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Ambient sulfur dioxide pollution in Montana: An economic analysis of alternative standards

Theodore Patrick Otis
The University of Montana

AMBIENT SULFUR DIOXIDE POLLUTION IN MONTANA:
AN ECONOMIC ANALYSIS OF ALTERNATIVE STANDARDS

By

Theodore P. Otis

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A perennial problem facing the State of Montana is the allocation of its air resources between the disposal of industrial wastes and all other uses dependent on clean air such as health, visibility, and vegetation. This paper attempts to identify a preferred allocation by examining the economic implications of alternative ambient sulfur dioxide standards in Montana.

The alternative standards examined were the federal annual average standard of 80 ug/m^3 and the proposed Montana annual standard of 50 ug/m^3. The analysis considered three areas of Montana (Anaconda, Billings, and Helena) where 96 percent of the state's sulfur dioxide originates and where sufficient ambient monitoring has been performed. All three areas are federally designated noncompliance areas, i.e., they currently exceed the 80 ug/m^3 standard.

The major finding is that compliance with an 80 ug/m^3 ambient sulfur dioxide standard would result in annual health, vegetation, materials, and visibility benefits of $5.7 million. Compliance with the 50 ug/m^3 standard was found to result in an $8.6 million annual benefit. These estimates require the acceptance of two premises: (1) the higher of two alternative dose-response coefficients relating ambient sulfur dioxide to human mortality, and (2) the higher of two alternative values assigned to a reduced occurrence of premature death. Evidence from the literature suggests that the acceptance of both premises is valid.

Although substantial benefits were estimated, the available air pollution control cost information is not sufficiently refined as to allow selection of a preferred standard. The resulting policy implications are the substantial estimated benefits, a possible case for site specific standards, and definite need for further Montana specific research into both the benefits and costs of ambient sulfur dioxide air pollution. Key research needs are sulfate monitoring, vegetation damage functions, and applicable control techniques.

TABLE OF CONTENTS

LIST OF TABLES

Table		Page

LIST OF FIGURES

Chapter 1

INTRODUCTION AND METHODOLOGY

Introduction

Increasing public awareness has resulted in a greater demand for clean air in Montana. However, because of its public good characteristics, clean air is not allocated by market mechanisms; thus a collective decision-making procedure is required to allocate it among competing, mutually exclusive uses. The basic trade-off is using clean air to dispose and disperse wastes versus benefits associated with an unpolluted atmosphere: health, vegetation, aesthetics, etc.

Several basic criteria exist to allocate clean air. The efficiency criterion pursues an allocation based on maximizing the net social benefits associated with alternative levels of clean air. The distributional criterion examines how these net social benefits are shared or distributed among different groups in society: workers, consumers, property owners, stockholders, proprietors, and the general public. The ethical criterion considers, of course, issues of right and wrong, e.g., the ethical sanction against destruction of a plant or animal species. The economist can recommend a best allocation (assuming data availability) only on efficiency grounds. Distributional implications (who will benefit, who will lose from a given allocation) can be described by economists, but the relative worth of a gain to one individual versus a loss to another must be based on political decisions. Ethical considerations are beyond the economist's calculus.

This paper provides an analysis of alternative ambient sulfur dioxide standards based on the efficiency criterion. The selection of efficiency as a criterion to allocate Montana's clean air is generally in compliance with the legal mandate set forth by its residents; however, an apparent contradiction does exist between the use of efficiency criteria to set sulfur dioxide standards and the legal mandate. Section 69-3905 of the Clean Air Act of Montana, R.C.M. 1947, requires the Board of Health and Environmental Sciences to establish ambient air standards that

> achieve and maintain such levels of air quality as will protect human health and safety, and to the greatest degree practicable, prevent injury to plant and animal life and property, foster the comfort and convenience of the people, promote the economic and social development of this state, and facilitate the enjoyment of the natural attractions of this state.

Under this legal mandate ambient air standards must be based on a combination of ethical and efficiency criteria--ethical to the extent that standards must be set at a level that will at least protect public health and the efficiency criterion used only when going beyond the health threshold to protect vegetation, materials, aesthetics, etc. Evidence suggests, however, that a threshold level of ambient sulfur dioxide before which no health effect occurs may either be nonexistent or existent at a very low level. The most extensive study to date correlating health and air pollution supports this view. Lave and Seskin (1977, p. 316) argue:

> As discussed previously, our results do not lend support to the threshold concept. When we examined alternative specifications, we

> found that generally a simple linear specification "fit" the data as well as other functional forms. We found (for the air pollutants under consideration) little indication that specific levels could be considered "safe" levels, below which no health effects could be detected. Given our findings, we feel that the notion of an air pollution standard (or threshold) for a pollutant that will "protect the public health or welfare" is not meaningful. It translates into setting standards on the basis of damages (or benefits) alone. Instead, we would argue that standards should be based on benefit-cost trade-offs. That is, we believe that society must weigh the costs and benefits of achieving various levels of air quality, and once a desired level is selected, the standards corresponding to it should be determined.

Even if no health effects exist at low levels of pollution, a definite value in clean air remains which is associated with the risk aversion attitudes of the public. That is, there is a value associated with reducing the *potential* for adverse health effects. For both of the above reasons, this study abandons the ethical distinction in the Montana Clean Air Act between human health effects and other effects and treats all effects symmetrically.

Methodology

In order to identify the best standard based on efficiency criteria, two general types of information are needed: physical relationships and values. The key physical relationships include three items: (1) the relationship between actual plant-specific emissions and the ambient air quality which can be determined by sufficient *monitoring* sites or through dispersion models, (2) the effect of a given level of pollution (ambient level) on receptors--human, animal, plant, or material--the so-called *damage function,* and (3) the *control technology* (or other pollution reducing approaches such as different inputs or reduced production) that will achieve a given level of emission reduction. Given physical relationships, *values* must be placed on the extent of the damage and the resources used for control.

Given the availability of this information, a best standard can be identified by examining the costs and benefits of air pollution control. The methodology, in functional rotation, follows.

1. Ambient sulfur dioxide = f (point source emissions).
2. Physical damage = g (ambient sulfur dioxide).
3. Control benefit = h (physical damage).
4. Emission reduction = k (control technology).
5. Control cost = m (emission reduction).
6. Net social benefit = control benefit - control cost.

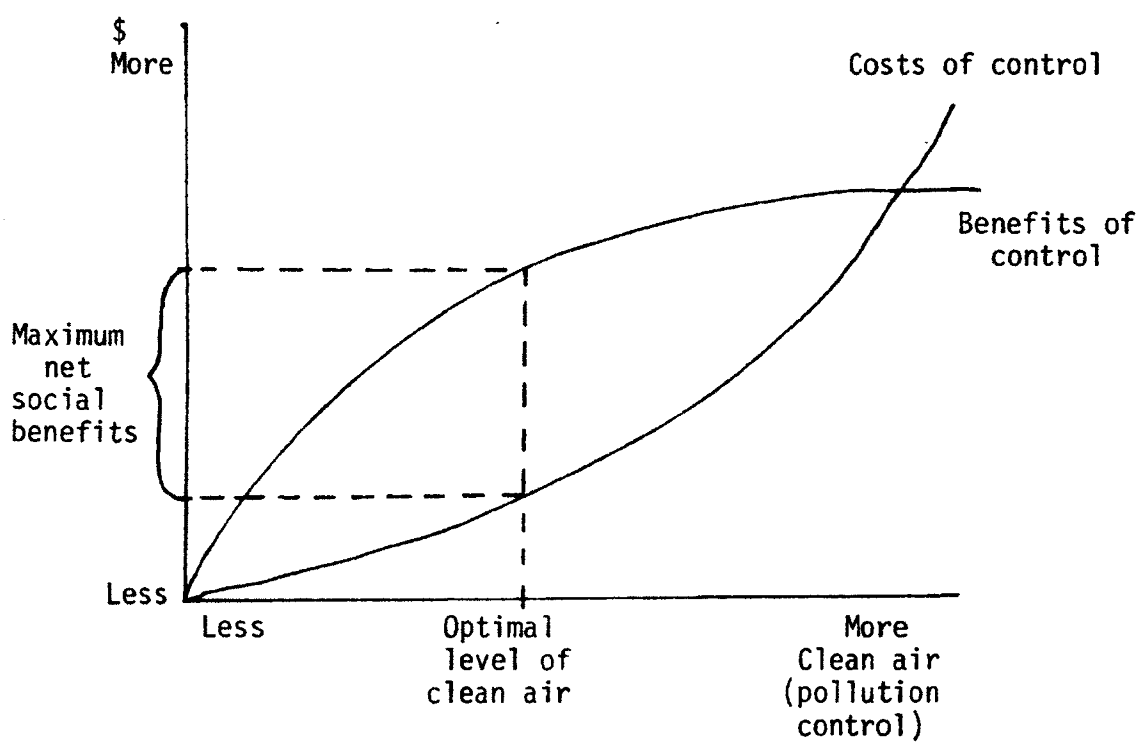

Figure 1

The Theoretical Basis for Selection of an Optimal Level of Clean Air

Figure 1 demonstrates the rationale. Associated with increasing levels of clean air (air pollution control) are increasing total benefits and increasing total costs. The optimal level of clean air results in a maximum level of net benefits to society.

