques. Both studies found smaller mortality effects than did those studies based on the work of Lave and Seskin. One (Liu and Yu, 1976) also found smaller morbidity benefits. However the other study (Crocker, et al., 1979) found much larger morbidity benefits.

In fact they are by far the most significant form of health effect according to their results.

Liu and Yu based their analysis of mortality on a sample of forty SMSA's which had sulfur dioxide readings in excess of 25 $\mu g/m^3$ (micrograms per cubic meter) between 1968 and 1970. To cope with anticipated multicollinearity they utilized a forward step-wise regression model with the following structure. First they estimated:

$$(1) \qquad M_i = a + bS_i + cW_i + u_i$$

where M_i is the observed mortality rate in the ith SMSA, S_i is a vector of socio-economic variables (percentage of population over 65, percent of population with incomes above the poverty level, percent of population white, and percent of population over 25 years of age with four years of college), W_i is a vector of weather variables, and u_i is the error term. They then regressed the residual from (1) on pollution in a non-linear specification:

$$(2) \qquad M_i - \overline{M}_i = e^{(d - f / P_i)}$$

where ${}_i$ is the estimated mortality rate as computed from equation (1) and

P_i is a pollution measure.[12] The coefficient on sulfur dioxide in (2) was positive and statistically significant. Liu and Yu also estimated a linear second stage equation with sulfur dioxide. In this form, the sulfur dioxide coefficient was negative but not significantly different from zero. Finally Liu and Yu estimated a separate second stage equation with suspended particulates as the independent variable. The particulate variable was not statistically significant.

I have three comments on the Liu and Yu procedure and results. First the estimation of separate equations for sulfur dioxide and particulates is inappropriate, especially since the two equations were then used to compute damages for the two pollutants. To the extent that the two pollutants are positively correlated, the coefficients in the single variable equations will be biased upward--each capturing some of the effect of the omitted variable.[13] The preferred approach is to include all relevant pollution variables in the second equation.

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[12]Liu and Yu also estimated a third stage equation which they called a "generalized average damage function." They claimed it could be used to estimate damages or benefits for SMSA's outside of their sample. They formed a dependent variable consisting of the sum of the computed mortality rate from equation (1) and the <u>computed</u> residual given by the estimated form of equation (2). This was regressed on the full set of socio-economic, weather, and pollution variables. This model has been dismissed as being virtually tautological by Smith (1977).

[13]Most air pollution data do display a correlation between sulfur compounds and particulate levels. See for example, Crocker, et al. (1979), p. 45, and Lave and Seskin (1977) p. 32. In fact it is puzzling that Liu and Yu report a partial correlation coefficient between the two pollutants of only .04.

Second, the forward stepwise regression procedure is known to bias the coefficients of the second stage equation toward zero if the variables in the two equations are correlated with each other. Thus the lack of significance of the pollution variable, except for the non-linear sulfur dioxide equation, could be attributable to the estimating procedure.

Third, Lave and Seskin also used the forward stepwise procedure, but used a linear form for the second stage and included both sulfur compound and particulate variables. Both pollution variables were positive and significant, and similar in magnitude to the coefficients in their simple linear model.[14]

These differences in the results between Lave and Seskin and Liu and Yu are puzzling. Thirty-three of the forty SMSA's in Liu and Yu's sample also appear in the sample of 117 SMSA's used in Lave and Seskin's analysis of 1960 data. Thirty-four of the forty also appear in Lave and Seskin's 1969 data set. The two studies used similar sets of socio-economic variables. Lave and Seskin also included population and population density, Liu and Yu included three climate or weather variables in their basic stage (1) equation. Lave and Seskin did not include such variables in their initial analyses reported in Chapters 3 and 4. However, they did devote a chapter to a study of the impact of adding additional climate variables. They found that in general the association between pollution and mortality was maintained. Because of the larger sample size, replication of the

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[14] See Lave and Seskin (1979) pages 50-52.

results with different data sets, and robustness of the results through various model specifications and functional forms, it seems prudent to place more confidence on the results of Lave and Seskin.

After estimating a pollution-mortality relationship, Liu and Yu employed a version of the productivity approach to assign monetary values to the increase in mortality associated with air pollution. They computed a discounted present value of earnings stream for a representative member of the labor force between the ages of 18 and 64. The earnings stream was adjusted upward to account for the expected increase in productivity over time. Also their approach to defining the opportunity cost of forgone earnings is less precise than that employed by Rice (1966) and Cooper and Rice (1976) in that it does not take into account the cross-sectional variation in earnings with age nor the differential impact of pollution induced mortality across the age structure of the population.

Finally, Liu and Yu used their estimated equations and imputed values to calculate the reduction in lost earnings associated with reducing sulfur dioxide from the levels observed between 1968 and 1970 down to a maximum of 25 $\mu g/m^3$ for each SMSA. This disaggregated approach is to be preferred to the aggregated elasticity approach used in the studies described earlier. The benefits of reducing sulfur dioxide to the assumed threshold of 25 $\mu g/m^3$ for the 40 SMSA's in the sample are $.9 billion per year in 1970 dollars. Liu and Yu used a similar calculation for mortality benefits due to particulate control, even though the particulate variable was not significant in their stage 2 regression. The total particulate control benefits for the 40 SMSA's is $1.0 billion per year. The total for particulates and sulfur compounds combined is $1.9 billion. It should be noted that this estimate does not include the reduction in direct expenditures on health care, nor benefits due to reduced morbidity. This estimate is not directly comparable with any of the other benefit estimates cited here. This is because Liu and Yu have postulated much larger decreases in pollution levels than any of the other studies and estimated benefits for only 40 SMSA's.

Crocker _et al_. (1979) is an interim report on an ongoing research effort being conducted at the University of Wyoming. This study is significant in that it was the first effort to model the impact of individual behavior and choice on epidemiological relationships. They argued that there may be reasons to expect that variables which are exogenous in some structural equations are in fact endogenous to the overall system being modeled. Empirical work on the health effects of environmental pollutants has generally ignored the possibility of such simultaneous equation relationships and their effects on statistical estimation. When single equations containing endogenous variables on the right-hand side are lifted out of the simultaneous system and estimated with ordinary least squares, estimated coefficients may be biased. Where complex processes with simultaneous relationships are involved, the most important of them must be modeled with a set of structural equations reflecting these interrelationships. Then appropriate statistical techniques such as two stage least squares can be employed. Attention must also be given to the identifiability of the relationships to be estimated.

The following example illustrates the potential for simultaneous relationships of the health effects model and the potential problems in estimation. First, suppose that some measure of morbidity (M) for a population is a function of exposure to an environmental pollutant (P), access to medical care (D) and other variables. Second, suppose that access to medical care depends upon average income (Y). Finally, income itself may depend upon the health status of the population. For example, higher morbidity means more days lost from work and lower earnings. Specifically:

$$(3) \qquad M = a_0 + a_1 D + a_2 P + u_1$$

$$(4) \qquad D = b_0 + b_1 Y + u_2.$$

$$(5) \qquad Y = c_0 + c_1 M + u_3$$

where u_1, u_2, and u_3 are error terms and the coefficients are hypothesized to have the following signs:

$$a_0,\ a_2,\ b_1,\ c_0 > 0$$

$$a_1,\ c_1 < 0$$

Morbidity, doctors and income are all endogenous to the system even though they each appear as an independent variable in one structural equation. If (3) were estimated by ordinary least squares, the parameter estimates would be inconsistent. Furthermore, in this specific case, (3) cannot be identified.

In their analysis of mortality, Crocker _et al_. focused on the simultaneous determination of mortality and the availability of medical care as measured by doctors per capita. They did not present an explicit statement of the structure of the underlying model. But the structure that is implied by their specification of estimating equations is of the following form:

$$M = a_0 + a_1D + a_2P + a_3S + a_4B + a_5F + a_6E + u_1 \tag{6}$$

$$D_d = b_0 + b_1Y + b_2P + b_3S + b_4B + b_5F + b_6E + u_2 \tag{7}$$

$$D_d = D_s = D \tag{8}$$

where (8) implies that the supply of doctors always is adjusted to equal demand, and where S represents a vector of socioeconomic variables such as the age structure of the population, racial composition, and educational attainment; B represents a vector of behavioral variables such as smoking habits and exercise; F represents a vector of dietary variables; and E represents a vector of other environmental variables such as radiation exposure and climate.

For the first stage, they estimated a reduced form equation for doctors which included all of the exogenous variables except pollution. For the second stage, they computed a "doctors" variable from the estimated reduced form equation and used it in estimating equation (6), first excluding the pollution variables, and then including pollution for purposes of comparison. The pollution variables included were annual averages for nitrogen dioxide, sulfur dioxide, and suspended particulates. When equation (6) was estimated for total mortality, only two of the three pollution variables had positive signs, and none even approached conventional levels of statistical significance. The computed elasticity for the three pollution variables combined is about an order of magnitude smaller than those reported by Lave and Seskin (1977). They also re-estimated (6) for nine separate disease specific mortality rates, including those for heart disease, vascular disease, cirrhosis, and cancer. They found statistically significant pollution variables in two of the nine disease specific mortality rate equations. Suspended particulates were significant in the equation for pneumonia and influenza; and sulfur dioxide was significant in the equation for early infant diseases.

The elasticities of the disease specific mortality rates with respect to the significant pollution variables were .09 for early infant disease/sulfur dioxide, and .39 for pneumonia/particulates.

Crocker _et al_. computed benefits for reduced mortality due to pneumonia and early infant deaths by postulating a 60% reduction in pollution in urban areas. They assumed a value of life of $1 million in 1978 dollars. The urban population at risk was estimated to be 150 million. Applying the estimated elasticities to the disease specific mortality rates led to a benefit estimate of $15.9 billion per year. 15 This estimate of benefits is very close to that provided by Lave and Seskin (1977). But this is primarily because of two offsetting effects, a much higher imputed value of life combined with a much lower estimated impact of pollution reduction on mortality.

Because of the substantial differences between Lave and Seskin and Crocker et al. in their underlying epidemiological relationships, this aspect of these two studies deserves careful scrutiny. In what follows, I will focus first on differences in model structure and the simultaneous equation model presented by Crocker et al., and second on differences in their choice of variables to include in the mortality equation.

Lave and Seskin and others chose to work with a simple (usually linear) model in which all variables except mortality were taken to be exogenous. The simultaneous equation approach chosen by Crocker et al. is in principle superior, provided that there are in fact significant interdependencies among the variables, and that these interdependencies are correctly modeled. One major form of interdependency is that between the adverse effects of exposure and individual actions undertaken to reduce or avoid exposure. These could include changes in residence or

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15 Using a lower value of life derived from the work of Thaler and Rosen (1976) of $340,000 in 1978 dollars led to a benefit estimate of $5.1 billion per year.

occupation, migration from high pollution to low pollution areas, and a variety of other types of averting behavior.[16] Crocker _et al_. did not choose to investigate this set of interdependencies. If they turn out to be important in practice, then for this reason alone, the Crocker _et al_. model would be misspecified.

Rather Crocker _et al_. focused on the role of medical care in mediating the effect of pollution on mortality. This may also be important, but their model does not appear to be capable of disentangling the separate influences of the availability of medical care and the exposure to pollution on mortality.

In the equations they chose to report, they did not include any measures of pollution in the first stage reduced form equation explaining doctors per capita. They eventually reported that they included pollution variables in a separate set of estimates and stated that for the mortality equations, "the results are consistent for the effect of medical care and for the positive associations between sulfur oxides and infant diseases and for particulates and pneumonia (1979, p. 67)". They also reported a significant _negative_ association between air pollution and doctors in the first stage reduced form equation. They said:

> "It appears that doctors may choose _not_ to live in polluted cities (perhaps for aesthetic reasons). If this is the case, one can easily explain false positive

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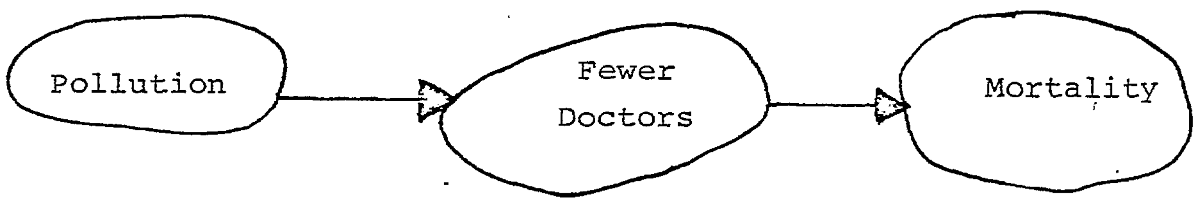
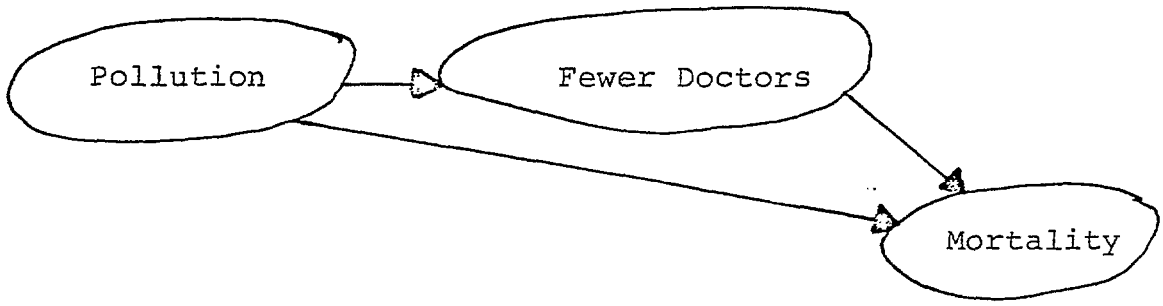
[16] See Zeckhauser and Fisher (1376).

associations between air pollution and mortality where medical care is excluded as an explanatory variable. If doctors avoid polluted cities, and if doctors do reduce mortality rates, then pollution could well be associated with higher mortality rates; but not because of any direct health effect of air pollution or mortality (1979, p. 68)."