Although the methodology suggested in Figure 1 is theoretically ideal for purposes of this analysis, practically it is not possible to derive continuous functions. Due to limitations in the availability of required information previously mentioned (primarily damage functions and control technology relationships), this analysis considers the benefits and costs associated with but two alternative standards, or levels of clean air, in the three Montana areas where sufficient ambient monitoring has been performed.

The alternative sulfur dioxide standards addressed are the National Ambient Air Quality Standard (NAAQS) of 80 ug/m^3 arithmetic annual average and the proposed Montana Ambient Air Quality Standard (MAAQS) of 50 ug/m^3 arithmetic annual average. The three areas examined are Anaconda, Billings, and Helena. These account for 96 percent of the point source sulfur dioxide emissions in Montana (Gelhaus et al. 1978). All three are federally designated noncompliance areas, i.e., they currently have ambient sulfur dioxide concentrations in excess of the NAAQS standard (80 ug/m^3).

Whereas the analytical techniques necessary to estimate control benefits are available from the literature, an analysis of Montana specific control costs require an engineering component that is beyond this analysis. Because of the extremely limited information available regarding the control techniques applicable to Montana's industrial sulfur dioxide sources, the primary focus of this analysis is the quantification of benefits resulting from abatement of sulfur dioxide.

Chapter 2 provides estimates of the health, vegetation, materials, and aesthetic benefits resulting from compliance with the NAAQS and proposed MAAQS standards. Chapter 3 provides estimates of the control costs associated with each standard. Benefits and costs are compared and the resulting policy implications are examined in Chapter 4.

Chapter 2

CONTROL BENEFITS

The economic effects of sulfur dioxide air pollution can be grouped into four categories: (1) health, (2) vegetation, (3) materials, and (4) amenities.

Health Benefits

Numerous studies have investigated the correlation between ambient sulfur dioxide and mortality (death) and morbidity (illness). Among the most notable are those by Gregor (1977), Lave and Seskin (1977), Liu and Yu (1976), and Mendolsohn and Orcutt (1978). The most extensive of these is by Lave and Seskin (1977). This recent publication summarizes 10 years of work during which time the authors explored the statistical relationship of ambient pollution levels and total mortality. The basic method was a linear regression analysis applied to epidemiological, demographic, and socioeconomic data for a large number of cities in the United States for the years 1960, 1961, and 1969.

Robert Haveman's (1979, p. 141) review of the Lave and Seskin analysis emphasizes the extensiveness of their methodology.

> The catalogue of variations is impressive: a principal components analysis is used to select air pollution variables for inclusion in the regression; a jackknife analysis is employed to test the sensitivity of the results to extreme observations; the analyses are replicated on 1961 and 1969 data; specifications involving quadratic

terms, a linear spline, and a split sample are attempted; additional variables (e.g., region, population, change, occupation, climate, home heating characteristics) are added; interactions among the air pollution variables and between them and the socioeconomic variables are examined; more refined mortality rates (infant, age-sex-race adjusted, age-sex-race specific, and disease specific rates) are substituted for the total mortality rate; a test of the contribution of omitted variables (by substituting suicide, crime, and venereal disease rates for mortality rates to determine if the air pollution variables are serving as surrogates for omitted variables in explaining these clearly non-pollution-related social ills) is made; the residuals are analyzed; a cross-legged, cross-sectional test for uncontrolled variables is performed; and additional air pollution variables are added and the interaction examined.

In light of that extensiveness, which includes a benefit-cost calculation on the economic wisdom of compliance with the current NAAQS, this analysis applies the methodology and results of the Lave and Seskin (1977) study to Montana.[1]

This methodology requires four basic steps: (1) establish the relationship between the air pollution related mortality and a given level of pollution, (2) calculate the actual percentage reduction in air pollution required to realize the federal standards, (3) calculate the decrease in mortality from (1) and (2) and conservatively assume that this decrease in mortality will be accompanied by an equal decrease in morbidity,[2] and (4) compute a benefit value equal to the same percentage reduction in total health costs in the United States using the

[1]In his analysis of a copper smelter in Tacoma, Washington, Ruby (1978) estimated mortality and morbidity based on all of the above-mentioned studies except Gregor. He found that the "range of estimates from these studies is remarkably small, in light of the different data and techniques used by each study" (p. 20).

[2]Ruby's (1978) analysis, based on averages from both Nelson (1976) and Liu and Yu's (1976) morbidity functions, projected that reductions in mortality would be accompanied by greater than proportional reductions in respiratory disease, heart, and lung symptoms, and asthma attacks.

predicted percentage reduction in actual U. S. mortality and morbidity.

A similar four-step methodology was followed in order to apply this model to Montana. It was found, however, that the resulting health benefits were highly sensitive to steps (1) and (4)--the actual dose-response function and the value of the decreased health effects--thus variations in those steps were used to generate high and low estimates of health benefits.

Lave and Seskin found that, based on a 1969 sample of 69 cities, a one percent reduction in ambient sulfur dioxide air pollution (as measured by the minimum, mean, and maximum of biweekly readings) resulted in a .0216 percent to .0548 percent reduction in total mortality rates depending on the inclusion of other pollution variables. When the mortality rates were adjusted to account for different age-race-sex proportions,[3] the resulting elasticities[4] were .0099 percent to .0395 percent. Although the unadjusted elasticities originated from equations statistically superior to the adjusted elasticities, the adjusted elasticities originated from a more theoretically valid regression.

In this analysis an average of the unadjusted elasticities and Montana's mortality rate is used to derive a high estimate of health benefits. An average of the adjusted elasticities and an age-race

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[3]The adjustment procedure allows for correction of variation in mortality rates caused by differing age-race-sex proportions in the population, e.g., the mortality rate of Miami would be adjusted to take into account a high population of elderly people.

[4]By definition: the percent change in mortality for a one percent change in pollution.

adjusted Montana mortality rate is used to generate the low estimates.

These elasticities correspond to linear coefficients that measure the response of mortality rates to a one ug/m^3 decrease in ambient sulfur dioxide. This linear relationship is used in this analysis because the actual statistical estimate is linear and because the alternative of assuming constant elasticity results in an overestimate for values below the mean and an underestimate for values above the sample mean.[5]

Appendix A provides the Lave and Seskin equations, the derivation of the two linear coefficients, and the calculated unadjusted and adjusted Montana mortality rates. When converted to a linear coefficient, the average of the unadjusted elasticities correspond to a linear relationship such that a one ug/m^3 decrease in ambient sulfur dioxide results in a 1.01 unit change in the total mortality rate (deaths per 100,000 population). The average adjusted elasticity results in a .65 linear coefficient. Montana's unadjusted and adjusted mortality rates were 968.16 and 981.34 deaths per 100,000 population, respectively.

The second part of the analysis is the identification of the reduction in ambient pollution that results from compliance with a given pollution standard. To achieve the standards, the major point sources in Anaconda, Billings, and Helena will be required to modify or reduce their emissions significantly. Table 1 lists the current

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[5]It was found, for example, that the benefits of pollution reduction in Billings to a 50 ug/m^3 level are generally overestimated by about 30 percent if the Lave and Seskin assumption of constant elasticity (rather than linearity) is followed.

Table 1

Reduction in Sulfur Dioxide Ambient Pollution Levels Resulting from an 80 ug/m^3 or 50 ug/m^3 Standard

Site	Mean SO_2*	Resulting mean SO_2 change	
		80 ug/m^3	50 ug/m^3
Anaconda			
Post Office	31.5 ug/m^3	7.0 ug/m^3	14.3 ug/m^3
Highway Junction	108.7	28.7	58.7
Billings			
N. 27th	5.7	.0	.0
Lockwood	77.2	9.9	27.2
CENEX Farm	148.7	68.7	98.7
Helena			
Cogswell	17.2	3.0	6.4
East Stack	37.2	7.9	17.0
Microwave	31.5	6.5	14.0
Saddle Mountain	34.3	7.1	15.5
Broudy Farm	105.8	25.8	55.8

*The ambient sulfur dioxide data are the official Montana State Air Quality Bureau readings for 1977 as reported in Gelhaus et al. (1978).

stations and most recent mean ambient sulfur dioxide readings.

In absence of accurate dispersion modeling in the areas examined, a uniform emission rollback technique was applied to each area to estimate the ambient conditions resulting from compliance to the alternative standards. Appendix B provides the derivation of the reductions in ambient sulfur dioxide concentrations listed in Table 1. The uniform emission rollback technique assumes a linear relationship between emissions and ambient concentrations; it takes into account background concentrations of ambient pollutants. This approach is a common one in the literature--even diffusion models are linear in emissions. The estimated reductions in ambient sulfur dioxide resulting from compliance to alternative ambient standards of 80 ug/m^3 and 50 ug/m^3 are listed in Table 1.

The third step in the analysis is to calculate the percentage reduction in Montana mortality resulting from compliance with the alternative standards. In Anaconda, for example, compliance with the 80 ug/m^3 standard results in a decrease of 7.0 ug/m^3 at the Post Office site (see Table 1). For the high estimate this results in 7.07 fewer deaths per 100,000 population (7.0 x 1.01), a .72 percent reduction in the unadjusted Montana mortality rate (7.07/968.10 x 100). A similar calculation at the other Anaconda monitoring site (Highway Junction) results in a 2.99 percent reduction. Based on these two sites, the population-weighted mean reduction in total mortality in Anaconda associated with an 80 ug/m^3 sulfur dioxide standard is .79 percent. The low estimate is calculated in the same manner. Similar calculations can be repeated for Billings and Helena for both 80 ug/m^3 and 50 ug/m^3 standards. The results are shown in Table 2. These percents (following Lave and Seskin) are assumed to apply to both mortality *and morbidity*.

The fourth and last analysis step is to assign values to the reduced health effects. Lave and Seskin use Cooper and Rice (1976) estimates of national health costs of $75.2 billion (1972 dollars) for direct expenditures (both mortality and morbidity) and $99.7 billion for indirect expenditures (both mortality and morbidity). The direct expenditures are such things as medical and funeral costs. The indirect

Table 2

Estimated Reduction in Mortality Associated
With Reduced Sulfur Dioxide Pollution

Site	Population* and percent city by site	Percent reduction in mortality			
		80 ug/m³		50 ug/m³	
		High	Low	High	Low
Anaconda					
Post Office	9,224 = 97%	.72	.46	1.49	.95
Hwy. Junction	285 = 3%	2.99	.90	6.12	3.89
Population Weighted Av.		.79	.51	1.63	1.04
Billings					
N. 27th	60,226 = 80%	.0	.0	.0	.0
Lockwood	7,528 = 10%	1.03	.66	2.84	1.80
CENEX Farm	7,528 = 10%	7.17	4.55	10.30	6.54
Population Weighted Av.		.82	.52	1.31	.83
Helena					
Cogswell	25,820 = 91%	.31	.20	.67	.42
East Stack	1,986 = 7%	.82	.52	1.77	1.13
Microwave	284 = 1%	.68	.43	1.46	.93
Saddle Mtn.	142 = .5%	.74	.47	1.62	1.02
Broudy Farm	142 = .5%	2.69	1.71	5.82	3.70
Population Weighted Av.		.37	.23	.78	.50

*The population data used is a 1978 projection by the Department of Community Affairs (1977). The total area population has been apportioned to the monitoring sites in the area.

are strictly foregone earnings. Of the $99.7 billion indirect expenditures, $42 billion were related to morbidity. The remainder ($55.7 billion) were estimates of foregone earnings from premature death. Lave and Seskin merely adjusted the values to account for inflation and real price increases (price increases in excess of the general rate of inflation) to arrive at their estimates of $154.4 billion (1978 dollars) direct expenditures and $180.8 billion indirect expenditures. Of the $180.8 billion indirect expenditures, $76.3 billion are foregone earnings due to morbidity. The remainder, $104.6, is attributed to foregone earnings due to premature death. This approach to valuing life is that taken in court cases involving compensation. It essentially views a person's value as capital equipment. The problem is that this approach does not reveal how the person valued his life, only what his earnings would have been. There is no necessary connection. A starving artist may value his life more highly than a wealthy industrialist.

The alternative is an implicit value approach based on wage differentials as a function of voluntary occupational risks. This approach is, in other words, an empirical measure of how much people have to be paid in order to face a higher probability of death in their work. The situation is closely analogous to that in which pollution increases the probability of disease or death; the objective is to estimate the value to people of reducing this probability a certain amount. It is more theoretically valid than foregone earnings, but it is still probably a low estimate due to the lower than average risk aversion of the occupational sample.

Lave and Seskin's average foregone earnings value per death, corrected to 1978 dollars, is $51,790. The wage differential approach (Thaler and Rosen 1976) places a value of around $400,000/death (in 1978 dollars). The Lave and Seskin estimate of indirect mortality costs is $104.6 billion (1978 dollars). An alternative estimate, based on a wage differential approach, is $806.3 billion. The latter value is used in this analysis to derive the high estimate; the former, the low estimate.

Lave and Seskin's estimate of direct expenditures and indirect morbidity costs used with the Thaler-Rosen mortality estimate results in a total health cost of $1,037.0 billion.[6] In per capita terms this is $4,823 per death. Based on the monitoring averages at Helena, Billings, and Anaconda[7] and that these are urban areas, it is assumed that the national average may be reasonably applied.

The .79 percent reduction in morbidity and mortality costs in Anaconda associated with an 80 ug/m^3 standard is therefore predicted to result in a $38.10 reduction in health costs (.79 percent of $4,823) per capita. For a population of 9,509 this is an annual health benefit in Anaconda of $362,000 associated with achieving an 80 ug/m^3 annual standard as reported in Table 3. Other entries in Table 3 are similarly derived. Benefits are reported for reduction to the federal primary standard (80 ug/m^3) and reduction to the state 50 ug/m^3 standard.

Evident from the table is the sensitivity of the estimates to both the linear coefficient and the value attached to a premature death. For example, the $362,000 benefits derived for Anaconda are reduced 39 percent (to $221,000) by using the adjusted coefficient and mortality. It is reduced 64 percent (to $131,000) by using the foregone earnings

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[6]Lave and Seskin's estimates of indirect morbidity costs ($76.3 billion) and total direct expenditures ($154.4), plus a revised mortality loss due to 2.0 million deaths, is $1,037.0 billion.

[7]The population weighted mean current SO_2 concentration for Montana is 25.6 ug/m^3 (see Table 1, p. 11); the national sample mean is 33.0 ug/m^3 (see Appendix A, Table A.3).

Table 3

Estimated Annual Health Benefits Resulting from Alternative Levels of Sulfur Dioxide Air Pollution in Montana*

Site	80 ug/m³ ($1,000)		50 ug/m³ ($1,000)	
	High	Low	High	Low
Anaconda	362	75	748	153
Billings	2,977	611	4,756	978
Helena	506	103	1,067	220
TOTALS	3,846	789	6,571	1,351

*Some columns may not add due to rounding. Such is the case with all tables in this analysis.

approach to valuing human life. Use of both factors (the low estimate) reduces the annual health benefits in Anaconda by 78 percent (to $75,000).

Vegetation Benefits

Previous assessments of air pollution induced vegetation damage have been based on one of three methodologies. The most theoretically appealing one combines dispersion models or ambient monitoring with plant damage functions that relate the ambient sulfur dioxide concentrations to yield losses or leaf injury which, in turn, must be transformed into yield losses. Analyses of this type are limited due to the information required. Application of a disperson model/damage function type of methodology to this analysis requires the modeling, by a qualified meteorologist, of seven point sources and an inventory of exposed vegetation in each area modeled. Furthermore, ambient sulfur dioxide damage functions for the eight vegetation types examined are virtually non-existent. The limited knowledge at this time concerning the relationship between yield and ambient sulfur dioxide is based primarily on short-term exposure to relatively high concentrations as opposed to the chornic effects associated with annual average concentrations.

Accepting the impossibility of applying the most theoretically appealing methodology, one must fall back to a less appealing one. The most widely used method of assessing air pollution induced vegetation damage on a statewide basis is a field survey. Field surveys have been used to estimate vegetation damage since 1949. The first statewide survey was conducted in 1969 by Weidensaul and Lacasse (1970). Their survey of air pollution induced vegetation damage in Pennsylvania was followed by numerous others--primarily in the industrialized northeastern United States and California. The accuracy of a field survey critically depends on the quality of agricultural technicians trained to objectively estimate leaf injury and the associated yield losses on a field by field basis. The cost of conducting a field survey was prohibitively beyond what this analysis could afford.

Two alternatives remain. One is to assume that sulfur dioxide induced vegetation damage is negligible. The other, that adopted in this analysis, is to pursue a "best guess" based on a methodology developed by the Stanford Research Institute (SRI). The SRI analysis (Benedict et al. 1973) has provided the basis for more recent vegetation studies. The SRI study grouped SMSA counties into pollution potential classes on a scale of 0 to 5 according to the presence of sulfur dioxide emitting industries. The scientific literature was reviewed to develop damage indices for each various agricultural crop. The damage indices considered the resistance of the crop and the susceptibility of the economic portion of the plant. The product of the damage index and the pollution potential yielded a countywide total crop damage estimate.