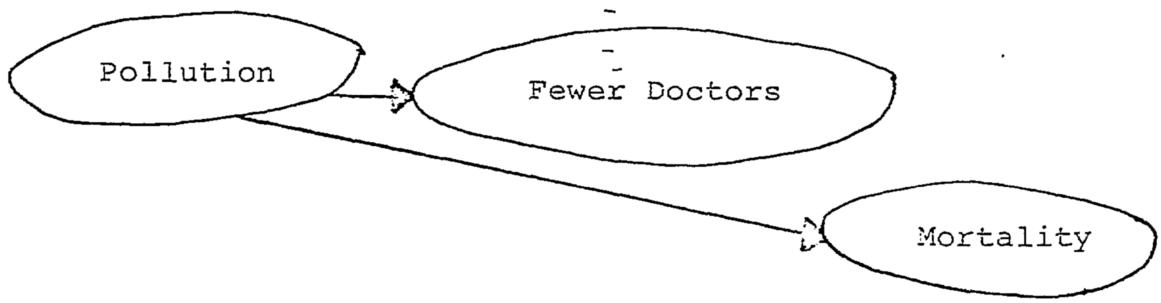
In schematic terms, Crocker _et al_. are proposing the following causal relationship:



But the reported evidence is consistent with either of the following causal models:



or



In the latter model, the availability of doctors does not influence mortality, a result that Crocker _et al_. say is consistent with the existing epidemiological literature (1979, p. 25).

There is another question concerning their modeling of the availability of doctors. Equation (5) above is implicitly a demand function for doctors. The observed availability of doctors may be the result of some equilibrating process between demand and supply functions. Crocker et al. suggest a supply mechanism in which pollution enters negatively in comparison with a possible positive effect of pollution on the demand for doctors. But the supply side has not been explicitly modeled in deriving the reduced form equations. This aspect of the model has not been given sufficient attention.

There are several differences in the data used in the regression between Crocker _et al_. and Lave and Seskin which might help to explain the differences in the results. First, Lave and Seskin used SMSA's as the unit of observation while Crocker et al. used cities. Both choices lead to problems because of the non-homogeneity of populations within the units, spatial variation in actual air pollution levels within the unit, etc. It is not clear that either should be preferred on _a priori_ grounds. But it is conceivable that the differences in results could arise because of differences in the choice of unit of observation. More experimentation with alternative combinations of units of observation and model specification would be desirable.

There is a major difference in the treatment of the age characteristics of the population. Lave and Seskin used percent over 65 in the total mortality and the disease, sex, and race

Another possibly important difference is that Crocker _et al_. attempted to control for diet while Lave and Seskin did not include dietary variables. Controlling for dietary factors (as well as other environmental health hazards such as contamination of drinking water, occupational exposures to toxic substances, or ionizing radiation) is desirable in principle. But as in the case of smoking, it is difficult to find data on population aggregates which adequately reflect the idiosyncracies of individual behavior. Crocker _et al_. constructed variables for consumption of protein, carbohydrates, and saturated fatty acids for each city on the basis of a U.S. Department of Agriculture survey of food consumption by income level in four regions of the country. Each city's consumption was a weighted average of the consumption by income class in that region, where the weights represented the income distribution of the city.

Protein is significant and positive in the equations for total mortality, cancer, and emphysema/bronchitis; and animal fat is significant in heart disease. This illustrates the importance of trying to control for dietary factors. But one must question whether the variables used by Crocker, _et al_. reasonably reflect differences in diet across cities or differences in income distribution.

All in all, the results of the Crocker _et al_. study should make one more cautious about basing conclusions about the air pollution-mortality relationship on the results of Lave and Seskin alone. However there are enough questions about the model structure and choice of variables employed by Crocker _et al_. to suggest that one should not reject Lave and Seskin and accept the Crocker _et al_. results yet. Rather, the questions identified here should provide a basis for further investigation. In fact, both sets of authors are engaged in additional research at this writing. And it is to be hoped that a clearer picture will emerge shortly.

All of the studies discussed so far have used dose-response information on the mortality/air pollution relationship to derive monetary measures of benefits. Several studies have had the more limited objective of obtaining information only on the dose-response function. These studies may be useful as a check on the magnitude and accuracy of benefit estimates. In one such study, Schwing and McDonald (1976) investigated mortality among white males in 46 SMSA's in the years 1959-61. The pollution measures were: hydrocarbon potential, derived from vehicle miles traveled on the assumption that emissions and pollution levels were proportional to vehicle miles; sulfur dioxide potential, derived in a similar manner from estimates of stationary source emissions; annual averages of nitrogen dioxides in 1969; and sulfate and nitrate measures from 1965. The "potential" measures of pollution may be quite poor because of photo chemical transformations and dispersion. Also they did not use a measure of total suspended particulates. Schwing and MacDonald controlled for various climate and socioeconomic variables and used a proxy for smoking derived from cigarette tax data. They employed ordinary least squares, along with ridge regression and sign constrained least squares to cope with multicolinearity problems.

One set of regressions investigated disease specific mortality rates for fifteen categories of disease. The results were somewhat mixed but with a tendency for sulfur compound variables to show positive elasticities. A second set of age specific total mortality rated equations showed a number of positive elasticities for sulfur compound measures. Some representative elasticities are:

	by ridge regression	by constrained least squares
White male lung cancer and sulfur compounds	.016	.041
White male arteriosclerotic heart disease and sulfur compounds	.005	.027
White male total mortality and sulfur compounds	.022	.045

Estimates of elasticities for nitrogen compound varied substantially in sign and magnitude across equations. And the results for hydrocarbons were generally inconclusive. The elasticities for sulfur compounds are the same order of magnitude but somewhat smaller than those reported by Lave and Seskin for sulfur and particulate compounds combined. Thus while there are reservations about this study (the quality of the smoking and air pollution measures) the results are not inconsistent with those of Lave-Seskin.[17]

In a recently published study, Mendelsohn and Orcutt (1979) combined data on illness related death rates, pollution levels, climate, and socio-economic variables all classified by county group. The mortality rates were for illness only. Pollution measures were for 1970 and included sulfates, nitrates, total particulates, carbon monoxide, sulfur dioxide, nitrogen dioxide, and ozone.

Results were reported only for white males and white females. Total mortality and age specific mortality rates were computed. Sulfates had positive and significant coefficients for all adult whites. Sulfur dioxide and carbon monoxide also had positive coefficients for most adult age and sex groups. However ozone was typically negative and significant in over half the specifications for total mortality for white males and females. It was also negative but only occasionally significant in age specific equations. Other pollutants had both positive and negative signs and were generally not statistically significant. Because of incomplete reporting of the data, I was not able to compute any weighted average aggregate elasticities. However the sum of elasticities for sulfur dioxide and

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[17] In Jackson et al. (1976) the Schwing-McDonald results provided the basis for an estimate of national air pollution control benefits for controlling sulfur, nitrogen compounds and hydrocarbons. The age specific regression equations for white males were assumed to apply to the total urban population. These equations were used to determine the reduction in mortality for the total abatement of pollution. This involves extrapolation beyond the range of the data used in the regression equations and assumes background levels are zero. Rice's data on cost of illness were used to compute benefits of $12.5 billion per year for total abatement of sulfur compounds and particulates in 1968 dollars.

sulfates in the adult male and female age categories ranged from approximately .10 ~~and~~ to .20.

The results for sulfates and sulfur oxides tend to support a conclusion that there are significant health benefits for controlling sulfur emissions. However, the results also provide some support for a conclusion that ozone reduces mortality. Viren (1978) argued that both results may be due to a spurious association of pollution measures with a strong west to east gradient in regional mortality rates. But this leaves unanswered the question of what factors, environmental or otherwise, are responsible for the higher mortality rates in the eastern part of the country.

Lipfert (1979a) used cross section data on annual average levels of sulfur dioxide, total suspended particulates, suspended sulfates, benzo-(a)pyrene, iron, and manganese for over 150 U.S. cities to examine the relationship between air pollution and age specific mortality rates. Regression equations included percent over 65 years, percent non-white, percent in poverty, birth rate, and variables reflecting age of housing stock, smoking, education, and population. In a stepwise regression procedure sulfur dioxide was excluded as not adding to the explanatory power of the equations. Particulates were significantly positive in the equation for total mortality and for some of the age specific rates. Generally the effect of particulates increased with age. The elasticity for particulates in the total mortality equation was .071. Sulfate particles were significant and positive in one equation but significant and negative in several others. In a second paper Lipfert (1979b) used the same explanatory variables to examine disease specific mortality rates. Sulfur compound and particulate variables were generally insignificant.

Finally Gregor (1977) examined the variation in sulfur dioxide, suspended particulates and mortality across census tracts in Allegheny County which includes Pittsburgh. Control variables included income, race, education, a measure of smoking, access to medical care, population density and climate variables. The particulate variable was significant, but sulfur dioxide was not. Particulate elasticities in various age specific equations and for different groupings of cause of death (all causes, pollution related, non-pollution related) ranged from .2 to .89. These are much higher than those found in other studies.

In the last study to be reviewed, Finklea <u>et al</u>. (1977) used best judgment estimates of adverse health effects associated with sulfur oxides to estimate the benefits of meeting ambient air quality standards while economic growth and fossil fuel combustion to produce electricity continued over the period 1975 to 1980. They did not provide monetary estimates of benefits, but only estimates of the reduction in premature death and various forms of morbidity. Their estimates of reduced mortality, etc. are comparable in magnitude to those obtained by other researchers using regression analysis.

We turn now to morbidity and consider the results of five separate studies. The first is by Liu and Yu (1976). Their estimate of morbidity benefits was derived from the CHESS study (Environmental Protection Agency, 1974). As indicated above, the CHESS data are not in a form amenable to regression analysis. However, Liu and Yu employed an imaginative technique to try to overcome this difficulty. The CHESS study covered four geographic areas. In each, sulfur dioxide and suspended particulate levels were recorded for each of several sub-areas categorized as low, intermediate, or high pollution areas. Liu and Yu regressed the bronchitis prevalence rate against pollution readings. Separate regressions were run for sulfur dioxide and particulates. There were no other control variables in either set of equations. The number of observations for each equation ranged from 3 to 5. Significant coefficients for a pollution variable were found in two of the eight regressions.

All eight regression equations were then used to generate a larger sample of pollution and morbidity data through Monte Carlo techniques. This sample was used to regress morbidity on sulfur dioxide and particulates in separate non-linear regression equations. Liu and Yu then applied their own estimates of direct (physicians, hospitalization, drugs, etc.) and indirect (lost earnings) morbidity costs. Applying these imputed cost figures to the reduction in morbidity predicted by their Monte Carlo regression technique produced estimates of morbidity benefits of $99 million for sulfur dioxide and $141 million for particulates.[18]

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[18] There are errors in the tabular presentation of the data. The figures here include corrections provided by Liu.

The major weakness with the Monte Carlo technique is the weak empirical foundation on which the Monte Carlo data generator is based. Liu and Yu do not investigate the errors of the estimates of the means and standard deviations employed in the Monte Carlo mechanism to generate the sample data for their regressions. Thus one cannot estimate the impact of these initial errors on the error properties of the dose-response functions used to generate the benefit estimates.

Crocker et al. used a University of Michigan longitudinal survey of approximately 5,000 households to estimate the morbidity effects of air pollution. The survey consisted of annual interviews from 1968 through 1976. The survey questions applied to the work and health experience of the head of household. The data include socioeconomic variables, work experience, and place of residence. Thus it is possible to determine residential air pollution levels for a substantial part of the survey group and to control for migration and its effect on the exposure histories of the members of the sample.

The research consisted of two stages. In the first, the authors sought to determine the association between measures of acute and chronic health effects and annual averages of nitrogen dioxide, sulfur dioxide, and total suspended particulates in the counties in which the sample resided. Acute illness was measured by number of work days lost as reported in each annual interview.

In nine different linear regressions on seven samples drawn randomly from the total survey group, a pollution measure was statistically significant with a positive sign in explaining acute illness. Elasticities computed at the mean ranged from about .3 to .6. Partitioned samples were also drawn. Pollution was significantly positive for those households always living in one state and for those with real incomes equal to or less than $7,500, but not significant for households with heavy cigarette consumption or chronically disabled households.

Chronic illness was defined in terms of disability, that is, some limitation on ability to work. The variable analyzed was the length of the disability period. This variable was scaled as follows: Equal to or less than two years = 1; 2-4 years = 2; 5-7 years = 3; 8 years or more = 4; and otherwise = 0. There are two questions to be raised about this measure of chronic illness. First there are many forms of disability which are entirely unrelated to air pollution, for example those due to war injury, occupational or other forms of accidental injury, or congenital defects. Second, it is not clear from biomedical considerations why the length of the disability, or more precisely, the interval from the onset of the disability to the time of the survey should be a function of air pollution levels at the time of the administration of the survey. These questions about the definition and measure of chronic illness take on considerable significance below when it is seen that according to Crocker, *et al.*, chronic illness accounts for about 95% of the total economic impact of the morbidity associated with stationary source air pollution.

In the unpartitioned samples, a pollution variable was significant and positive in four of the seven equations to explain length of disability. Pollution was significantly positive in all four of the partitioned samples. Reported elasticities ranged from .25 to .95.[19]

In the second stage of analysis, Crocker _et al_. utilized a recursive model to assess the impact of air pollution on both acute and chronic illness, and through them on hours worked and wages. This model had the following structure:

$$\text{Length of Disability} \equiv L = f(P, D, S, B, M, E)$$

$$\text{Acute Illness} \equiv A = g(L, P, S, B, M, E)$$

$$\text{Wage} \equiv W = h(L, D, A, S, B, C)$$

$$\text{Hours Worked} \equiv H = k(W, L, A, F)$$

where M represents access to medical care, D is severity of disability, C represents a vector of human capital and related variables influencing productivity, and F represents other conventional determinants of labor supply.

The value of reduced morbidity due to air pollution control is based on a resource-opportunity cost concept measured by forgone earnings, but it does not include direct medical expenses. Benefits are defined as:

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[19] These figures are offered for illustrative purposes only. True elasticities cannot be computed because of the open-ended class interval for the dependent variable.

$$\text{Benefit} = -\frac{\Delta\ \text{income}}{\Delta\ P} = -\left[\frac{\Delta H}{\Delta P} \cdot W + \frac{\Delta W}{\Delta P}\ H\right]$$

They calculated the per household benefit for a one $\mu g/m^3$ reduction in pollution as measured by suspended particulates to be $4.80 per year in 1978 prices or $288 per year for a 60% reduction in particulate pollution. Remembering that the $288 benefit figure was derived from the responses from heads of households and was not necessarily a per capita benefit figure, there are several ways in which this number can be used in estimating national aggregate benefits for controlling air pollution induced morbidity. At one extreme, Crocker *et al*. assumed that every member of the urban population of the United States is or will be a head of household and therefore experiences (on average) the same annual benefit. Given this, a 60% reduction of pollution (as measured by particulates) results in annual benefits of between $28.4 billion and $58.1 billion with a most reasonable point estimate of $43.2 billion. To provide a more conservative estimate, Crocker *et al*. assumed that the earnings of working wives are 60% of their husbands', and that there are additional benefits due to improved productivity in household activities. They computed benefits to the 1970 urban population of $25 billion per year.[20] The lower and upper bounds are $16 billion and $34 billion. Finally as an alternative to these assumptions, one

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[20] For details see Crocker *et al*. (1979), p. 160.

could assume that the individual figure applied only to heads of households in urban areas and that no other members of the household experienced pollution induced morbidity losses. Then if 68% of the 63.4 million households in 1970 lived in urban areas, annual benefits would be $12.4 billion.