Although it is a landmark study, the SRI methodology has some serious limitations. The following discussion summarizes the criticisms raised by others. Waddell (1974, p. 99) argues that the SRI's damage index factors are "best guesses" and "reflect a lower bound of the true plant-associated losses due to air pollution." Jansson (1975) observed that some plumes exceed county boundaries while other point sources affect only a small portion of a large county. Hershaft et al.'s (1976) review points out such shortcomings as the failure to incorporate other uncontrolled factors (climatic conditions) and the failure to account for the indirect loss occurring because of the substitution of more resistant for less resistant but more production crops. Heck (1976) pointed out that a major shortcoming of such vegetation studies is the low estimation of yield loss when the absence of visible damage is taken to mean the absence of physical damage.

Leung et al. (1977, p. 88) report, in regard to the SRI study, that a

> number of approximations and subjective judgments were used in this method of loss estimation . . . [and] there is no way presently to test the accuracy of the SRI loss estimates. With so many unverified assumptions in the method the values obtained may not be presumed to have great accuracy.

Liu and Yu (1976) reiterate the shortcomings of the SRI study but, using it as a data base, they derive some vegetation damage functions by including climatic variables in a multiple regression analysis. The only crop with a statistically significant sulfur dioxide explanatory variable is soybeans. In regard to the Liu and Yu (1976) study, Leung et al. (1977, p. 89) commented:

> It appeared that the model worked well for oxidant damage to vegetation . . . [but] . . . other variables (sulfur dioxide) appear to be non-significant and have high variances. The greatest deficiency of the model is the use of an objective standard against which the calculated estimate can be compared.

On the positive side, a comparison of the original SRI results with published field surveys indicates general consistency. Waddell (1974, p. 99) concludes his review of comprehensive vegetation damage techniques by stating that

> the loss estimate . . . generated in the SRI study for 1964 is the most defensible of those reviewed. Also, the SRI estimates are consistent with the individual estimates for California, Pennsylvania and elsewhere.

Leung et al. (1977, p. 96) conclude their review of damage estimation techniques: "It may be necessary or desirable to continue using less objective models for assessments and analyses until more objective methods are available." For purposes of this analysis the SRI model, modified where possible according to the availability of knowledge and data, is used to generate a "best" vegetation damage estimate.

Table 4 lists the pollution potential class of the Montana counties within the analysis area. The original SRI approach assigned counties to classes based on the number or type of sulfur dioxide sources in a county. While this may have been reasonable during an era of relatively uncontrolled emissions, it is clear that it ignores the effect of existing pollution control equipment on pollution potential. The pollution potential classes therefore have been reassigned

Table 4

Pollution Potential Class for Each County

County	Present sulfur dioxide tons/year*	Class: Present	Class: 80 ug/m³	Class: 50 ug/m³
Anaconda Deer Lodge) Silver Bow)	281,750	5	3	3
Billings Yellowstone	34,850	3	3	2
Helena Lewis & Clark†	14,000	3	2	2

*Gelhaus et al. (1978).

†SRI criteria would include Lewis and Clark County as Class 5 but, considering existing pollution controls, it was given an initial Class 3 standing.

under the alternative standards to reflect the lower level of sulfur dioxide emissions.

One of the primary difficulties with this approach is evident in the table. Because the five classes are necessarily broad, the changes in pollutant emissions necessary to achieve compliance with different standards cannot be fully differentiated. As explained in Chapter 3 the changes in emissions necessary to meet the different standards vary dramatically from source to source. Furthermore, even compliance with the standard does not reflect the magnitude of the effect of a source. Large, high-level sources may affect vegetation over a much larger area even though they comply with the standard at the present monitors. Smaller, low level sources can violate the standards, but they do not affect a very large area. In the final analysis the class assignment is subjective.

Table 5 lists the exposed Montana crops and their corresponding damage functions in terms of a multiplier for each pollution potential class. Although the SRI multipliers remain unchanged, several

Table 5

Crop Damage Multipliers for Exposed Crops

Crop	Damage class	Pollution potential class				
		1	2	3	4	5
Alfalfa	A	.002	.005	.015	.040	.120
Barley	B	.000	.002	.007	.022	.064
Beans	B	.000	.002	.007	.022	.064
Hay	C	.000	.000	.003	.008	.025
Oats	B	.000	.002	.007	.022	.064
Sugar beets	C	.000	.000	.003	.008	.025
Timber	A	.002	.005	.015	.040	.120
Wheat	A	.002	.005	.015	.040	.120

crop damage classes were modified to reflect more recent information as reported by Guderian (1977), Materna (1973), and EPA (1973). Appendix C provides a terse summary of these reports and the rationale for revising several of the damage classes. The multipliers represent the proportion of the county crop value lost annually due to sulfur dioxide air pollution. For example, the damage multiplier for alfalfa in a pollution potential Class 3 county is .015. This implies that the alfalfa in the

Table 6

Estimated Crop Valuation and Economic Damage
Resulting from Alternative Levels of
Sulfur Dioxide Air Pollution
in Montana

County crop	Value ($1,000)	Loss ($1,000) Present	80 ug/m^3	50 ug/m^3
Deer Lodge) Silver Bow)		664	83	83
Alfalfa	786	94	12	12
Barley	116	7	1	1
Hay	749	19	2	2
Oats	42	1	0	0
Timber	4,445	533	67	67
Wheat	74	9	0	1
Yellowstone		253	253	74
Alfalfa	2,164	32	32	11
Barley	2,252	16	16	5
Beans	280	2	2	1
Hay	424	1	1	0
Oats	280	2	2	1
Sugar beets	8,684	26	26	0
Timber	305	5	5	2
Wheat	11,296	169	169	56
Lewis & Clark		286	93	93
Alfalfa	2,783	42	14	14
Barley	583	4	1	1
Hay	1,497	4	0	0
Oats	687	5	1	1
Timber	14,334	215	72	72
Wheat	1,026	15	5	5
TOTALS		1,203	430	251

county will suffer an *average* of 1.5 percent yield reduction, i.e., while some areas near the point source may suffer yield losses as high as 100 percent, the majority of the county will not be affected.

Table 6 summarizes the application of the modified SRI model to Montana. The crop values listed are Montana Department of Agriculture (1976) estimates for 1974 and 1975 which have been averaged. The value of timber in each county is in terms of net annual growth, i.e., gross growth less mortality. Schweitzer et al. (1975, p. 12) have characterized Montana's forests as,

> although some land is highly productive, most is in the middle or lower end of the range of growth potential. In productivity, Montana's forested land compares favorably with the Rocky Mountain region and the United States as a whole.

They estimate a current gross annual growth rate of 75 cubic feet per acre. Green and Setzer (1974) estimate an annual mortality rate for the northern Rocky Mountain states of 11 cubic feet per acre. These figures suggest that the current productivity of Montana's forest resources is approximately 63 cubic feet per acre.

Green and Setzer (1974) use a cubic foot (CF) to board foot (BF) conversion factor of 3.7 BF to 1 CF. Current research within the School of Forestry, University of Montana, indicates a Montana specific conversion factor of 3.12 BF to 1 CF.[8] The conversion factor yields an estimated net annual growth of Montana's forest resources of 196.6 BF per acre. Weighting current stumpage values[9] by species according to

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[8]Statement by Alan McQuillan, research associate, University of Montana, personal communication, July 1978.

[9]Because of the variation in this value, the previous four quarters of the Northern Region High Bid were averaged.

percent occurrence as reported in Green and Setzer, yields a stumpage of $64.7 per thousand board feet (MBF). Coupled with the growth rate, the resulting value per acre of Montana's forests is $12.72 per acre per year. This value, multiplied by the number of acres per county, yields estimates of the annual value of the forest resources in each area.

Table 7

Estimated Annual Value of Timber Per Analyzed County

County	Timbered lands (1,000 acres)			Value per year ($1,000)
	Private*	National forest†	Total	
Deer Lodge) Silver Bow)	51	298	349	4,445
Yellowstone	24	0	24	305
Lewis & Clark	129	999‡	1,127	14,344

*Acres of private timber per county were obtained from the Property Assessment Division of the Montana Department of Revenue (V. Bickford, personal communication, 1978).

†The National Forest Acreage per county was obtained from the Forest Service Northern Region (J. Baker, personal communication, 1978).

‡Includes three acres of state-owned forest.

Table 7 lists the counties, acres of private and national forest timber, and the estimated value in terms of net annual growth. This method of estimating the annual value of Montana's forest resources in each county rests on a commercial harvest valuation of noncommercial timber. It assumes that the annual growth value is an adequate proxy for the nonharvest uses of forest land such as wilderness recreation.

As shown in Table 6, application of the modified SRI model to Montana yields a statewide annual estimate of sulfur dioxide induced agricultural damage of $1.20 million per year under present conditions. This estimate may be compared to other state specific and national estimates. Table 8 lists some statewide field surveys. As the table illustrates, estimates for the same state vary widely. The key

Table 8

Previous Statewide Field Surveys

Author	Year of survey	State	Economic loss (millions 1978 $)
Weidensaul and Lacasse (1970)	1969	Pennsylvania	6.4
Lacasse (1971)	1970-1971	Pennsylvania	.3
Millecan (1971)	1970	California	44.8
Feliciano (1972)	1971	New Jersey	1.9
Pell (1973)	1972	New Jersey	.2
Naegle et al. (1972)	1971-1972	New England	1.6
Millecan (1976)	1974	California	72.2

variables are air pollution, crop value, and environmental conditions; the latter are uncontrolled and could cause substantial variation. Most of the loss reported in this table is attributed to oxidants.[10]

National assessment of sulfur dioxide-induced damage has been

[10]Usually the estimated economic loss was only broken down into pollutant categories, qualitatively.

performed only by Benedict et al. (1973). Other assessments of sulfur dioxide damage have been adjustments of their results to reflect inflation. The SRI study found that 12 percent of air pollution induced vegetation damage is due to sulfur dioxide. The SRI national survey estimated an annual loss of .44 percent of the total crop value--.05 percent to sulfur dioxide. Application of the modified SRI model to Montana yields a vegetation loss of .06 percent[11] which is slightly larger than the original SRI analysis despite the major reductions in sulfur dioxide pollution that has occurred since the original analysis. Jansson (1975) suggests several factors which could account for this.

1. Recent studies have shown that crop yields are reduced more seriously than previous leaf injury studies have indicated.

2. These same studies show that the damages resulting from chronic exposure to lower levels of air pollution are often greater than once thought.

3. Recent measurements indicate that air pollution levels, particularly oxidants, are substantially higher in rural areas than was thought previously.

In addition to these factors, the SRI analysis considered primarily highly urbanized areas where the major sulfur dioxide sources are relatively isolated from the fertile farming areas. This is in direct contrast to the situation in Montana.

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[11]This is the total nontimber damage estimate ($450,000) as a percent of the total Montana nontimber crop value ($706 million) as necessitated for comparison reasons.

Several studies have dealt with sulfur dioxide induced vegetation damage in Montana. Asmus and Stroup (1972) looked at the effects of sulfur dioxide from ASARCO's lead smelter in Lewis and Clark County. They combined a dispersion model with a leaf damage function to obtain estimates of economic loss from yield reductions in alfalfa and barley. Whereas the modified SRI application yields values of $41,745 and $4,081 for alfalfa and barley, Asmus and Stroup estimated $3,927 and $1,191 (converted to 1978 dollars) or about 10 percent and 30 percent of the SRI estimates. The Asmus and Stroup analysis was also conducted prior to an installation of an acid plant which reduced the smelter's sulfur dioxide emissions from an estimated 80,000 tons per year to its current rate of 14,000 tons per year.

Asmus and Stroup mentioned several factors which "almost certainly bias downwards" (1972, p. 26) their estimates. One such is the inability of their long-term dispersion model to pick up short-term variations in sulfur dioxide concentration. Another was their inability to account for certain climatic conditions such as presence of inversions. They also limited exposure time to daylight hours and the leaf damage/yield loss function, a key component in their analysis, is a subject of much debate in the literature.[12] Whether these limitations are substantial enough to account for a difference of at least tenfold in estimates generated by a more theoretically valid methodology is not

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[12]Asmus and Stroup (1972) use a leaf damage elasticity of .302 for alfalfa and an average elasticity for barley of .168. For alfalfa this implies that a 30 percent leaf injury would result in a nine percent yield loss. In contrast, Millecan (1976) found cases where 30 to 40 percent leaf injury resulted in 100 percent yield loss.

known. Regardless, the presence of a potentially wide error margin is evident.

In 1974 the Forest Service conducted a field survey near the Missoula Hoerner Waldorf pulp and paper mill. The report (Carlson et al. 1974) concludes that the potential exists for a decrease in timber yield of 2.5 MMBF of Douglas Fir if fumigation persists. Assuming this loss will occur over a five- to 10-year period,[13] this could result in an annual loss of between $16,000 and $32,000 from this one source. Application of the modified SRI model to Missoula County (a pollution potential of Class 1) yields an estimated sulfur dioxide timber damage of $28,000 per year. This is reasonable because the Hoerner Waldorf mill accounts for only 75 percent of the sulfur dioxide point sources in the county (Gelhaus et al. 1978).

The Forest Service has also monitored sulfur dioxide damage to forests in Deer Lodge County (Scheffer and Hedgcock, 1955) and Lewis and Clark County (Carlson, 1978). Although estimates of yield losses were not made, Carlson's investigation in March 1978 revealed that

> 4 miles south of the ASARCO smelter, 75-80 percent of the pines had nearly totally necrotic needles on the side facing the smelter. Nearly 100 percent of the needles, all ages, were dead for 50-75 percent of their length (Carlson 1978, p. 1).

Scheffer and Hedgcock (1955) summarized a 1911 survey that documented injury to subalpine fir 22 miles from the smelter. The survey found tip necrosis in areas that now are bare and badly eroded.

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[13]A personal interview with Clint Carlson, Forest Insect and Disease Management, U.S. Forest Service, Missoula, Montana, April, 1979, revealed that 10 years is a conservative assumption; the yield loss may (have) occur(red) within a four- to five-year period.

Sulfur dioxide induced injury to vegetation in the Billings area has been documented by the Environmental Studies Laboratory (EVST) at the University of Montana (EPA 1978). Monitoring of damage at a site near the Corrett power plant in 1977 revealed a high incidence of tip burn and low needle retention in Ponderosa Pine. The EVST has also done extensive monitoring in the Anaconda and Colstrip areas. Sulfur dioxide induced damage to native grasses was found at numerous sites in both areas (EPA 1978).

Based on these studies it appears safe to conclude that vegetation damage in Montana, due to ambient sulfur dioxide, is not negligible and that the modified SRI model can be used to generate a best estimate of the damage of the alternative concentrations of ambient sulfur dioxide addressed in this analysis.

The SRI study estimated sulfur dioxide induced damage to ornamental plants (private and public such as public gardens and roadside trees) was equal to about one half the crop loss. Their estimation of ornamentals is probably an underestimate; they used replacement costs as the approximation for the value of private ornamental plants and maintenance costs for the value of public ornamental plants. Considering the generally rural status of Montana, however, damage to ornamental plants is assumed to be but one fourth the crop damage. The inclusion of ornamental damage using one half of the SRI national estimate of average damage to ornamental (or one fourth of crop damage) results in total annual vegetation losses of $1.50 million in Montana under present conditions.

Table 9 lists the counties and corresponding vegetation losses

Table 9

Estimated Vegetation Losses and Benefits Resulting from Sulfur Dioxide Air Pollution in Montana*

County	Annual loss ($1,000)			Annual benefit ($1,000)	
	Current	80 ug/m³	50 ug/m³	80 ug/m³	50 ug/m³
Deer Lodge) Silver Bow)	830	104	104	728	726
Yellowstone	317	317	93	0	224
Lewis & Clark	357	117	117	240	240
TOTALS	1,504	537	313	966	1,190

*Includes the ornamental damage estimate.

for present conditions and alternatives of the federal annual standard and the state annual standard. The estimated benefits of avoided vegetation losses, going from present conditions to an 80 ug/m^3 annual standard, are $966,000 annually; to a 50 ug/m^3 annual standard the benefits are $1,190,000 annually.

The estimated vegetation damage is a "best" estimate; however, several major shortcomings warrant discussion. The absence of dispersion models for each point source results in crude approximations of exposure to the pollutant. The use of county boundaries to define an exposed population is obviously not an accurate portrayal of the exposure area. Sulfur dioxide plumes have particularly been tracked over several hundred miles from their source (Wilson et al. 1977). Silver Bow County has been included with Deer Lodge County for this reason.

Also, the broad categorization of the countys into pollution potential classes does not allow for a fully differentiated benefit associated with compliance to a 50 ug/m^3 standard. This results in an underestimation of the incremental benefits in the Anaconda and Helena areas and, possibly, an overestimation in the Billings area.

An additional point that warrants discussion is the potential significance of a vegetation response lag. Because ecosystems require a long time to recover from past pollution damage, it could thus be argued that the benefits from reduced pollution should not be counted until two, 10, or even 40 years in the future when the full benefits will begin to be received. Under the accepted discounting procedures this could substantially reduce the estimated benefits of reduced timber damage, e.g., a 10 year response lag at a six percent discount rate[14] would reduce the estimated timber benefits by one half. A 40 year response lag would reduce them to one tenth of the estimated amount. With agricultural crops this effect is not as significant because they are harvested annually.