These estimates, if valid, suggest that air pollution induced morbidity may be more significant in welfare terms than mortality. This is an important conclusion. But there are several problems in the study which should make one cautious about placing great weight on the precise figure. We have already mentioned the large role played by the length of disability in the benefit measures and the questions about the use of this as a measure of the prevalence of chronic illness. The second issue is possible biases in the sub-sample selected for each regression run. For both acute and chronic illness, only those households residing in counties with air pollution data were included. This probably biases the sample toward urban and more-highly polluted areas. The sample for the chronic illness equations was more restricted to include only those who had lived within the same state throughout their lives.

In several instances, Crocker _et al_. have used integer values rather than dummy variables to represent different possibilities. For example, for education attainment, grades 0 to 5 are assigned a value of 1, grades 6-8 a value of 2, etc. In one case (degree of disability) the integer values assigned do not correspond to a monotonic ranking. And last, the study did not control for occupational exposures to toxic substances.

Two morbidity studies limited their attention to hospitalization rates. Carpenter, et al. utilized data on hospital admissions in Allegheny County, Pennsylvania and sulfur dioxide and particulate levels at the patient's residence. They found significant associations between pollution levels and respiratory and circulatory system diseases. It is not possible to compute the elasticity from the data reported. Hospital costs per day were used as a measure of the benefits of achieving air quality standards. Benefits were estimated to be $9.8 million per year in 1972--or $6.13 per capita. This is an underestimate of the total cost of pollution related morbidity. In contrast to these results, in a similar study of Portland, Oregon, Bhagia and Stoevener (1978) did not find a significant association between daily suspended particulate levels and utilization of in-patient medical services by already hospitalized patients. Both of the studies can be criticized for not treating the decision to seek hospitalization as in part an economic one which would be influenced by price and income.

The last study to be discussed involved best judgment estimates of dose-response relationships rather than empirical estimation. Goldstein contributed a thorough review of the literature on the health effects of sulfur compounds to the National Academy of Sciences study of the control of stationary source air pollution (National Academy of Sciences, 1975). To conclude his review, Goldstein offered subjective, best judgment estimates of the percentages of total observed health effects in different categories which could be attributed to sulfur oxides pollution.[21] He attributed the following percentages of disease prevalence or incidence in various categories to the presence of sulfur oxides: chronic bronchitis prevalence - 10%; acute morbidity during the course of chronic respiratory disease - 10%; upper respiratory infections - 1%; asthma attacks - 5%. These estimates imply an elasticity of about .05 for this type of morbidity.

It may difficult to discern a pattern from this lengthy and detailed review and evaluation of existing estimates of the health benefits from stationary source air pollution control. Table 2 organizes and summarizes the information reviewed above on eight different studies. The first five columns of the table present information that may make it possible to identify the major sources of differences among the estimates. The next to the last column presents the aggregate benefit figure as stated in the original study. These figures are not directly comparable because the studies utilized different bases for imputing values, assumed different percentage reductions in pollution, and they are for different years with different prices, incomes, etc. The last column represents an effort to correct for these latter two sources of differences. The Consumer Price Index has been used

[21] See National Academy of Sciences (1975), pp. 144-149.

Table 2

Summary of Estimates of Health Benefits from Stationary Source Air Pollution Control

Source	Basis of Response Function	Implied or Computed Elasticity	Basis for Values	% Reduction in Pollution	Year	Benefits (Billion)	Adjusted to 1978 and 20% Reduction in Pollution [a]
Lave & Seskin (1970)	Subjective Judgment based on literature review and own regressions.	.09	Resource/ Opportunity Cost-Rice (1966)	50%	1963	$2.1	$1.8 b
Waddell (1974)	Lave-Seskin (1973) regressions on 1960/ 61 data by SMSA CHESS-EPA (1974) for respiratory morbidity.	.09	Resource/ Opportunity Cost-Rice (1966) adjusted	26%	1970	$4.6[b]	$5.9 b
Small (1977)	Lave-Seskin (1973)	.09	Resource/ Opportunity Cost-Rice (1966)	50%	1963	$4.2	$3.6
Heintz, Hershaft, & Horak (1976)	(Same as Waddell)	.09	(Same as Waddell)	26%	1973	$5.7	$6.4
Lave & Seskin (1977)	Lave-Seskin (1977) $. Regression on 1960/61 data by SMSA	.09	Resource/ Opportunity Cost-Cooper & Rice (1976)	88% for Sulfur compounds 58% for particulates	1973	$16.1	$4.7[c]
Liu & Yu	Liu & Yu (1976) two stage regressions by SMSA	--	Resource/ Opportunity Cost	d	1970	$ 2.2[e]	$3.7[f]

Table 2 (continued)

Source	Basis of Response Function	Implied or Computed Elasticity	Basis for Values	% Reduction in Pollution	Year	Benefits (Billion)	Adjusted to 1978 and 20% Reduction in Pollution[a]
Crocker, et.al.1979	Mortality Crocker et al (1974) regressions by city	--	Willingness to pay $1 million per life	60%	1978	$15.9	$5.3
Crocker, et al 1979	Morbidity Longitudinal household survey	--	Productivity changes in wages & work hours	60%	1978	$43.2 ($14.5)[g]	$14.4 ($ 4.8)[g]
							TOTAL $19.7 ($10.1)[g]
Finklea, et al (1977)	Subjective judgements	--	Willingness to pay[h]	i	1977	5.1[h]	$ 5.5[h]

Notes to Table 2:

a) Original estimates were inflated to 1978 by the Consumer Price Index. For comparison purposes, estimates were then adjusted to an assumed 20% reduction in pollution by assuming proportionality or constant elasticity.

b) Of this total $2.0 billion is attributed to morbidity and $2.6 billion is due to mortality.

c) Reduced by .2/[(.88 + .58)/2] for assumed change in pollution and by .73 to cover only urban population.

d) Varies for each city in sample - in order to achieve assume threshold levels.

e) Morbidity and mortality combined - for 40 SMSA's.

f) Not adjusted for assumed reduction in pollution. See also note e.

g) Crocker et al. assumed that the benefit per head of household of $289 applied to every member of the urban population (assume to be 150 million). The figure in parentheses is a lower bound obtained by applying the per head of household benefit only to the heads of each of the 68% of all households who lived in urban areas in 1977 (the latest year for which data are available).

h) Finklea et al. only estimated changes in mortality, days or cases of acute illnesses and cases of chronic respiratory symptoms. I valued these at: $ 1 million per death avoided
$ 20 per day or case of acute illness avoided
$100 per case of chronic respiratory symptoms avoided.

i) Not specified.

as a crude means of adjusting all estimates to 1978 dollars. Also on the assumption that the elasticity of the damage function is constant over the relevant range, a proportionality factor has been used to compute the benefits (in 1978 dollars) of a 20% reduction in pollution. The major source of difference remaining among the figures in the right hand column are differences in the dose-response function and in the basis of imputing values.

With the exception of Crocker et al. (1979), the adjusted figures lie within the range of $1.8-6.4 billion per year, with the median estimate being about $5.1 billion. The very high morbidity benefits estimated by Crocker et al. bring their adjusted total for morbidity and mortality combined to $19.7 billion per year, or $10.1 billion based on a very conservative assumption about the number of individuals affected. These figures will serve to place in perspective the best judgment estimates presented in the next section.

Stationary Source Air Pollution--Synthesis: In this section I draw upon the information and analyses already summarized and reviewed here to derive a synthesis or "best judgment" estimate of the benefits of air pollution control. This will be an estimate of the benefits actually realized in 1978 by the U.S. urban population due to the reduction in air pollution from 1970 levels associated with the Clean Air Act Amendments of 1970. In order to calculate these benefits, we need three major pieces of information: the reduction in air pollution levels actually experienced, the resulting decrease in morbidity and mortality, and the values to be assigned to avoiding illness and premature death. I take each of these points up in turn.

As discussed earlier, the conceptually correct basis for estimating the benefits of an air pollution control policy is to compare actual pollution levels in the year of interest with a prediction of what pollution levels would have been in that year in the absence of policy. Given the resources available for this project, it is not possible to develop such a prediction. Rather, I will base the estimate on the actual reduction in levels of suspended particulates and sulfur dioxide between 1970 and 1978. Table 3 shows several measures of air quality for the years 1970-76 for suspended particulates and 1972-77 for sulfur dioxide. The table presents both composite national averages and the 90th percentiles of the distributions across sampling sites. The last column of the table shows that by various measures, suspended particulate pollution has been reduced by from 12 to almost 17 percent between 1970 and 1976 and sulfur dioxide pollution has been reduced by between 16 and 23 percent between 1972 and 1977. On a national average basis, pollution levels are within the national primary air quality standards for suspended particulates and sulfur dioxide. But as the 90th percentile rows show, some parts of the country do experience suspended particulate levels above the national primary standards of 260 $\mu g/m^3$ for 24-hour readings and 75 $\mu g/m^3$ for the annual average.

These aggregate data give only a crude picture of changes in the degree of exposure of the American population to air pollu-

Table 3

Trends in National Suspended Particulates and Sulfur Dioxide Concentrations and Emissions

I. Total suspended particuiates (2350 sampling sites)

A. Annual Mean	1970	1971	1972	1973	1974	1975	1976	1977	% Reduction
1. Composite Average (μg/m³)	70.4	69.6	67.1	65.4	62.5	60.8	61.8	61	-13.4
2. 90th Percentile (μg/m³)	106	104	100	96	92	88	90	86	-18.9
B. Peak Daily Reading -24 hr. maximum)									
1. Composite Average (μg/m³)	234	230.1	2lL6	209.1	198.8	l9Q3	200.4		-14.4
2. 90th Percentile (μg/m³)	390	380	345	345	320	310	325		-16.7

II. Sulfur Dioxide (722 Sampling sites)

A. Annual Mean	1972	1973	1974	1975	1976	1977	% Reduction
1. Composite Average h&m3)	23.0	21.4	20.7	20.4	19.7	19.4	-15.7
2. 90th Percentile (μg/m³)	52	48	44	42	40	40	-23.0

Source: Environmental Protection Agency (1977) and Environmental Protection Agency (1978).

tants over time. But as a first approximation, it seems reasonable to assume that the urban population of the United States experienced a reduction in suspended particulate and sulfur dioxide pollution of approximately 20% between 1970 and 1978. There is some corroboration for this assumption in the report from the Environmental Protection Agency that between 1972 and 1977 there was a 29% reduction in the number of people who were exposed to annual mean suspended particulate levels in excess of the national primary standard. (Environmental Protection Agency, 1978, p. 2-1.) However if the health effects of concern here are due primarily to exposure to sulfate particles, this assumption will lead to a substantial overestimate of realized benefits. This is because there has been virtually no decline in sulfate levels over the past ten years.

It is interesting to compare this assumption about air quality improvements with those used in other studies:

Lave and Seskin (1970)	50%
Waddell (1974)	26%
Lave and Seskin (1977)	88% and 58%
Crocker et al. (1979)	60%

The Waddell figure was based on an estimate of the required reduction in pollution levels to assure that the primary standard for suspended particulates was met. The first Lave and Seskin figure and that of Crocker _et al_. are essentially arbitrary. And the second Lave-Seskin figure is based on an EPA estimate of the reduction in pollution emissions expected from the implementation

Of stationary source control requirements of the Clean Air Act Amendments of 1970. They assumed that ambient pollution levels would decrease proportionately with emissions. However the relationship between national emissions and national average air quality levels is not one of simple proportionality. While particulate levels were being reduced by between 12 and 17 percent between 1970 and 1977, particulate emissions fell by 44%; and the 15-23% reduction in amount sulfur dioxide levels between 1972 and 1977 was accompanied by a reduction in national emissions of sulfur compounds of only 8% (Environmental Protection Agency 1978, p. 5-1).

It would also be useful to know the expected benefits of further improvement in air quality, for example to full attainment of national primary or national secondary ambient air quality standards. However, the aggregate approach being employed here is not suitable for making such estimates. This approach implies a uniform percentage reduction in pollution levels experienced by all of the affected population. This is surely not precisely true but it is more reasonable as a representation of the experience between 1970 and 1978 than it would be of the results of moving to obtain compliance in the present nonattainment areas. This is because where national primary standards are not now being met, the appropriate strategy would be selective controls in these areas. This would mean that only that portion of the population presently exposed to violations of the air quality standards would experience improved air quality. EPA estimates that in 1977, only

27% of the metropolitan population of the United States was exposed to suspended particulate levels above the national primary standard (Environmental Protection Agency, 1978, p. 2-1).

Given the postulated 20% reduction in pollution levels, the next step is to use a dose-response relationship to predict the change in mortality. Several of the analyses of the health effects of stationary source air pollution have reported their results in elasticity form. This dimensionless coefficient is a useful device for simplifying computations. I will assume an elasticity and use it to predict the percentage change in mortality associated with the postulated percentage change in pollution.

This approach does have several limitations. First, it assumes that the elasticity coefficient computed for the mean values of the sample applies to all members of the relevant population. Actually, if the dose-response relationship is linear (and most of the research reviewed here used a linear specification), the point values of the elasticity vary with the position on the dose-response curve. Population units with higher pollution levels will have higher point values for their elasticities. Second, this approach assumes that all of the population units experience the same percentage reduction in pollution. This ignores variation around the mean values for pollution and other factors influencing mortality. The most dirty urban areas in the United States have probably experienced larger percentage reductions in pollution. It is certainly true that they must experience larger percentage reductions to meet existing standards.

Third, this approach assumes that the elasticity is constant over the relevant range; but the actual point value of the elasticity varies with movements along a linear function.

A more accurate approach to computing the change in mortality would be to utilize directly the dose-response function underlying the elasticity estimate to compute the change in mortality for each urban area with its own specified change in pollution levels, conditional on the values for all other control variables for that particular urban area. This is the approach utilized by Liu and Yu (1976). However neither the resources available for this study nor the published data permit us to utilize this approach.

What is our best estimate of the elasticity of mortality with respect to stationary source air pollution? Lave and Seskin's results suggest an elasticity of .1. On the other hand, Crocker et al. found a computed elasticity with respect to total mortality of only .01, and this was not significantly different from zero. Lipfert (1979a and 1979b) and Schwing and McDonald (1976) also found elasticities smaller than those of Lave and Seskin but above those of Crocker et al. It seems likely that these values will bracket the true elasticity. Therefore, I will assume a most reasonable point estimate of .05 with a low-high range of .01-.10.