The appropriateness of discounting future benefits concerns intergenerational equity. Although a detailed discussion of this phenomenon is beyond this analysis, the following example provides a brief description of the problem.[15] A generation will reap the benefits of a pollution product (e.g., copper) in light of a heavily discounted cost (e.g., devastated alpine environment). A second generation (ours)

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[14]See Appendix D for a discussion of discount rates.

[15]For a more detailed discussion of intergenerational equity see Ferejohn and Page (1978) or Marglin (1963)

suffers the damage, but argues that the discounted benefits of cleaning up are too small. The result is a third generation stuck with a depleted and poisoned environment.

The response of vegetation to reductions in ambient sulfur dioxide is a question of soil chemistry and plant pathology. A soil and field analysis of each area to approximate a response time is beyond this analysis. It can be argued, however, that this possible upward bias in the benefit estimates is sufficiently countered by, probably, a substantial downward bias. With the exception of nonharvestable timber and ornamental plants, the estimation of vegetation benefits fails to account for nonmarket values associated with vegetation such as lichens, mosses, ferns, etc. Williams (1978, p. 9) reports that

> The threat to lower plant species--mosses, ferns, lichens, and liverworts--appears to be particularly severe. Field studies in areas where monitoring for SO_2 has been extensive have defined the ultrasensitivity of this group of organisms. Chronic responses observed in the field suggest that many lower plant species are damaged by SO_2 levels as low as 0.012 ppm [30 ug/m^3]. Injury to lower plant floras can result in watershed and aquifer degradation, accelerated soil erosion, reduced soil fertility, reduced water quality, and additional water stress to organisms more tolerant of sulfur pollutants.

The social disutility associated with these unaccounted for damages surely accounts for enough downward bias to control for any possible upward bias introduced by not considering a possible lag between the time of sulfur dioxide abatement and realization of vegetation benefits.

Material Benefits

Sulfur dioxide reacts with various materials in the presence of moisture. This causes damage which results in an economic loss by a reduction in the useful life of the material. Studies have examined sulfur dioxide induced (most likely sulfuric acid) damage to steel, zinc, aluminum, copper, nickel, titanium, cotton, limestone, marble, paint, cellulosic fibers, and electrical components. Critical reviews of these studies have been published by Gillette and Upham (1973), Liu and Yu (1976), Waddell (1974), and Yocum and Grappone (1976). Generalized conclusions from the reviews suggest the following:

1. Relative humidity and sulfur dioxide are the key variables.

2. Sulfur dioxide damage occurs primarily to those materials continually exposed to the outdoor environment.

3. Concrete, marble, and limestone are negligibly affected by current levels of sulfur dioxide.

4. Most sulfur dioxide induced economic loss is due to corrosion of metals (both ferrous and nonferrous) and deterioration of paints.

Gillette (1975) developed a model for estimating the economic loss induced by sulfur dioxide on galvanized, painted, and other non-coated metals. Gillette converted a zinc (galvanized coating) corrosion function developed by Haynie and Upham (1970) into an economic damage function.

$$E = \frac{(RC)\ .00103\ (RH - 49)\ SO_2}{TH} - \frac{RC}{UL} \qquad [1]$$

where: E = economic loss per capita

RH = relative humidity

SO_2 = ug/m^3 ambient sulfur dioxide concentration

RC = replacement cost of the metal

TH = thickness of the galvanized coating

UL = useful life of the metal.

In the case where E is a negative value, there is no economic loss due to reduced service life. In this case E is set equal to zero. Values for TH, UL, and RC were derived by Gillette for five different categories of metals from industrial indices. They are listed in Table 10. The area's population, assuming that the geographic distribution of materials is similar to that of the population, is used as a proxy for material exposure to sulfur dioxide.

Gillette extimated damage to other metals by utilizing data generated by Battelle Memorial Institute (Fink et al. 1971). By assuming a linear relation between sulfur dioxide levels and damage costs,

Table 10

Product Characteristics and Costs of Major Types of Galvanized Products*

Galvanized product	Dollars per capita		UL	TH
	RC replacement values	RC/UL annual replacement costs	ordinary use life, years	thickness of zinc coating, um
Building accessories	123.75	2.75	45	26
Wire fencing	20.00	1.00	20	26
Fence posts	27.50	.92	30	54
Wires and gables	30.00	1.50	20	37
Pole line hardware	20.00	.67	30	85

*The dollar estimates are 1972 dollars.

Gillette proposed a damage function with a threshold of 20 ug/m^3.

$$E = (SO_2 - 20)\ .88 \qquad [2]$$

This function proposes a per capital economic loss estimate of 88 cents for each additional ug/m^3 of ambient sulfur dioxide in excess of 20 ug/m^3.

The key assumption behind its use is that relative humidity is constant among different areas. Since Montana's relative humidity is, in general much less than the national averages; use of this function results in an overestimation of corrosion damage to other metals. It is also not clear whether this function considers only corrosion or whether it also considers soiling. The former is attributed to sulfur dioxide and the latter is considered primarily an effect of total suspended particulate pollution.[16] An alternative approach is to follow Salmon (1970) who found that, on a national scale, damage to galvanized metals was about one half the economic loss attributed to corrosion of all metals.[17] Multiplying the Haynie and Upham zinc corrosion function by a factor of two will take into account the corrosion of all metals. Relative humidity is also accounted for since the zinc function considers it as a variable.

The economic loss ascribed to paint deterioration has been estimated by numerous studies. The most notable are by Salmon (1970),

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[16]Gillette (1975) specifies that the function estimates "maintenance" costs which usually are composed of both paint damage and soiling.

[17]Salmon (1970) estimated a total metals damage of $1,502 million of which $778 million was attributed to zinc corrosion.

Spence and Haynie (1972), and Yocum and Grappone (1976). All three estimates are fairly consistent and utilize an approach that is based on observing the differences in paint life in urban versus rural areas.[18] One possible approach to estimate paint damage in Montana is to use, once again, Salmon's estimate of paint damage as a percent of his zinc damage estimate, multiplying this factor into the zinc corrosion function developed by Haynie and Upham. Salmon estimated paint damage to be 150 percent of his estimated zinc damage. Yocum and Grappone, however, found that only 30 percent of their paint damage estimate was due to sulfur dioxide; the remainder was due to suspended particulate. This suggests that a factor of .45 can be included in the zinc damage function (Equation [1]) to account for sulfur dioxide induced paint damage.[19]

Because no viable alternative exists, the above-described method for estimating paint damages will be used in this study. This procedure results, however, in an underestimated paint damage since the relationship between ambient sulfur dioxide and paint differs from that represented by the zinc corrosion function. Whereas Haynie and Upham (1970) found relatively high sulfur dioxide and relative humidity thresholds for zinc corrosion,[20] Haynie et al. (1976) found that the

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[18]Salmon (1970) estimated total paint damage for 1968 to be \$1.2 billion, Yocum and Grappone (1976) estimated 1970 damage as \$.7 billion, and Spence and Haynie (1972) estimated 1971 damage to be \$.7 billion.

[19]Thirty percent of Salmon's paint estimate as a percent of his zinc estimate (\$1,195/ 778 x . 3 = .45).

[20]The zinc corrosion function, evaluated at 60 percent relative humidity, has a 51 ug/m^3 sulfur dioxide threshold. The relative humidity threshold is 49 percent.

relationship between sulfur dioxide and oil base paint erosion was best represented by a zero threshold (for both sulfur dioxide and relative humidity) erosion function. Furthermore, latex paints contain calcium carbonate which is highly reactive with sulfur dioxide (Haynie et al. 1976).

Table 11

Estimated Material Damage Loss and Benefits Resulting from Alternative Levels of Sulfur Dioxide Air Pollution in Montana

Site	Annual loss ($1,000)			Annual benefit ($1,000)	
	Current	80 ug/m^3	50 ug/m^3	80 ug/m^3	50 ug/m^3
Anaconda	0	0	0	0	0
Billings	123	21	0	102	123
Helena	2	0	0	2	2
TOTALS	125	21	0	104	125

Table 11 shows the results of applying the Gillette zinc economic damage function (Equation [1]) modified to take into account other metals (multiplied by two) and paint damage (multiplied by 1.45). The ambient sulfur dioxide concentration and population data are as reported in Tables 1 and 2 (see pages 11 and 13). Relative humidity was obtained from the National Oceanic and Atmospheric Administration (NOAA) Local Climetological Data for 1977. Normals for quarterly averages were used rather than annual averages. This allowed for correlation between high humidity and high ambient concentrations of sulfur dioxide as occurs during certain periods of the year.[21] Since there is no available relative humidity data for Anaconda, it was assumed that Anaconda's quarterly relative humidity is five percent less than Helena's or a maximum of 55 percent. The NOAA national map of relative humidity places Anaconda between the 50 percent to 60 percent isopleths.

As evident in Table 11, the estimated material damage in Montana is largely limited to the Billings area. This is due to the greater population in Billings and the proximity of the sulfur dioxide sources to the population. The sources in Anaconda and Helena are relatively isolated from the population center. The total benefits of reduced material damage, going from current conditions to an 80 ug/m^3 annual sulfur dioxide standard, are $104,000 annually; to 50 ug/m^3 they are $125,000.

The key variable in the material damage estimate is the relative humidity. It appears that Montana's relatively low humidity[22] results in minimal amounts of damage to metals and paints. In light of the procedure for estimating paint damages, however, these estimates are probably substantial understatements of the true material losses attributed to ambient sulfur dioxide.

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[21] Since the chamber type experiment from which Haynie and Upham (1970) developed the zinc corrosion function considered an average of short-term exposures, ideally the relative humidity factor should be based on monthly or even daily averages.

[22] Montana's average humidity is about 57 percent compared to a national average of about 67 percent.

Aesthetic Benefits

Reductions in property values are often used to determine aesthetic damages induced by air pollution. These estimates will, however pick up all other damages, including health and property, to the extent that people are aware of the damage(s) being done to them and their property.

Waddell's (1974) review and national assessment of air pollution damage estimates a national aesthetic damage of $5.8 billion (in 1970 dollars), of which one half is attributed to sulfur dioxide. Estimates of effects on animals and the functioning of the ecosystem were not attempted. A principal component of quantified sulfur dioxide induced aesthetic damage is visibility impairment.

Aesthetic loss due to visibility impairment has become a major concern in our society (U.S. House of Representatives, 1977). The National Park Service estimated that the proposed Kaipotowits power plant would have resulted in a loss of $24 million in recreation income and a $78 million loss in use value in Utah over a 35-year period due to the adverse effects on visibility (U.S. Department of Interior, 1976). Similarly, Vars and Sorenson (1972) have estimated an annual loss of at least $30,000 to $45,000 due to smoke and haze in the Salem, Eugene, and Corvallis areas of Oregon. In a Florida study the social aesthetic cost of reduced visibility was estimated at $26.1 million a year. This was greater than the sulfate induced health cost also estimated for Florida (SRI, 1978).

Waggoner and Charlson (1977) present an in-depth review of the relationship between visibility and suspended particulate. An application to Montana of their findings is described below.

Visibility impairment generally occurs when liquid and solid airborne materials absorb and scatter light. This reduced visibility in a polluted area is related to the size, concentration, and physical characteristics of the particulate pollutants present in the atmosphere. The majority of light-scattering particulates in the air are sulfates.

Visibility, or meteorological range, is defined by the distance at which a totally black object has a contrast of only two percent with the horizon sky. As the distance to the object increases, gases and particles in the intervening air decrease the perceived contrast exponentially.

$$C = C_o \ e^{-bx} \qquad [3]$$

where: $C_{(o)}$ = the (initial) contrast

x = the distance

b = a scattering coefficient--a factor dependent on the amount and kind of particles in the air.

At the distance x where C is reduced to two percent from an initial C_0 of 100 percent

$$x = \frac{\ln .02}{-b} = \frac{3.9}{b} \qquad [6]$$

or, when the terms are expressed in common units,

$$L_v = \frac{2.44 \times 10^{-3}}{b(m^{-1})} \qquad [5]$$

where: L_v = the visual range in miles

m = meters.

The relationship between the mass of particles and the scattering coefficient (b) is

$$\text{mass } (\text{ug/m}^3) = (3 \times 10^5)\ b\ (\text{m}^{-1}) \rightarrow b = \frac{m}{3 \times 10^5} \qquad [6]$$

Of primary interest are the changes in the visibility due to decreases in the emission of sulfur dioxide. The sulfate portion of the atmospheric particles will closely obey the mass-scattering coefficient given above. Therefore, we can write

$$\Delta L_v = \frac{2.44 \times 10^{-3}}{b_o} - \frac{2.44 \times 10^{-3}}{b_o + \dfrac{\Delta m}{3 \times 10^5}} \qquad [7]$$

$$= \frac{L_o\ \Delta m}{732/L_o\ \Delta m}$$

or

$$\Delta m = \frac{732\ \Delta L_v}{L_o(L_o - \Delta L_v)} \qquad [8]$$

where: b_o, L_o : = initial conditions

ΔL_v, Δm : = visual range changes and ambient sulfate concentrations

Table 12

Reduction in Visibility by Addition of Sulfate to the Atmosphere

Visibility (miles)	Percentage reduction		
	1 ug/m^3	4 ug/m^3	8 ug/m^3
160	18	47	64
140	16	43	60
120	14	40	57
100	12	35	52
80	10	30	47
60	8	25	40
40	5	18	30
20	3	10	18

Table 12 shows the decrease in visibility that could occur by the addition of three concentrations of sulfate to the air (uniformly) between the viewer and an observation feature such as a mountain range. The addition of a small amount of sulfate has a large effect where visibility is already high. The added sulfate has but a small effect when visibility is presently poor.

The amount of sulfate present at any point is primarily a function of the time that has elapsed since the air mass left a source of sulfur dioxide. The sulfur dioxide converts to sulfate at a rate of 0.5 percent to 16 percent per hour depending on atmospheric conditions and the presence of catalysts and moisture. If a steady wind is blowing, it is necessary to be quite a distance downwind before there is sufficient conversion in the plume for much visibility degradation to occur. On the other hand, if there is very little wind, sulfur dioxide emissions may remain close to the source; this results in high sulfate levels and poor visibility in the immediate vicinity. With detailed knowledge of the meteorological conditions, it is possible to estimate the relationship between sulfur dioxide emissions at one point and visibility degradation at another point. Latimer et al. (1978) developed some meteorological models that do present just such an estimate; unfortunately, the necessary meteorological information has not been collected in Montana.

It is possible to make some very rough estimates for several situations in Montana to illustrate the magnitude of the effects and to permit some lower bound estimates for visibility degradation due to sulfur dioxide emissions. Latimer et al. (1978) report that the days of lower visibility at Great Falls and Billings are primarily associated with winds out of the west. The principal source of sulfur dioxide emissions to the west of Great Falls is the Bunker Hill smelter in Kellogg, Idaho. An additional important source of visibility degrading particulate might be slash fires in the Rocky Mountain forests. The Anaconda smelter and the ASARCO smelter, approximately 250 and 175 miles west of Billings, respectively, are two major sources of sulfur dioxide.

Latimer et al. (1978) calculated the reduction in visibility that is possible from the plume of a sulfur dioxide source under the meteorological conditions of the southwest. If their calculations are adjusted to the prevailing winds of the high plains, they indicate that on days with a direct, steady wind from the source to Billings, the visibility in Billings could be reduced by eight percent by sulfate from Anaconda and less than one percent by sulfate from ASARCO. Because there is more moisture in the air in Montana than in the southwest, these estimates are probably low. The estimate is consistent with the actual improvement in visibility that was observed in Billings during the 1967-1968 copper industry strike (Latimer et al. 1978). If the winds are calm, a substantial portion of the visibility degradation will be due to the sulfur dioxide sources in and near Billings.

If one assumes that compliance with either the 80 ug/m^3 or 50 ug/m^3 standards will force control of emissions by Anaconda and ASARCO to approximately 85 percent and 75 percent of the process input sulfur, respectively,[23] the above calculation implies that the sulfate level in the Billings area would be reduced by about 1 ug/m^3 on an annual average.

At approximately 65 miles average visibility,[24] with the addition of 1 ug/m^3 of sulfate uniformly across the sight path, visibility will be reduced 5.8 miles (from Equation [7]). Although there is no market mechanism to express a value of this visibility reduction, there are two techniques that can be used to ascertain consumer valuations of atmospheric visibility: (1) the bidding game model, and (2) the household substitution model. The first technique is rather direct. Individuals are asked how much they would pay for an improvement in their situation or require as a bribe to accept a decline in current fortunes. The household substitution approach aims at establishing a link between visibility and already priced market goods which affect leisure

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[23] See the discussion of control costs in Chapter 3.

[24] Montana average as reported by Latimer et al. (1978).

decisions. It tries, in other words, to measure the reallocation of time by a consumer in any particular activity when faced with different levels of visibility.

Even though the two techniques are different, they have produced similar prices for visibility in three southwestern studies. In a Farmington, New Mexico survey, Blank et al. (1977) found an annual average bid of \$3.90 per household per mile per year. This can be compared to \$3.80 per household calculated by Randall et al. (1974). Brookshire et al. (1976), in their Glenn Canyon National Recreation Area study, also found comparable values to those determined by Blank et al. although the gaming situation was substantially different.

When converted to 1978 dollars these estimates of willingness to pay for visibility improvement are within one dollar of each other (from \$4.17 to \$5.33). Although, given the similar topography, these values can be reasonably applied to Montana, in light of the assumptions required to derive the visibility improvement estimate, a willingness to pay for visibility improvement of \$3.80 per household per mile per year will be used.