Recalling that stationary source air Pollution is primarily an urban problem and that most of the epidemiological studies on which our estimates are based used urban units (SMSA's) as units of observation, the expected reduction in U.S. urban mortality can be computed. These computations are detailed in Table 4. It is expected that the experienced reductions in sulfur oxide and particulate pollution have reduced annual mortality by between 2,780 and 27,800 per year with a most reasonable point estimate of 13,900 per year. Mortality reductions for the nation as a whole may be somewhat greater to the extent that rural areas also experience air quality improvements. Ignoring possible rural mortality effects lends a conservative bias to the estimates of benefits.

In order to compute a monetary measure of benefits, it is necessary to specify some value for death avoided. As indicated above, I choose the value of $1 million per death avoided as being representative of those individuals' willingness to pay for reduced mortality based upon observations of individual behavior with respect to risk of death. This value of statistical life is substantially higher than the values based on productivity which were utilized in most of the studies summarized in Table 2. This difference in the value of life helps to account for the higher benefit figures estimated here. Using this value leads to an estimate of mortality related benefits of between $2.78-27.8 billion per year with a most reasonable point estimate of $13.9 billion per year. The reader is free to apply alternative values of life in computing alternative benefit measures. An illustrative calculation based upon a value of life of $500,000 is included in Table 4.

Table 4

Mortality Benefits for Stationary Source Air Pollution Control

ASSUMPTIONS:	Reduction in Pollution =	20%
	Elasticity of mortality with respect to pollution (most reasonable point estimate = .05) =	.01-.1
CONDITIONS:	U.S. Mortality in 1978	1,900,000
	Percent of U.S. Population living in metropolitan areas	73% in 1976
COMPUTATIONS:	U.S. Urban Mortality	1,390,000
	20% reduction in pollution reduces urban mortality by (most reasonable point estimate = 1%)	.2%-2%
	Mortality avoided (most reasonable point estimate = 13,900)	2,780-27,800 deaths
BENEFITS:		
	A. At $1 million per death avoided (most reasonable point estimate = $13.9 billion)	$2.8-$27.8 billion/yr.
	B. For illustrative purposes, if the assumed value of life is $500,000, benefits are (most reasonable point estimate = $7.0 billion)	$1.4-13.9 billion

The calculation of the benefits of reduced morbidity can be carried out in a similar fashion. Again the key questions are the choice of an elasticity of morbidity with respect to pollution and the choice of a basis for valuing morbidity reductions. The morbidity studies reviewed here can provide some guidance as to the appropriate elasticity. Lave and Seskin assumed that the morbidity elasticity was the same as that estimated for mortality, about .09. Goldstein's (National Academy of Sciences, 1975) best estimates of the percentages of specific diseases and symptoms attributable to sulfur oxides imply an elasticity of around .05 for pollution induced morbidity. Crocker _et al_. estimated elasticities for acute morbidity ranging from .3 to .6 with an average of over .4. I think that it is unlikely that the true elasticity would be as high as the highest elasticity estimated by Crocker, _et al_. from one partitioned subsample. Rather, I take the average of their elasticities, about .4, to be an upper bound estimate of the true value. Somewhat arbitrarily, I take .01 to be the lower bound, the same as for mortality. The most reasonable point estimate is .1.

There are two approaches to estimating the value of changes in morbidity. The first, employed by Lave and Seskin (1977), is to compute the proportionate reduction in the direct and indirect costs of morbidity as estimated by Cooper and Rice (1976). Cooper and Rice estimated the direct costs, that. is expenditures on doctors, drugs, hospitalization, etc. to be $75,231 million in 1972. This figure was inflated by the population growth and growth in the medical care cost component of the Consumer Price Index between 1972 and 1978. The component of this cost accruing to the urban population (73% of the total) is $95,161 million. Indirect morbidity costs are measured by lost productivity and wages. Cooper and Rice estimated this to be $42,323 million in 1972. This figure is adjusted to account for the sex and age composition of the labor force and to include imputations for household productivity and an imputed value for the institutionalized component of a population. This figure was inflated to 1978 by a factor representing the growth in average gross weekly earnings in the private non-agriculture sector. The urban component of indirect costs was $48,021 million in 1978.

If the morbidity elasticity lies between .01 and .4, a 20% reduction in pollution will reduce morbidity costs by between .2 and 8% (with a most reasonable point estimate of 2%). This would lead to benefits as follows:

	Low	High
Direct Costs	$190 million	$ 7613 million
Indirect Costs	96 million	3842 million
Total	$284 million	$11455 million

Morbidity benefits are estimated to lie in the range of $.3-11.5 billion per year. (The most reasonable point estimate is $2.9 billion per year.)

An alternative approach to computing values involves a direct estimate of the reduction in some physical measure of morbidity and the application of some imputed value per day of morbidity avoided. To this figure should be added some measure of direct medical cost such as that estimated by Cooper and Rice. Table 5 provides details of these calculations. Morbidity can be measured either by work days lost or restricted activity days. The work days lost measure applies only to people in the labor force. Restricted activity days applies to all people of all ages. And it includes degrees of illness and incapacitation that are not severe enough to result in absence from work. Given data on total morbidity and urban morbidity (by either measure), the reduction in morbidity expected to accompany a 20% reduction in air pollution can be calculated by applying the assumed elasticity.

Work days lost and restricted activity days were alternatively valued at $20 per day and $40 per day. The latter figure represents average gross daily earnings in the total private non-agricultural sector of the economy. Thus it is an approximation of lost productivity (but without adjustments for household services, etc.). The $20 per day figure is used to take account of the fact that many restricted activity days are not severe enough to result in a full loss of earnings. In any event, as Table 5 shows, the results are relatively insensitive to alternative approaches to valuation. This is because in this approach to estimating benefits the major component of morbidity

Table 5

Most Likely Values Under Alternative
Assumptions of Morbidity Benefits

I. WORKDAYS LOST		
A. Workdays lost due to acute illness in 1977		314.8 million days
B. Workdays lost to the urban population (73% of A)		229.8 million days
C. Reduction in lost work days with 20% reduction in pollution (elasticity of .1) (2% of B)		4.6 million days
1. Valued at $20 per day	$ 92 million	
plus reduction in direct medical costs (2% of $95,161 million)	$1,900 million	
	TOTAL	$1.99 billion
2. Valued at $40 per day[a]	$ 184 million	
plus reduction in direct medical costs (2% of $95,161 million)	$1,900 million	
	TOTAL	$2.08 billion
II. RESTRICTED ACTIVITY DAY		Should this be 1596?
A. Restricted activity days due to acute illness		1,196 million days
B. Restricted activity days for the urban population (73% of A)		1,457 million days
C. Reduction in restricted activity days with 20% reduction in pollution (elasticity of .1) (2% of B)		29 million days
1. Valued at $20 per day	$ 580 million	
plus reduction in direct medical costs (2% of $95,161 million)	$1,900 million	
	TOTAL	$2.48 billion
2. Valued at $40 per day	$1,160 million	
plus reduction in direct medical costs (2% of $95,161 million)	$1,900 million	
	TOTAL	$3.06 billion

Source: U.S. Department of Health, Education and Welfare (1978).

Notes: [a]This approximates average gross daily earnings in the private non-agricultural sector in 1978.

benefits is reduced medical expenditures.

Finally,combining morbidity and mortality, we estimate the total health benefits associated with reducing sulfur oxide and particulate pollution levels to those of 1978 to lie within the range of $3.1 billion to $39.3 billion per year. The most reasonable point estimate is $16.8 billion per year. As the range indicates, there is a good deal of uncertainty about both the epidemiological basis for this estimate and the imputed values for health and mortality reduction. Over 80% of the benefits estimated here are attributed to the reduction in mortality. However it should be noted that the direct health expenditure component of morbidity benefits includes expenditures associated with chronic morbidity, including morbidity ending in pollution related death.

Mobile Source Air Pollution: Those pollutants coming primarily from mobile sources and for which ambient air quality standards have been set are carbon monoxide, nitrogen dioxide, and photochemical oxidants. In comparison with stationary source pollutants, there have been substantially fewer published estimates of the benefits of controlling mobile source pollutants. And the weight of the evidence suggests that the magnitude of such benefits is small.

Babcock and Nagda (1973) estimated total damages due to mobile source air pollutants to be $2.3 billion in 1968 dollars. But this was not derived from an independent estimate of the effects of mobile source pollutants. Rather it was derived by extrapolation from an earlier estimate of the health benefits associated with suspended particulates and sulfates. Babcock and Nagda took as their starting point a preliminary estimate of health damages developed by Barrett and Waddell (1973). This in turn was derived from the first study of Lave and Seskin (1970) discussed above and on a crude approach to inflating health costs from the 1963 Rice data. Babcock and Nagda estimated "severity factors" for the five major criteria pollutants based on the national primary ambient air quality standards and their implied threshold or no-effect levels. They then applied these severity factors to estimates of the emissions of nitrogen oxides, hydrocarbons, and carbon monoxide to impute additional dollar damages to these substances. The imputed damages due to these mobile source pollutants was about 35% of the basic health damages attributed to particulates and sulfur compounds.

This imputation is based on the assumption that the national primary ambient air control standards accurately reflect the relative health effects of various substances and that the Lave-Seskin mortality equation captures <u>only</u> health effects due to sulfur compounds and particulates so that the additional imputations are fully additive.

Small (1977) also used severity factors to impute damages to mobile source air pollutants. The basis for his imputation was the 1973 mortality study by Lave and Seskin. However Small chose to assume that the Lave-Seskin estimate reflected damages due to all forms of pollutants, not just suspended particulates and sulfur compounds. Small also used a revised set of severity factors to reallocate the Lave-Seskin total among the five criteria pollutants. However since neither Babcock and Nagda nor Small base their estimates on an independent measure of mobile source pollution levels or assessment of their separate effects, they will not play a further role in the analysis here.

Schwing and McDonald's (1976) epidemiological study of 46 SMSA's included observations on ambient nitrate levels for the year 1965, and a measure of potential hydrocarbon pollution based on gasoline consumption. Given the complex photochemistry and transport processes involved in the transformation of hydrocarbons into photochemical oxidants, the hydrocarbon potential variable is not a good proxy for ambient concentrations. However Schwing and McDonald did find a positive association between some age adjusted mortality rates for white males and their hydrocarbon and nitrate variables. Using Rice's data on cost of illness and assuming zero background levels, they estimated the benefits of total abatement to be $1.3 billion per year for nitrogen compounds and $1.6 billion per year for oxidants.

Lave and Seskin (1977) included an investigation of the role of nitrates, nitrogen dioxide, and nitric oxide (NO) in their comprehensive study of air pollution and human health. They conclude:

> "The results across SMSA's indicated that nitrates were unimportant in explaining the variation in either the total mortality rates (unadjusted and adjusted) or race

(adjusted)mortality rates for infants under one year of age. There was however some indication of an association between levels of nitrogen dioxide and the variation in total (unadjusted and adjusted) mortality rates. No association was exhibited between nitrogen dioxide levels and infant mortality.

"In our cross sectional time series analysis, involving 1.5 SMSA's over the period 1962-68 (see Chapter 8), we failed to isolate a consistent significant association between either nitrates or nitrogen dioxide and either the total mortality rates (unadjusted and adjusted) or the race (adjusted) mortality rate for infants under one year of age.

"Regarding analysis of daily mortality (See Chapter 9): although we found an association between levels of nitric oxide (NO) and daily mortality in Chicago, the relationship between sulfur dioxide and daily mortality was even more significant. We found no strong relationships between daily mortality and the observed levels of other mobile source air pollutants (including carbon monoxide, nitrogen dioxide, and hydrocarbons) in any of the cities examined (Lave and Seskin, 1977, pp. 221-222)."

On the other hand, Crocker, _et al_., did find some evidence of an association between illness and nitrogen dioxide levels in their longitudinal morbidity study. NO_x was significant in some subsamples for both acute and chronic illness. When NO_x was significant, elasticities were generally in the range of .3-.6, with one exceeding 1.0.

Negative or inconclusive results have generally characterized efforts to find an association between photochemical oxidants and human health effects. Seskin (1979) examined data on unscheduled visits to a group health practice and photo chemical oxidant levels in Washington, D.C. There were some significant positive associations. But with the exception of the results for unscheduled visits to the ophthalmology department, results were inconclusive. While acknowledging the weak empirical basis for extrapolation, Seskin estimated benefits for the Washington metropolitan area that would be associated with the roughly 45-55% reduction in oxidant levels required to meet the old national primary standard (.08 ppm.) in Washington. Seskin estimated direct and indirect medical costs per visit and suggested metropolitan area benefits of roughly $90,000 (in 1973 dollars). Just to put this figure in perspective, if the same per capita benefit figure applied to all of the urban population of the United States, national benefits would not exceed $5 million per year.

The next two estimates to be reviewed were based upon dose-response functions derived from the judgment of experts. Aherne's

(1973) primary purpose was to establish a framework for the analysis of data. The accuracy of the data was apparently of a secondary concern. Aherne employed a panel of three experts to provide estimates of the incidence of chronic and acute health conditions at different levels of carbon monoxide and oxidant concentrations. The three experts displayed sharp differences in their estimates of effects.[22]

Aherne used data on the frequency distribution of carbon monoxide and oxidant readings, relationships between emissions and ambient levels, and the spatial distribution of pollution and population in urban areas to estimate the hours of exposure at various emission levels. This was combined with dose-response information and weighting factors to compute estimates of equivalent days of restricted activity. Finally these were combined with assumed willingnesses to pay to avoid days of restricted activity to compute damages associated with 1967 levels of emissions and benefits associated with 50% and 75% emissions reductions. Because of the variety of alternative assumptions which were built into the final estimates, of total damages, they range over two orders of magnitude from a low of $120 million to $20.75 billion per year. Because of the imprecision of these estimates, and because of the lack of a firm empirical basis in epidemiological data, this study should best be considered as a demonstration of the feasibility of developing a consistent

[22] See Aherne (1973), p. 194.

framework for data analysis; but the specific figures cannot be considered very accurate.

A similar but less elaborate approach was taken by Gillette (1975). The dose-response function was derived from a panel of medical and air pollution experts compiled by the California Air Resources Board. Frequency distribution data were used to estimate the population at risk for different patterns of ambient concentrations of photochemical oxidants. Assumed values for different degrees of discomfort or disability were used to compute damages associated with oxidant levels above an assumed threshold of 100 $\mu g/m^3$. Gillette estimated damages due to oxidant levels in 1973 were $183 million per year. He also estimated the benefits associated with a 20% reduction in ambient oxidant levels. The estimated benefits were $131 million per-year.

Finally, the National Academy of Sciences (1974) estimated health damages associated with mobile source air pollution to be between $360 million and $3 billion per year. There were two sources for this estimate. The first was a study of self-reported symptoms of eye discomfort, chest discomfort, cough, and headache conducted with a panel of student nurses in Los Angeles. On the assumption that the student nurses were representative of the urban population of the United States, the data on incidence of the symptoms at various oxidant levels were used to compute the incidence of symptoms for the U.S. population given 1973 ambient levels of oxidants. The total oxidant related person days of symptoms were estimated to be 195 million for the U.S. as a whole. Total person days due to all symptoms were estimated to be 26.8 billion.[23] If avoiding a symptom day is valued at between $1 and $10, oxidant related health damages are estimated at between $200 million and $2 billion per year.

The National Academy of Sciences (1974) also added an estimate of the benefits of reduced mortality due to nitrogen dioxide pollution. This is derived from an extrapolation from data presented in a preliminary, unpublished version of Lave and Seskin (1977). The National Academy of Sciences estimated that the elasticity of mortality with respect to nitrogen dioxide is .025.[24] Then based on alternative assumptions concerning the population at risk and the reduction in nitrogen dioxide levels, they estimated a reduction in mortality of between 800 and 4,400 per year. But there is a puzzling aspect of this estimate. The published version of Lave and Seskin (1977) states that there was not a significant association between nitrogen dioxide and mortality in the cross section time series analysis utilized by the National Academy of Sciences.[25] And the cited elasticity estimate cannot be found in any of the published regression equations.

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[23]This seems high compared with the estimate of less than 2 billion restricted activity days reported in the Vital and Health Statistics (Department of Health, Education and Welfare, 1978).

[24]See National Academy of Sciences (1974), p. 356-358. In a personal communication Seskin informed me that although the printed percentage reduction in mortality is incorrect (a typographical error), the predicted reductions in mortality are correct, given the data.

[25]See Lave and Seskin (1977), Chapter 8, and especially page 185.

Hence we must discount this aspect of the National Academy of Sciences' estimate.

In a study that did not deal directly with the epidemiology of mobile source air pollution, Brookshire, et al. (1979) asked households in the Los Angeles area their willingness to pay for a thirty percent reduction in air pollution. An analysis of responses suggests that about 70% of the total willingness to pay is attributable to a desire to avoid perceived health effects. This amounts to almost .5 billion per year in 1978 for the South Coast Air Basin of Los Angeles.[26]

After this review of existing estimates of mobile source air pollution benefits, we find little hard data to support a hypothesis of significant mortality benefits. And there is virtually no sound epidemiological data on a national level on which to base any firm estimate of morbidity benefits. Morbidity benefits on a national level might be as high as $2 billion per year; but this is an upper bound estimate. I take zero to be the lower bound and $1.0 billion, the midpoint, to be the most reasonable point estimate.

What data we have found refers to the potential benefits to be associated with significant reductions in ambient concentrations of mobile source pollutants. It is difficult to make a case that any significant national benefits have been realized by actual reductions in mobile source pollution between 1970 and

[26] For further discussion of this Brookshire, et al., see the sections of this report on Aesthetics and Visibility and Property Value studies.

1978. Environmental Protection Agency reports of trends in emissions and ambient levels of mobile source pollutants show little or no downward movement except for carbon monoxide (Environmental Protection Agency (1978). And we have not found any empirical basis for attributing significant health benefits to reduced carbon monoxide levels.[27]

On the other hand the Council on Environmental Quality (1978, pp. 19-25) has reported that in a sample of 15 monitoring stations in 13 cities across the country, the number of days for which violations of the national oxidant standard occurred fell by about 20% between 1973 and 1976. The number of severe violations (over 0.20 ppm.) fell by 6%; and there was a 22% reduction in less severe violations.

Based on this Council on Environmental Quality report, and assuming a linear dose damage function, realized morbidity benefits could lie between $0.0 and $0.4 per year in 1978 dollars. The most reasonable point estimate is $0.2 billion.

It should be noted that in the absence of the Clean Air Act Amendments of 1970, ambient levels of oxidants, carbon monoxide, and nitrogen dioxide might have risen significantly over 1970 levels due to increases in vehicle utilization. It is possible that there might be evidence of more serious health effects at

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[27] This is not to say that health effects of carbon monoxide pollution may not be significant for sensitive portions of the population. As yet there has not been sufficient quantification of these effects or development of data on the population at risk to provide a basis for estimating carbon monoxide benefits.

higher ambient levels of mobile source pollutants. Therefore one cannot rule out the possibility that the Clean Air Act Amendments have yielded significant benefits in preventing a rise in mobile source pollution levels and associated health effects. It is not possible to quantify those benefits here, both because of the difficulty in estimating potential vehicle use and emissions in the absence of the Clean Air Act Amendments and because of the lack of data on the dose-response functions for these three pollutants in the relevant range.

Soiling and Cleaning

In this section we review three estimates of the national benefits to households due to decreased soiling and cleaning costs associated with the abatement of suspended particulates. The first, by Jackson *et al*. (1976), is an extrapolation to 65 SMSA's from an earlier survey by Michelson and Tourin (1966).[28] Michelson and Tourin examined differences in cleaning frequencies for households in Stuebenville, Ohio and Uniontown, Pennsylvania. They identified an additional $84 per capita in cleaning costs for households in the higher pollution town. The respective suspended particulate levels in the two towns were 235 $\mu g/m^3$ and 115 $\mu g/m^3$, annual average, both well above the national primary ambient air quality standard. Jackson *et al*. reduced this figure to $42 per capita to account for the lower particulate levels typical of urban areas and applied this to an

[28] For a review and discussion of the Michelson-Tourin work, see Waddell (1974) PAGES 109-111.

unspecified sample of 65 SMSA's. They estimated additional cleaning costs for this group of $2.077 billion per year, apparently in 1967 dollars. This sample covers only about a third of the urban population. Adjusting for this and for increases in the consumer price index produces a revised estimate for the total urban population of $12.2 billion in 1978.

There are three qualifications that must be attached to this estimate. First the single comparison of cleaning costs in two cities is a weak empirical foundation on which to build an estimate of national benefits. Second, the extrapolation technique is crude at best. In particular, the extrapolation does not involve a precise specification of the change in pollution levels over which benefits are presumably being measured. And third, as will be discussed in more detail below, changes in cleaning costs are not the most appropriate measure of benefits. Reliance on changes in cleaning costs may lead to a substantial underestimate of the benefits to households of reduced soiling due to particulate pollution.

The second study, by Liu and Yu (1976), utilized a survey of the cleaning practices of households in Philadelphia conducted by Booz, Allen, and Hamilton (1970).[29] Liu and Yu applied a Monte Carlo technique to the survey data to create a sample of data pairs, cleaning frequency and total suspended particulate levels, for each of nine household cleaning tasks. They then regressed cleaning frequency on total suspended particulate

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[29] See Waddell (1974), pp. 112-115 for a discussion of this survey.

levels without controlling for other social and economic factors. Each regression took the form:

$$F = a + bP$$

where F is cleaning frequency and P is the annual average particulate level at the household location.

Unit costs for each cleaning task were used to compute the change in total cleaning costs due to the predicted change in cleaning frequency as the level of suspended particulates was reduced from its observed value to 45 $\mu g/m^3$. Predictions were made for each of 168 SMSA's based upon the average suspended particulate reading for each SMSA. The total benefits for reducing suspended particulates from 1970 levels to 45 $\mu g/m^3$ summed for all of the 168 SMSA's were $5.0 billion per year.[30]

A recent study by Watson and Jaksch (1978) represents the best effort to employ a model based on the theory of household behavior to derive a measure of welfare change as distinct from a measure of changes in expenditure. They assumed that the observed level of cleanliness and the expenditure necessary to attain it are variables which are determined by supply and demand. They assumed that cleanliness matters to people, and people will want more of it when its price is lower. A demand curve for cleanliness can then be derived as a function of its price or cost and other variables such as income. Suppose that from an initial equilibrium, air quality is improved and therefore the supply price of

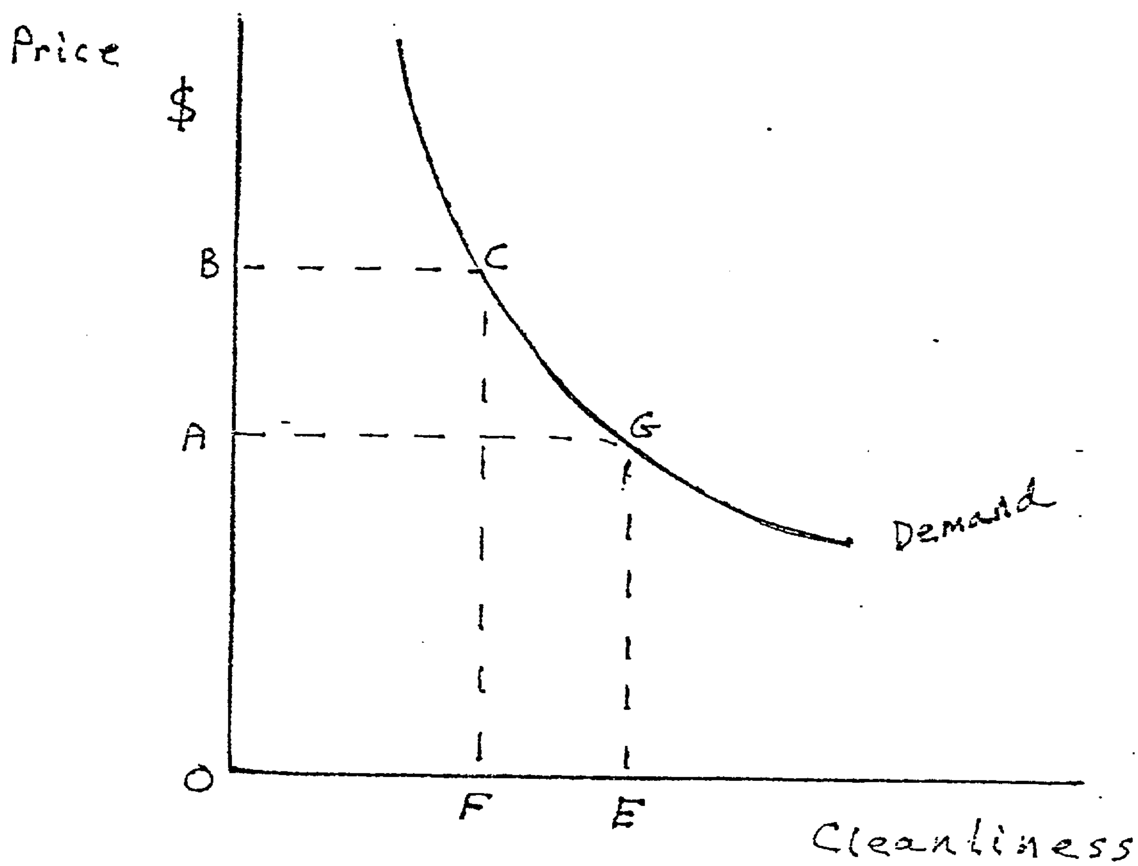
[30] Liu and Yu do not specify the year for which unit costs and prices apply. But I infer from their discussion on page 69 that they used current dollar values at the time the research was carried out, that is, in 1974-75.

cleanliness is reduced. Total expenditure on cleaning costs will increase, remain the same, or decrease depending upon whether the elasticity of demand for cleanliness is greater than, equal to, or less than 1. Changes in cleaning expenditure would measure benefits accurately only if the elasticity of demand for cleanliness were zero.

Watson and Jaksch estimated that this elasticity is close to unity, implying that changes in expenditure would be close to zero as air quality changes and that expenditure changes would substantially underestimate true benefits. This conclusion is based upon their regression analysis of the Booz, Allen, Hamilton (1970) data on frequency of cleaning and total suspended particulate levels in Philadelphia. In contrast to Liu and Yu, Watson and Jaksch used the original survey data and included variables reflecting socioeconomic characteristics and responses to questions concerning attitudes toward household cleanliness. The latter were intended to characterize differences in preferences or utility functions across households. They reported finding generally low $R^2 (<.2)$ in all cases. They concluded that "frequency of cleaning is not significantly different across households with different socioeconomic characteristics, attitudes, and exposure to pollution (page 22)." They did not present their regression results and tests of significance.

Watson and Jaksch's conclusion regarding the lack of a relationship between pollution and cleaning frequency implies a particular functional form for the demand curve for cleanliness,

Figure 1



namely one of unitary elasticity or constant expenditure. Watson and Jaksch then used a function relating pollution to degree of physical soiling and data on cleaning costs to estimate changes in the price of cleanliness associated with changes in suspended particulate levels. Benefits of reduced pollution are measured from changes in consumer surplus as shown in figure 1. With the initial price of cleanliness at OB, the household chooses cleanliness level OF. Lower pollution lowers physical soiling and the cost of cleanliness to OA, and the household responds by choosing a higher cleanliness level, OE. Given the unitary elasticity of the demand function, expenditures do not change; but consumer surplus is increased by the area ABCG. This is the measure of benefit per household.

Utilizing data on the spatial distribution of pollution in the Philadelphia SMSA in 1970, Watson and Jaksch estimated the benefits of reducing suspended particulate levels to the national primary and secondary standards at all monitoring points. The following figures are provisional and subject to possible revision. Benefits for the Philadelphia SMSA were calculated to be between \$23 and \$150 million for the primary standard and between \$44 and \$314 million for the secondary standard, in 1978 dollars. Gains per household were \$16-101 and \$31-213 for the moves to the primary and secondary standards respectively.

Assuming the spatial distribution around the SMSA mean is the same for all SMSAs, Watson and Jaksch also computed benefit measures for each of an additional 122 SMSA's. The aggregate for all SMSA's including Philadelphia is between $.9 and $6.1 billion for the national primary standard and between $1.5 and $11.7 billion for the secondary standard. The most likely values were $2.6 billion for the primary standard and $4.6 billion for the secondary standard.

Hamilton (1979) used the Watson-Jaksch framework to estimate soiling benefits for six California SMSA's. He also assumed a unitary elasticity of demand for cleanliness. Per household soiling benefits for a 25% reduction in particulates of about $40 are comparable to those estimated by Watson-Jaksch. Aggregate benefits for 25% reduction in particulates for the six California SMSA's (including San Francisco-Oakland and Los Angeles-Long Beach) are $223 million in 1978 dollars. Applying the per household figure of $40 to the approximately 50 million households in metropolitan areas yields a crude estimate of national benefits of $2 billion per year.

Although the estimates of Liu and Yu and Watson and Jaksch are quite comparable considering the differences in their postulated change in pollution levels, they are based on different analytical models and approaches to utilizing the Booz-Allen Hamilton data. The Watson-Jaksch conceptual framework is clearly preferable. But their inference of a unitary elasticity of demand for cleanliness must be considered to be tentative.

As Table 3 indicates the composite annual average of particulates had been reduced almost to the secondary standard of 60 $\mu g/m^3$, while the 90th percentile had closed almost half of the gap between the 1970 level and the secondary standard. On this basis I infer that between a third and a half of the Watson-Jaksch estimate of the benefits of attaining the secondary standard had been realized by 1978. This works out to a range of between \$.5 and 5.0 billion per year. The most reasonable point estimate is \$2.0 billion per year. Since the Watson-Jaksch estimates are derived by examining postulated changes in pollution levels for each SMSA taken separately, they also provide the best estimate of the benefits to be realized by achieving the national primary and secondary standards for suspended particulates. Incremental benefits of moving from 1978 air pollution levels to full achievement of the primary and secondary standards are estimated at \$.6 and \$2.6 billion per year respectively.

Vegetation

Damages to ornamentals and commercial crops may be caused by ambient concentrations of oxidants, sulfur compounds, and fluorides. However fluoride pollution is a localized problem and will be given no further attention in this report. The first effort to estimate national damages to vegetation due to oxidant pollution was made by Eenedict, Moore and Smith (1973), hereafter referred to as BMS. The BMS estimate was based on pollution levels and cropping patterns in 1964. Photochemical oxidant pollution potential was estimated by county based upon fuel consumption data. Plant sensitivity or crop loss coefficients were estimated based on a review of the literature. Crop loss factors were apparently derived from the most part from visual inspection of such things as leaf damage. More subtle effects on plant vigor and yield are probably underestimated here. These factors were applied to estimates of the values of crop production and ornamental plants in counties with high pollution potential.[31] BMS estimated crop damages of $78 million and the replacement of damaged ornamental plantings at $43 million per year. The total oxidant related damages summed to $121 million per year in 1964 dollars.[32]

There are several limitations to this somewhat crude method. First, emissions and fuel consumption are likely to be a poor proxies for oxidant levels. Oxidant formation depends not only on emissions of both hydrocarbons and nitrogen oxides, but also on meterological conditions and the movement of air masses. In many parts of the country, because of air pollutant transport patterns,

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[31] For a further description of the BMS method, see Waddell (1974) and National Academy of Sciences (1974). These studies also include reviews and discussion of other efforts to quantify vegetation losses on a local level. For further discussion of the appropriate methodology for defining and estimating benefits due to reduced crop loss and improved agricultural productivity see Freeman (1979a) chapter 9, and Adams, Thanavibulchai, and Crocker (1979).

[32] Total estimates of vegetation damages by Waddell (1974), National Academy of Sciences (1974), Babcock and Nagda (1973), Jackson et al. (1976), and Liu and Yu (1976) are also based on the Benedict data and methodology.

peak photochemical oxidant readings occur tens of miles downwind from the major sources of emissions. And the degree to which emissions are converted into photochemical oxidants depends strongly upon local climate and weather conditions. BMS did adjust emissions estimates by local dispersion factors. But these adjustments are not likely to adequately reflect the complexities and spatial and temporal variability of the processes involved.

Second, the method does not take into account the possible effect of reduced crop output on food product prices. If reduced output leads to higher prices, consumer welfare losses may be magnified, especially for products with inelastic demands. Producers may respond to high pollution levels by choosing to grow more resistant types of crops, or by changing locations. Where these adjustments occur, applying crop loss factors to observed crop patterns will underestimate true economic losses since it will fail to capture the costs associated with these adjustments.[33]

The estimate by Heintz, Hershaft, and Horak (1976) is a marked improvement over that of BMS in that it employed data on actual oxidant levels rather than oxidant potential, and it is based on more recent studies of dose-response relationships for crops at ambient pollution levels. Heintz, Hershaft, and Horak reviewed and summarized a number of controlled field experiments dealing with actual yield reductions observed with ambient concentrations of oxidants in the range of .05 ppm to .08 ppm. On

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[33] See National Academy of Sciences (1974) pp. 371-372 for further discussion of the BMS methodology.

the basis of this review, they concluded that the best estimate for the average yield loss to crops exposed to that level of photochemical oxidants is about 15% with a low-high range of 5-25%.[34] They identified those counties within 100 miles of major urban centers for which on the basis of existing measurement of oxidant levels and/or knowledge of oxidant transport patterns, there is reason to expect high oxidant readings. They then determined the total value of agricultural crops raised in these counties in 1973 ($18.7 billion). Applying the expected loss factor to this total production gives an estimate of 1973 damages due to oxidants of between $.9 and $9.5 billion per year in 1973 dollars. Their most reasonable point estimate is $2.8 billion. Losses to ornamental plantings were estimated at $.1 billion per year by inflating the BMS estimate.

I take the estimate by Heintz, Hershaft and Horak as the best available measure of the damages to vegetation due to photochemical oxidant pollution levels. Adjusting this figure to 1978 levels by the growth in gross farm income leads to an estimate of

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[34] Some corroboration for this loss factor is found in Hamilton (1979). He used a theoretical loss function derived by Larsen and Heck (1976) to derive estimates of percent of leaf injury to be expected at different ozone concentrations for ten different crops. The averages of these figures for different ozone levels fell within the range assumed by Heintz, Hershaft and Horak. On the other hand, Adams, Thanavibulchai, and Crocker (1979), using a more comprehensive economic methodology which also took into account input and output substitution and price effects, estimated losses due to oxidant pollution for 14 vegetable and field crops in southern California. Their estimate of losses amounted to only 1.48% of the total value of crop production in the study area.

between $1.2 billion and $12.2 billion per year. The most reasonable point estimate is $3.6 billion per year. This estimate may still be too low since it fails to account for producers' changes in cropping patterns in response to pollution levels and possible changes in product price (Freeman 1979a, pp. 238-240). It also probably underestimates the welfare losses associated with damaged ornamental plants. Finally, Hamilton (1979) suggests that more recent data on the impact of oxidants on western forests, especially in southern California, gives reason to believe that forestry damages have also been substantially underestimated.

Given that this level of damages is based on 1973 oxidant levels, what can be said about the benefits which have been realized by control programs since 1970? National data from the Environmental Protection Agency (1978) show little improvement in the emissions rates of oxidant precursors and ambient oxidant concentrations. However EPA reports some improvement in both the emission rates and oxidant levels in southern California, the area in which a substantial portion of the national crop damages have been experienced. Data from the Council on Environmental Quality (1978) also indicates a reduction in violations of the national standard nationwide. Using the 20% figure, benefits due to the actual reduction in oxidant levels, primarily in southern California may lie in the range of $.2-2.4 billion per year. The most likely point value is $.7 billion per year. Additional unquantified benefits may be attributed to the likelihood that the Clean Air Act Amendments of 1970 have prevented a more significant increase in oxidant levels arid related vegetation losses.

There is very little hard data on vegetation losses due to sulfur compound pollution. Benedict, Moore, and Smith (1973) estimated them to be \$6 million per year as of 1964. Since the date of the BMS study, there has been increasing concern with the effect of acid rain from sulfur compound pollution on agricultural and forest productivity. The National Academy of Sciences (1975, page 181-182) suggests that sulfur dioxide damages to vegetation may have been underestimated in the past and that the potential for damages from acid rain is serious. They also suggest \$.5 billion per year as a best guess for acid rain damages in the northeastern United States (p. 624). This figure may be too low because of the potential for long term, cumulative impacts on soil fertility and because of ecological impacts to fish, wild life, etc. For these reasons, I take the range of possible benefits to controlling sulfur compounds to be \$0-2 billion per year. The most reasonable point estimate is \$.5 billion per year. Since there has been virtually no downward trend in sulfur particulate levels in the eastern part of the country, it seems prudent to conclude that <u>realized</u> benefits as of 1978 are zero.

<u>Materials Damages</u>

The early 1970's saw the publication of a number of estimates of damages to various types of materials and equipment due to air pollution. Waddell (1974) and National Academy of Sciences (1974) provide useful descriptions, summaries and critiques of many of these studies. Table 6 summarizes these studies and includes an allocation of monetary damages by type of material and pollutant most responsible.[35]

Most of these studies used technical, engineering, or best judgment damage functions, made some estimate of the quantity of materials at risk to pollution exposure and damage based on data on aggregate production or inventories, and estimated monetary values on the basis of the costs of more frequent repair, maintenance, painting, or replacement. This basis for valuation may not capture the more subtle or indirect costs such as those due to down time or reduced performance. On the other hand producers and households may respond to materials damages by other less costly adaptations and adjustments, for example material substitutions, than those on which these estimates are based. These factors would tend to cause estimates of damages to be biased upward. It is difficult to judge the net effect of these offsetting biases.

The weakest link in most of these studies is the estimation of quantities of materials at risk and exposure levels. Gillette's (1975) study is noteworthy in its utilization of actual ambient air quality data for sulfur dioxide to estimate degrees of exposure of materials. Other studies used more crude approaches such as dichotomous variables, that is, polluted vs. nonpolluted. The National Academy of Sciences (1975, pp. 695-699) argues that these estimates are biased downward including omitted

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[35]Many of the studies do not specify the reference year and price level. In what follows, it will be assumed that all studies are based on 1970 prices.

Table 6

Estimates of Types of Materials Damages

Class of materials and source of study[a]	Attributed to: Sulfur compounds & Particulates	Oxidants & Nitrogen Oxides	TOTAL
1. Electrical Switches (Robbins, 1970)	65 million		65 mil.
2. Electrical Components (ITT, 1970)	16. "		16. "
3. Paints (Salmon, 1970)	1,195 "		1,195 "
4. Paints (Spence & Haynie, 1972)	704 "		704 "
5. Zinc Corrosion (Salmon, 1970)	778 "		778 "
6. Corrosion (Fink, et 'al, 1971)			
Zinc	1,353 "		1,353 "
Other	97 "		97 "
7. Corrosion and Paint (Gillette, 1975)	400 "		400 "
8. Textiles and Dyes[b] (Salvin, 1970)	636 "	330 million	936 "
9. Fibers (Salmon, 1970)		358 "	358 "
10. Rubber and Elastomers (Mueller & Stickney, 1970)		355[c] "	355 "
11. Rubber (Salmon, 1970)		194 "	194 "
12. All Other Materials (Salmon, 1970)	1,272[d] "		1,272 "
13. All Other Materials not covered by other studies (Salmon, 1970)	400[e] "		

Notes to Table 6:

a) Information on these studies is based on Waddell's (1974) survey and review, except where otherwise noted.

b) These totals and allocations are based on the review of Salvin (1970) in National Academy of Sciences (1974), pp. 384-388. Soiling of fabrics is excluded.

c) This is adjusted to exclude retail mark-ups on higher manufacturing costs. See Waddell (1974), pp. 80-81 or National Academy of Sciences (1974), pp. 379-383.

d) Salmon does not allocate this total to pollutants. But since the vast bulk of the materials are metal or masonry, they have been allocated to sulfur compounds and suspended particulates.

e) This estimate is due to Waddell (1974). See also note d.

Sources:

Fink, F. W., F. H. Buttner, and W. K. Boyd, Technical Economic Evaluation of Air Pollution Corrosion Costs in Metals in the United States, Environmental Protection Agency, Research Triangle Park, 1971.

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ITT Electra-Physics Laboratories, Inc., A Survey and Economic Assessment of the Effects of Air Pollutants On Electrical Components, Environmental Protection Agency, Research Triangle Park, 1971.

Mueller, W. J. and P. B. Stickney, A Survey and Economic Assessment of the Effects of Air Pollution on Elastomers, National Air Pollution Control Administration, Raleigh, North Carolina, 1970.

Robbins, R. C., Inquiry Into the Economic Effects of Air Pollution on Electrical Contacts, National Air Pollution Control Administration, Raleigh, North Carolina, 1970.

Salmon, R. L., Systems Analysis of the Effects of Air Pollution on Materials, National Air Pollution Control Administration, Raleigh, North Carolina, 1970.

Salvin, V. S., Survey and Economic Assessment of the Effects of Air Pollution on Textile Fibers and Dyes, National Air Pollution Control Administration, Raleigh, North Carolina, 1970.

Spence, J. W., and F. H. Haynie, Paint Technology and Air Pollution: A Survey and Economic Assessment, Environmental Protection Agency, Research Triangle Park, 1972.

categories of damages (corrosion and paint damage to automobiles, and deterioration of art works and historic buildings) and under-estimation of materials at risk.

Despite these weaknesses, these studies provide the only basis for developing an estimate of national benefits due to reduced materials damages. After evaluating the conceptual and empirical bases of the various studies and considering problems of overlaps and gaps in coverage, Waddell selected the following figures to compile a national estimate of materials damages:

Elastomers	(line 10)	$500 million
Corrosion	" 7	400 "
Textiles	" 8[36]	200 "
Paints	" 4[37]	700 "
Other	" 13	400 "
Total		$2.2 billion

The studies used by Waddell have also been the basis for estimates by Liu and Yu (1976), Heintz, Hershaft, and Horak (1976), and Small (1977). These authors made various adjustments for inflation and other factors.

In what follows I have chosen to make some minor revisions in Waddell's treatment of these various studies and to attempt to

[36] Waddell's description of the Salvin study differs substantially from that in National Academy of Sciences (1974). The Table 6 figures are based on the latter; while the figure in the text comes from Waddell.

[37] Waddell included the allowance for retail markup on increased production costs. The markup had been deducted to arrive at the figure included in Table 7.

allocate them by class of pollutant. The results are shown in Table 7. First, I have included estimates of damages to electrical switches and electrical components. Second, I have utilized the National Academy of Sciences (1974) description of the Salvin study of textiles and dyes. Finally, I have used the lower figure for elastomers, that is excluding retail markups on production costs. The total damages in 1970 dollars due to suspended particulates and sulfur compounds comes to $2.2 billion. Using the consumer price index, the 1978 dollar figure is $3.7 billion. The total for oxidants and nitrogen oxides is $655 million in 1970 or $1.1 billion in 1978. The total for all classes of damages is $4.8 billion. Lacking a better basis for doing so, I assume a low-high range of ± 50% or $2.4-7.2 billion per year.

Remember that this is for total damages in 1970. To estimate benefits realized by actual improvements in air quality since 1970, I assume a 20% improvement in suspended particulates, sulfate pollution and oxidant levels since 1970. Assuming that the damage function is linear in this range, realized benefits to stationary source control are estimated to lie in the range of $.4-$1.1 billion per year. The most reasonable point estimate is $.7 billion. Realized benefits due to the control of oxidants and nitrogen oxides are between $.1 and $.3 billion per year. The most reasonable point estimate is $.2 billion.

Table 7

Materials Damage Estimate

(1970 Dollars)

Attributed to:

	Category	Sulfur compounds and particulates	Oxidants and Nitrogen Oxides
1.	Electrical Switches (line 1 of Table 6)	$ 65 million	
2.	Electrical Components (line 2 of Table 6)	16 "	
3.	Textiles and Dyes (line 8 of Table 6)	636 "	300 million
4.	Paints (line 4 of Table 6)	704 "	
5.	Elastomers (line 10 of Table 6)		355 "
6.	Corrosion (line 7 of Table 6)	400 "	
7.	Other (line 13 of Table 6)	400 "	
		2,221 Million	655 Million

TOTAL = 2.9 Billion

Aesthetics and Visibility

Air pollution can cause odors which affect people's utility and welfare. When smog impairs the view of a nearby mountain range, this could result in a loss of amenity values. Since aesthetic effects of this sort are not easily quantifiable and may not be associated with the production or consumption of market goods and services, they pose difficult measurement and valuation problems. But to the extent that they involve utility changes and willingness to pay, they are nevertheless every bit as real, in an economic sense, as the impairment of health or agricultural productivity.

Those aesthetic and visibility effects that impinge on the residential sites of households are likely to be reflected in housing prices or property values. We discuss benefit estimates based on property value studies in the next section. An alternative approach to estimating aesthetic and visibility benefits is to ask people what they would be willing to pay rather than do without the aesthetic and visibility improvements. There have been several studies designed to determine willingness to pay for air quality and related aesthetic improvements through survey questionnaires and bidding games with respondents.[38]

The first study (Randall, Ives, and Eastman, 1974) dealt with the aesthetic impact of the Four Corners Power plant complex in

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[38]For discussions of some of the methodological and empirical aspects of estimating benefits from survey data, see Freeman (1979a) and the three studies cited below.

the Southwest. Aesthetic impacts covered by the study included both those directly associated with air pollution (haze, visible smoke plume, and so forth) and effects, on the landscape due to unreclaimed soil banks from mining and EHV transmission lines. Respondents indicated an average bid per household of $50 per year to move to a somewhat improved aesthetic state and an additional $85 per year to go beyond that to a substantially improved state (both figures in 1972 dollars). On the basis of their sample, Randall, _et al_. estimated that the aggregate bids for the region are $15.5 million and $24.6 million for the two stages of improvement.

The second study also examined the aesthetic impacts of power plants in rural environments. Brookshire, Ives, and Schulze (1976) asked local residents and recreationists their willingness to pay to prevent the scenic and aesthetic degradation associated with a proposed power plant siting near Lake Powell. Responses were in the range of $2-4 per recreation day in 1975 dollars. Aggregate bids were estimated to be $.4 million, and $.7 million per year respectively.

In the third study (Brookshire, _et al_., 1979), households in the Los Angeles area were asked their willingness to pay for improvements in air quality as portrayed by photographs showing different degrees of visibility impairment. Respondents were also given information on the health effects of air pollution. Household bids for a 30% improvement in air quality averaged about $29 per month in 1978 dollars. Average annual benefits for the South

Coast Air Basin were estimated to be $.65 billion. A breakdown of the responses indicates that about 30% of this total is attributable to aesthetics with the remainder due to acute and chronic health effects.

None of these studies lends itself to an estimate of national aesthetic and visibility benefits. However the results do indicate the potential for significant benefits in this category, perhaps especially in the western and southwestern regions of the country.

Benefits to Residential Property Values

People may experience disutility or welfare losses when exposed to reduced air quality even when there are no significant effects on health, materials, vegetation or soiling. Because these "aesthetic" effects are not typically linked to specific economic activities or tangible physical effects, they are particularly difficult to measure directly. One approach to measuring aesthetic benefits is to describe potential aesthetic changes and ask people through survey instruments for their willingness to pay to achieve those changes.

Aesthetic effects which are specific to housing locations may be reflected in housing price differentials. To the extent that this is so, aesthetic benefits can be measured by estimating the demand for the aesthetic-creating characteristics of the house. Such characteristics might include ambient air quality, ambient noise levels, proximity to a lake or recreational water body. Estimating the demand for such characteristics of housing involves a two-step procedure in which first the implicit price of the characteristic is estimated by examining the relationship between the prices of housing units and their characteristics, and then the implicit price is regressed against observed quantities to estimate the demand function itself.[39]

Houses constitute a product class differentiated by characteristics such as number of rooms and size of lot. Any large urban area has in it a wide variety of sizes and types of housing with different locational or neighborhood characteristics and, if air quality varies across the urban area, different levels of air quality. An important assumption of the technique is that the urban area as a whole can be treated as a single market for housing services. Individuals must have information on all alternatives and must be free to choose a housing location anywhere in the urban market. It is as if the urban area were one huge supermarket offering a wide selection of varieties of housing. Of course, households cannot move their shopping cart through the supermarket. Rather their selection of a residential location fixes for them the whole bundle of housing services. It is much as if shoppers were forced to make their choice from an array of already filled shopping carts. Households can alter the

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[39] This procedure is known in the economics literature as the hedonic price technique. See Rosen (1974) for an elaboration of the theoretical basis of the technique. A number of theoretical and empirical issues regarding the application of the technique to air and water quality are discussed in Freeman (1979a), chapter 6.

level of any characteristic by finding an alternative location alike in every respect but offering more of the desired characteristic. It must be assumed that the housing market is in equilibrium, i.e., that all households have made their utility maximizing residential choices, given the prices of alternative housing locations, and that these prices just clear the market given the existing stock of housing and its characteristics.

In principle, if there are enough houses with different combinations of rooms and lot size, it is possible to estimate an implicit price relationship which gives the price of any type of housing as a function of the quantities of these two characteristics. Where P_h is the price of housing, this function can be represented as:

$$P_{h_i} = F(S_i, N_i, Q_i),$$

where S_i represents a set of structural characteristics for the ith housing unit such as size, number of rooms, age, type of construction; N_i represents a set of characteristics of the neighborhood in which the ith house is located, e.g., quality of local schools, accessibility to parks, stores, or work place, crime rates; and Q_i is the level of air quality at the ith site. The derivatives of this function with respect to the characteristics give the implicit prices. For example, the difference in price between two houses with different numbers of rooms but identical in all other respects can be interpreted as the implicit price of additional rooms.

This first stage develops a measure of the implicit price or marginal willingness to pay for air quality, but it does not directly reveal or identify the inverse demand function for air quality. The second stage of the technique is to combine the quality and implicit price information in an effort to identify the inverse demand function for Q. It is hypothesized that the household's demand price or willingness to pay for air quality is a function of the level of air quality, household income, and other household variables which influence tastes and preferences. Each household's observed implicit price is taken to be a measure of its equilibrium marginal willingness to pay. Can this demand function be identified with the information at hand? This question is discussed in Freeman (1979a, Chapter 6). The answer is yes, in principle.

Once the households' marginal willingness to pay functions have been identified, they can be used to estimate the benefits of air pollution control policies. Each household's benefit is the area under (integral of) the marginal willingness to pay function between the old and new levels of air quality at that site. Aggregate benefits are obtained by summing over all households.

A number of studies have used this technique to estimate the marginal implicit price of air quality. Harrison and Rubinfeld (1978a) Nelson (1978) and Brookshire, et al. (1979) have also used implicit price data to estimate the demand for air quality and benefits of air quality improvement.

<u>Stationary Sources</u>: Waddell (1974) was the first to attempt an estimate of national benefits from controlling air pollution through the use of property value information. His estimate was derived from six different studies involving eight separate cities such as St. Louis, Kansas City, Washington, and Chicago.[40]

The conceptually correct approach to deriving a national estimate of benefits from property value studies would involve four steps: (1) Estimating the implicit price function for housing for all major urban areas; (2) estimating the inverse demand functions for air quality for each major urban area; (3) using each city's set of demand functions to estimate the benefits of the specified improvement in air quality; and (4) summing these estimates for each city to arrive at a national aggregate measure.

None of the property value studies available at the time of Waddell's work included sufficient information to enable an appropriate computation of benefits. Waddell was forced to use a less than ideal approach to extrapolation from the limited number of imperfect studies in his effort to obtain an aggregate national benefit measure. He proceeded in the following manner. After reviewing the existing studies, he picked a value for a representative household's marginal willingness to pay for air quality improvements (as measured by sulfation rates) which lay in the middle of the range found by those studies. He estimated marginal willingness to pay for a 0.1 $\mu g/100cm^2/day$ reduction in sulfation to lie in the range \$20-50 per year with a best estimate of

[40]For a more comprehensive and up-to-date review of existing studies, see Freeman (1979a, appendix to Chapter 6).

$35. Waddell assumed that this marginal willingness to pay would be the same for all households in all urban areas independent of their existing air pollution levels, income, and other socio-economic characteristics. He also assumed that this marginal willingness to pay would be constant for each household for non-marginal changes in air quality. This is equivalent to assuming that the demand curve for air quality improvements is horizontal. It can be shown that this results in an overestimate of benefits.[41]

Then for each SMSA (the actual number of SMSA's is not stated) benefits were computed according to the following formula;

$$B = - \sum_{i=1}^{n} W \; h_i \; \Delta P_i$$

where n is the number of SMSA's, W is the marginal willingness to pay per household (assumed the same and constant for all SMSA's), h_i is the number of households in the ith SMSA, and ΔP_i is the change in sulfation levels realized by reducing pollution from 1970 levels to a desired background level of 0.1 $\mu g/100 cm^2/day$ annual arithmetic mean. Note that this assumes that the estimated marginal willingness to pay of single family dwelling homeowners is representative of the marginal willingness to pay of all households. On the basis of these assumptions, Waddell estimated national benefits to be $5.9 billion per year in 1970 with a low-high range of $3.4-8.4 billion.

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[41]See Freeman (1974).

Waddell's estimate will be the basis of our revised and updated estimate of realized benefits to residential property values. The first adjustment is to take account of the upward bias due to the assumption of constant marginal willingness to pay for households. Harrison and Rubinfeld (1978b) have provided quantitative estimates of the magnitude of upward bias due to this and other elements of model misspecification. They showed that the magnitude of upward bias due to the assumptions employed by Waddell is itself an increasing function of the size of the postulated change in pollution levels. Their estimates of upward bias are as follows:

A Pollution	Upward Bias
10%	45.9%
25%	56.7%
50%	79.5%

Waddell's benefit estimate is probably based on observations on cities with a wide range of required air pollution improvements. Some may have been as small as 10%, and others may have been well over 50%. In the absence of any better basis for determining the degree of upward bias, we assume that Waddell's estimates are biased upward by 50%. Therefore adjusting to take account of diminishing marginal willingness to pay as air quality improves, the revised estimates for 1970 are $3.9 billion with a low-high range of $2.3-5.6 billion.[42]

[42] if B_t represents the true benefit figure, we have $B_t \times 1.5 = B_w$, or $B_t = B_w \div 1.5$.

This revised figure is updated to 1978 values through the use of the consumer price index and adjusted to take account of the growth in housing stock between 1970 and 1978. These adjustments to Waddell's figures lead to an estimate of benefits to the 1978 population of improving sulfation levels from 1970 values to the background level in the range of $4.7 to $11.5 billion per year. The most reasonable point estimate is $8.0 billion per year. Since sulfation and suspended particulate readings tend to be highly correlated, we take this to be an estimate of the potential benefit for controlling all sulfur compound and particulate pollution.

Assuming a realized 20% reduction in sulfur compound and particulate pollution levels between 1970 and 1978, this adjustment to Waddell's figures leads to an estimate of realized benefits of between $.9 to $2.3 billion per year. The most reasonable point estimate is $1.6 billion. Note however that this computation assumes that the relationship between national benefits and pollution levels is linear. If people have higher marginal willingness to pay for improved air quality at high pollution levels, then the realized benefits of the first 20% reduction in air pollution will be larger than 20% of the total estimated above. Thus the estimate of $1.6 billion realized benefits may be too low.

The only other estimate of the national benefits of controlling stationary source pollution based on property values was prepared by Polinsky and Rubinfeld in conjunction with the National Academy of Sciences' study of automotive air pollution control policy (National Academy of Sciences, 1974). Polinsky and Rubinfeld used three different models to provide alternative estimates of the benefits of a 45% control of suspended particulates and sulfur compounds in St. Louis. Property value data were for 1960. Pollution readings were taken in 1963. According to the preferred model, benefits in 1960 dollars were $81 million per year. Polinsky and Rubinfeld assumed that the St. Louis SMSA contains about 2% of the national urban population. Thus they estimated national benefits at $4.1 billion

This figure can be made comparable to the revision of Waddell's estimate in a similar manner. Thus after inflating for the growth in the consumer price index and the change in the housing stock between 1960 and 1978, and assuming only a 20% reduction in pollution, realized national benefits would be $6.9 billion per year in 1978 dollars. It should be noted that this extrapolation to national benefits from an estimate based on a single SMSA is dangerous since St. Louis may not be representative of other cities either in terms of demand for air quality or pollution levels. Specifically if pollution levels in St. Louis in 1960 were higher than the average of all urban areas, this figure would be an overestimate of national benefit.

Taking account of the Polinsky and Rubinfeld estimate and the fact that revision of the Waddell estimate is likely to be biased downward, we estimate realized benefits to residential property values from the control of stationary source air pollution to be in the range of $.9 to $6.9 billion per year in 1978. The most reasonable point estimate is taken to be one-third of the upper bound, or $2.3 billion per year. It should be noted that property value benefits almost certainly reflect such factors as reduced soiling and cleaning requirements, reduced materials damages to household paints, etc. Thus this estimate is not fully additive with all of those other estimates given for other categories above. The question of double counting will be taken up below.

Mobile Source Benefits: There are three studies relating property values to mobile source pollutants which lend themselves to the estimation of national air pollution control benefits. The first was conducted by Harrison and McDonald in conjunction with the National Academy of Sciences study of automotive air pollution control policy (National Academy of Sciences, 1974). Harrison and McDonald estimated housing price equations for both the Los Angeles and Boston SMSA's. They estimated separate equations using nitrogen oxides, hydrocarbons, and oxidants as alternative pollution variables; and they estimated all equations in both linear and log form. Pollutant and housing value data were for 1970.

To estimate benefits, they assumed full compliance with the national automotive emission standards and used an air quality model to predict air quality levels as of 1990. They assumed that the average willingness to pay estimated from the housing value equations was constant and the same for all households. On the assumption that 30 million households across the country would be affected by auto pollution control, they computed national benefits for each possible combination of city, functional form, and pollutant. The results are shown in Table 8. Because of measurement problems, and because hydrocarbons are not themselves a pollutant of major concern, the hydrocarbon equation results are probably the least reliable. Omitting them, national benefits are estimated to lie in the range 2.0-7.2 billion per year. Harrison and McDonald note that suspended particulates and nitrogen oxides and oxidants are correlated in their sample. Thus some part of this total may be attributable to particulates and stationary source pollutants rather than mobile source pollutants. After examining the results of two stage least squares regressions, they estimate that two-thirds of this total is due to mobile source pollutants, roughly $1.5-5 billion per year.

This estimate can be made comparable with others in this section by inflating to 1978 dollars with the consumer price index. Also, their assumption of constant marginal willingness to pay leads to an overestimate of benefits of perhaps 50%. Taking this into account leads to an adjusted measure of the benefits of meeting the standards of $1.7-5.6 billion dollars per year in 1978 dollars. The midpoint of the range, $3.7 billion, is taken as the best estimate.

Nelson (1975) employed a similar methodology to estimate national benefits extrapolated from his study of property values and pollution levels in the Washington SMSA. Assuming a 45% reduction in mobile source pollution levels, he estimated benefits in 1970 of $810 million. Adjusting this for the growth in the consumer price index and taking account of the overestimation due

Table 8

National Benefits of Mobile Source
Air Pollution Control

Based on Property Values - 1970

Regression based on:	Extrapolated From a/	
	Los Angeles	Boston
NO_x	$ 2.4 - 5.7 billion	$3.0 - 7.2 billion
Hydrocarbons	12.3 - 10.0 billion	2.6 - .3 billion
Oxidants	4.2 - 3.2 billion	3.1 - 2.0 billion

Source: National Academy of Sciences (1974), pp. 235-236.

Notes: a/ In each entry the first number is based on a linear housing price equation while the second is derived from a log-linear specification.

to the assumption of constant willingness to pay leads to an estimate of national benefits of $.9 billion in 1978 dollars.

The third study was an extensive analysis of property values and nitrogen oxide levels in Los Angeles. Brookshire, et al. (1979) estimated the benefits of mobile source control for Los Angeles both with property value data and with a survey or bidding game approach. Data were collected for the years 1977 and 1978. A 30% reduction in pollution was postulated. Using the housing value equation to estimate a marginal willingness to pay function, and using the latter to compute the benefits of non-marginal changes yielded an estimate of average annual benefits per household of $511 and annual benefits for the Los Angeles area of $.95 billion per year.[43] At the same time, Brookshire et al. conducted a survey of willingness to pay to obtain improved air quality. Average annual bids amounted to $316-353 per household and annual benefits for the Los Angeles area were $.58-.65 billion per year.

This figure can be used to extrapolate to a national estimate of benefits following assumptions similar to those utilized by Harrison and McDonald and Nelson. If the range of average annual benefits per household is $350-500, and 30 million households are affected by mobile source air pollution, national benefits could be $10.5-15 billion per year. This figure can only be

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[43] Results were also presented for a pair-wise comparison of communities and for Linear model with a constant marginal willingness to pay. Both were acknowledged by the authors to lead to overestimates of the true benefits. Those figures are not reported here.

a realistic estimate of national benefits if Los Angeles is representative of other cities with respect to both the demand for clean air and typical pollution levels. But since in fact Los Angeles has one of the worst mobile source air pollution problems in the country, extrapolation from the Los Angeles per household benefit to the nation as a whole is likely to lead to a substantial overestimate.

We have reviewed national estimates of mobile source pollution control benefits based on property values which range from $.9-$15 billion per year, with the latter being clearly an overestimate. The Harrison-McDonald best estimate of $3.7 billion per year may also be on the high side because the estimate is partly based on data from Los Angeles. We estimate that the benefits attributable to the control of mobile source pollutants lie in the range of $1 to $10 billion per year. The most reasonable point estimate is $2.0 billion per year. Again using the Council on Environmental Quality data on reduction in violations, we assume a 20% reduction in oxidant pollution, and realized benefits of between $0.2 and $2.0 billion per year. The most reasonable point estimate is $0.4 billion. Again, there is likely double counting involved if this number is simply added to estimates derived through other categories or benefits. This question will be taken up in a later section.

Inter-Urban Wage Differentials

In a pioneering study conducted in conjunction with the National Academy of Sciences' analysis of automotive air pollution control policies (National Academy of Sciences, 1974, pp. 243-255), Meyer and Leone interpreted differences in wages among cities as reflecting workers' willingness to pay (in the form of lower wages) to work and live in cities with higher levels of environmental and other amenities. After seeking to relate wage differentials to differences in a variety of urban amenities and disamenities, Meyer and Leone estimated the benefits associated with a 45% reduction in particulate, sulfur dioxide, and nitrogen dioxide levels.

First considering only the two stationary source pollutants, benefits were estimated at $6.1 billion and $2.1 billion respectively. Adjusting these figures for the growth in the consumer price index, growth in the labor force, and for an assumed 20% realized reduction in pollution leads to an estimate of $4.9 billion in 1978 dollars for suspended particulates and $1.7 billion for sulfur dioxide. This is a total of $6.6 billion for both stationary source pollutants. The Meyer-Leone estimate for nitrogen dioxides is $5.1 billion. Adjusting this for the growth in the consumer price index and the labor force since 1972 results in an estimate of $9.1 billion for 1978. These are potential, not realized benefits.

These estimates must be considered tentative. In addition to the empirical question of whether other influences on urban wage levels have adequately been controlled for, there are questions of how estimates of benefits should be computed from observations on wage differences across cities. For example, can benefits computed from wage differences be added to benefit measures derived by other means such as property values, or are they alternative ways of measuring the same phenomenon? It can be shown that under certain conditions, wage differential benefits should be added to property value benefits in computing an aggregate benefit measure (Freeman, 1979a, pp. 118-121). However it is not known whether these conditions hold in practice or are applicable to the type of study carried out by Meyer and Leone.

Cropper (1979) has developed a labor supply model which takes into account interurban amenity differences, housing site location within the urban area, and wages. Under certain assumptions, knowledge of the labor supply function makes it possible to estimate the coefficients of the utility function and to compute willingness to pay for changes in the level of urban amenities. Cropper estimated the model using 1969 wage data. SO_2 was one of the amenity variables. She estimated that a laborer with income of $29,000 per year would be willing to pay $55-65 per year for a 20% decrease in SO_2 levels. She did not estimate aggregate or national benefits.

Air Pollution Control Benefits--Conclusions

Table 9 shows for each category of benefits the potential benefits (or actual damages) due to 1970 levels of pollution in those cases where this figure was estimated, and realized benefits associated with the change in pollution levels between 1970 and 1978--in all cases in 1978 dollars per year. It is probably not valid simply to add the property value benefits to other categories in Table 9 since tangible impacts on households due to soiling, damage to household paints, and perhaps morbidity are also likely to affect property values. It is also possible that some forms of benefits have not been captured by any of the measures reviewed here. An example is benefits associated with amenity and aesthetic values away from the residence.

We must consider the question of gaps and overlaps in our measures. For example, do estimates of health effects and property value differences associated with suspended particulates involve some double counting of the benefits of reducing concentrations of suspended particulates, or are there additional effects not captured by either measure? Although definitive answers to this question cannot be given here, the discussion will show that it should be possible to determine when such gaps and overlaps exist. It should also be possible to make some judgment as to the significance and likely direction of any biases in the estimated total benefit figures.

TABLE 9

Summary of Air Pollution Control Benefits - By Category
(In Billions of 1978 Dollars)

Category	Potential Benefits (Damages) At 1970 Pollution Levels[a/]		Realized Benefits	
	Range	Most Reasonable Point Estimate	Range	Most Reasonable Point Estimate
1. Health				
A. Stationary Source Mortality	a/	a/	$2.8 -27.8 billion	$13.9 billion
Morbidity	a/	a/	$ .29-11.5 billion	2.9 billion
Total			$3.1 -39.3 billion	$16.8 billion
B. Mobile Source	$0 - 2.0 billion	$1 billion	$ 0 - .4 billion	.2 billion
Total Health			$3.1 -39.7 billion	$17.0 billion
2. Soiling and Cleaning	$2.0-11.7 billion	$5.2 billion	$ .5 - 5.0 billion	$ 2.0 billion
3. Vegetation				
A. Stationary Source	$0 - 2.0 billion	$ .5 billion	0	0
B. Mobile Source	$1.2-12.2 billion	$3.6 billion	$.2 - 2.4 billion	$ .7 billion
Total Vegetation	$1.2-14.2 billion	$4.1 billion	$.2 - 2.4 billion	$ .7 billion

TABLE 9 (cont'd.)

4. Materials				
A. Stationary Source	$1.9-5.6 billion	$3.7 billion	$ .4-1.1 billion	$.7 billion
B. Mobile Source	$ .6-1.7 billion	$1.1 billion	.1- .3 billion	.2 billion
Total	$2.7-7.2 billion	$4.8 billion	$ .5-1.4 billion	$.9 billion
5. Property Values b/				
A. Stationary Source	a/		$ .9-6.9 billion	$2.3 billion
B. Mobile	$1.0-10 billion	$2.0 billion	$ .2-2.0 billion	$ .4 billion
Total			$1.1-8.9 billion	$2.7 billion

Notes

a/ Not estimated because of uncertainties about the form of the dose response function at lower pollution levels.

b/ Because of overlap, property value benefits are not strictly additive with the above categories.

Consider air pollution where health benefits are measured by mortality rate studies and aesthetic, soiling, and materials benefits to households are measured through property value differentials. The justification for measuring the two classes of benefits separately and adding them to obtain aggregate benefits is that only those effects which are perceived by individuals can influence property values, and that people have for the most part been ignorant of the effects of air pollution on their health and life expectancy. However, this simple resolution of the problem creates uneasiness. Some kinds of short-term health problems such as eye irritation and shortness of breath may be directly perceived as being caused by poor air quality. In addition, there has been a substantial increase in the information available to the general public about long-term health effects and related air quality levels around urban areas. Thus, perceptions of health effects may be influencing property value differentials and leading to the possibility of double counting. To gain a better understanding of the problem, it is necessary to consider in more detail exactly what is captured by each of the two approaches to measurement.

It will be helpful to develop a system of classification for different types of effects associated with adverse air quality. Two broad classes are perceived non-health effects and health effects. The former includes effects such as odor, taste, reduced visibility, soiling, and damage to external paints, all of which are perceived by individuals. A distinction can be made between those health effects which are perceived by individuals and those of which they are ignorant. The former are likely to be primarily clinical manifestations of short-term exposures to relatively high concentrations. Finally, it is necessary to distinguish between those effects which are caused by the person's exposure at home and those due to exposure as he/she travels around the urban air shed, to work, for shopping, for recreation. These "away from home" effects are independent of the individual's place of residence.

This classification scheme creates six subsets of effects: perceived non-health, perceived health, and unperceived health, and in all cases further divided between home and away. Property value studies can only capture those effects associated with the home which are perceived. Health benefits derived from mortality and morbidity studies capture both home and away effects and both perceived and unperceived health effects.

There is both an overlap and a gap. Perceived health effects at home are captured by both property value and health effects approaches. But perceived non-health effects associated with away from home exposures are not captured by either approach. Whether the addition of property value and health effect benefits results in an over estimate or an underestimate depends upon the relative size of the double counted and omitted categories. Nothing can be said about this question on _a priori_ grounds.

It is interesting to compare the property value total with those portions of the other four categories in Table 3 which one might plausibly assume would affect property values. Virtually all of the soiling and cleaning benefits of $2.0 billion should be reflected in property values. The estimate of vegetation damages is derived primarily from commercial agriculture. So this figure would not be captured by residential property value measures. A large portion of materials benefits are related to commerce and industry. But some portion, perhaps $.2 billion, is related to household paints and coatings. The health effects are difficult to allocate. But for illustrative purposes, assume that all the mortality effects involve such subtle biomedical processes that people are unaware of the connection between mortality and air pollution. Then none of this category would be reflected in property values. Also assume that morbidity effects are perceived to be due to air pollution, and that half of the total or $1.6 billion is associated with exposures at home. This means that a total of $3.8 billion of benefits measured by other means should also be reflected in property values. This is somewhat greater than the estimated property value benefits, which indicates that we cannot completely reconcile our accounting of benefits in different categories. However, the orders of magnitude are similar. And this lends some plausibility to these estimates.

The most conservative approach to treating property value benefits is to assume that those beneficial effects reflected in property values are also fully captured by our estimates of benefits in the other four categories. This means that amenity and aesthetic effects, both at home and away, are assumed to be zero. This is very conservative. However there is very little evidence on the possible magnitude of aesthetic effects at the national level. In their survey or bidding game approach to measuring benefits in Los Angeles, Brookshire, et al., (1979) found that perhaps 30% of respondents' willingness to pay is attributable to aesthetic rather than health benefits. They also found that the survey approach yielded estimates comparable in magnitude to their property value estimates. If this proportion is representative of the urban population as a whole then $.8 billion of property value benefits can be added to the total in categories 1-4. Given these assumptions total national benefits which have been realized due to reductions in pollution since 1970 lie in the range of $4.6 to $51.2 billion per year. The most reasonable point estimate is $21.4.

It is important to bear in mind the major qualifications attached to the estimates presented here. The first, of course, is that these estimates are not based on new data or research, but rather are derived from a review of other studies. Many of those studies can be, and have been, criticized on theoretical or methodological grounds or for the quality of data. The estimates presented here reflect my own judgments as to the quality and reliability of the results of the studies reviewed. Others may differ with the judgments made and the weights given here to various pieces of evidence.

Second, it is important to emphasize the range of uncertainty reflected in my estimates. The low and high values differ by more than a factor of ten. I believe that the probability is 0.9

that the true value of realized benefits lies within this range. But there is an equal probability that the true value lies either above or below the most likely point estimate of $21.4 billion.

About three-quarters of the total benefits are estimated to be in the health category. This estimate is subject to uncertainty about both the air pollution--health relationship and the value of health and reduced mortality. Only the former uncertainty is reflected in the range between low and high values. If the reader believes that $500,000 per death avoided is more reasonable than the $1 million used here, the estimated total benefits would be reduced by about 20%.

There is much controversy regarding estimates of the air pollution--mortality relationship. This is reflected in the wide range of the estimates of mortality benefits. Additional research now being done in several places may help to resolve the uncertainty. But it is also possible that the interactions among socioeconomic, environmental, and health variables are so complex and the degree of correlation among independent variables so severe that existing statistical techniques may be incapable of resolving the questions of whether and to what extent ambient concentrations of air pollution cause mortality. If this turns out to be the case, we will be left with a substantial uncertainty about a major question affecting environmental policy.

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