Table 13 shows the resulting benefit estimate. The 5.8 mile visibility improvement resulting from a 1 ug/m^3 reduction in sulfate, valued at \$3.80 per mile per year, results in a \$730,000 benefit to the 33,200 households in the Billings area. This calculation, unfortunately, is not sufficiently fine-grained to differentiate between the 80 ug/m^3 and 50 ug/m^3 alternative standards.

This estimate may overstate the economic damage in visibility loss due to sulfur dioxide emissions from Anaconda and ASARCO since the

Table 13

Estimated Visibility Benefits Resulting from Alternative Levels of Sulfur Dioxide Air Pollution in Montana

Site	Annual benefits ($1,000)	
	80 ug/m^3 standard	50 ug/m^3 standard
Anaconda	0	0
Billings	730	730
Helena	0	0
TOTALS	730	730

winds do not blow consistently as assumed. However, this estimate does not include the reduction in sulfate levels from the Billings sources, neither does it estimate the visibility loss due to sulfur dioxide emissions in parts of the state other than Billings. It is also not clear that the $3.80 per household value fully captures the value of good visibility associated with national parks or wilderness areas, i.e., an additional mile of visibility in a wilderness area is probably of much greater value than the $3.80 per household derived from a nonwilderness environment. Therefore, this estimate can be presumed to be a substantial underestimate of the economic loss in visibility reduction due to sulfur dioxide emissions.

Aside from visibility, other effects of sulfur dioxide on amenities in Montana are difficult to quantify. Damage to ornamentals was estimated along with all other vegetation; it is assumed that damage to works of art is negligible. Possible sulfur dioxide induced damage to animals, the ecosystem as a whole, and unaccounted for aesthetics have not been quantified. The implication of not estimating the amenity/aesthetic effect of sulfur dioxide in its entirety is, of course, that the total aesthetic damage estimates are biased downward. In light of these omissions the damages estimated should be interpreted as a minimum figure.

Summary of Control Benefits

Table 14 provides a summary of the control benefits estimated in the previous four sections of this chapter. As evident from the table, the variation in the high and low health benefits results in greater than twofold variations in the total benefits of the 80 ug/m^3 and the 50 ug/m^3 standards. Whereas the total high estimates are dominated by the health benefits (68 percent and 76 percent of the total benefits of the 80 ug/m^3 and 50 ug/m^3, respectively), the low estimates are

Table 14

Summary of Annual Benefits Resulting from Alternating Levels of Ambient Sulfur Dioxide Air Pollution in Montana

Category	80 ug/m^3 ($1,000)		50 ug/m^3 ($1,000)	
	High estimate	Low estimate	High estimate	Low estimate
Health	3,846	789	6,571	1,351
Vegetation	966	966	1,190	1,190
Materials	104	104	125	125
Visibility	730	730	730	730
TOTALS	5,646	2,589	8,617	3,397

distributed relatively evenly between health, vegetation, and visibility.

The largest component of the total low estimate of benefits associated with compliance with an 80 ug/m^3 standard is vegetation; health remains the largest component of the benefits associated with the 50 ug/m^3 standard. Whereas the health benefits dominate the high estimate, they comprise only 30 percent and 40 percent of the low estimate of benefits associated with complying with an 80 ug/m^3 and 50 ug/m^3 standard.

Table 15

Summary of Area Specific Annual Benefits Resulting from Alternative Ambient Sulfur Dioxide Standards

Area	80 ug/m^3 ($1,000)		50 ug/m^3 ($1,000)	
	High	Low	High	Low
Anaconda	1,088	801	1,474	879
Billings	3,809	1,443	5,833	2,055
Helena	748	345	1,307	460
TOTALS	5,646	2,589	8,617	3,397

Table 15 provides a summary of the area specific benefits. Evident from the table is the dominance of the Billings area. This is due to the urban population and the fertile farming area surrounding the population.

Chapter 3

CONTROL COSTS

Introduction

In order to comply with either the NAAQS (80 ug/m^3) or the proposed MAAQS (50 ug/m^3) annual average sulfur dioxide standards, most major point sources of sulfur dioxide will have to reduce their emissions. Theoretically, the identification of a best standard would require control cost functions relating the level of emissions to a cost value.

Due to a combination of several factors, however, control cost functions are applicable to only highly aggregated situations. Development of accurate estimates of control costs has been inhibited by several factors. Among them are (1) lack of accurate disperson models, (2) discrete technology, (3) most point sources (especially the older ones) are unique, (4) engineering assessments are expensive, and (5) lack of incentives.

The implications of these five factors are evident from the following hypothetical example. Suppose a sulfur dioxide point source is forced to comply with the 50 ug/m^3 standard. Suppose further that the current sulfur dioxide emissions from this source are 10,000 tons per year and that an ambient monitor near the point source records an annual sulfur dioxide average of 100 ug/m^3.

The implications of the first limitation, lack of an accurate disperson model, is that it is unknown how much the 10,000 tons per year will have to be reduced to attain a reduction of 50 ug/m^3 at the ambient monitor. Discrete technology further complicates the estimate. Suppose it is found that a lime scrubber on an exhaust stack will achieve ambient sulfur dioxide reductions that are twice the amount required by the standard. One half of a lime scrubber is not feasible; the costs of complying with a 25 ug/m^3 standard must be compared with benefits of only a 50 ug/m^3.

The remaining three factors--uniqueness, engineering costs, and incentives--further increase the control cost variability. Most of the industrial point sources examined are unique and, therefore, it is not possible to compare control techniques implemented at other sources and results of those techniques with the sources examined. Each unique industrial point source requires a costly engineering assessment by qualified air pollution control engineers in order to arrive at accurate estimates of control costs.

This leads to the last factor--incentives. Unless the industrial source is forced to comply with a standard, it has no incentive to discoverthe costs associated with reducing their emissions, i.e., until a standard is enforced upon a point source the burden of proof falls not upon the source, but the receptor.

Arriving at estimates of control costs associated with the alternative annual sulfur dioxide standards is further complicated by the general wilderness of air pollution laws. There are currently 27 ambient pollutant standards (including the state's goals and guidelines) in Montana. These standards comprise 10 different pollutants (six federal and four additional state pollutants) varying from one hour to annual average concentrations. Further complicating the situation are emission laws--both federal and state--and even the definitions of air pollution and ambient air are not straightforward. In most cases (e.g., Anaconda) the limiting standard is not the annual average, but the one-hour or 24-hour standards. This causes difficulty in estimating costs associated with only the annual standard when the planned control technique is designed for compliance with a relatively more stringent short-term standard, emission standard, or even the number of times the standard can be exceeded.[1]

In light of these difficulties, rough estimates of control costs were obtained based on consultation with the Montana State Air Quality Bureau (AQB) engineers, representatives of the industrial point sources examined and EPA analyses. Intermediate values were used when the resulting information was contradictory. The reason for adopting intermediate values, rather than upper and lower estimates, is that the combination of information barely allowed for rough estimates whereas each source of information considered separately allowed for only partial estimates.

It was also felt that, given the resource constraints on this study, plant closure or production cutbacks did not warrant consideration as a viable control technique. Although this control technique is obviously effective, the probability of closure of any industrial point source examined is virtually zero. Table 16 lists the point sources and their

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[1]The significance of the allowable number of concentration violations pertains to mechanical maintenance or failure that disrupts the functioning of the control process.

Table 16

Major Sulfur Dioxide Emitting Industries in Montana

Industrial point source	Area	Sulfur dioxide emissions (tons/years)*
Anaconda Copper Smelter	Anaconda	281,750
CENEX Petroleum Refinery	Billings	10,380
Corrett Power Plant	Billings	9,986
Exxon Petroleum Refinery	Billings	9,800
CONOCO Petroleum Refinery	Billings	3,198
Montana Sulphur and Chemical Company	Billings	1,530
ASARCO Lead Smelter	Helena	14,000

*Obtained from Gelhaus et al. (1978).

emissions for each of the three analysis areas. A brief analysis of each point source follows.

Analysis of Point Sources

Anaconda Copper Smelter

Current Montana AQB rules require the Anaconda Copper Smelter to control 90 percent of the process input sulfur. A recent reconstruction of the main smelting facility has made it technically possible to achieve that level of control. At present a significant fraction of the gas stream from the main facility is treated in a sulfuric acid plant which extracts the sulfur dioxide from the exhaust gas and converts it to sulfuric acid. This results in about 34 percent overall control. Additional control could be achieved by constructon of another acid plant to treat more of the exhaust gas flow.

The Environmental Protection Agency has concluded that control of 86 percent of the process input sulfur is necessary to meet the federal 24-hour primary standard. Because of the meteorology of the Deer Lodge Valley, 86 percent control is expected to result in an annual average sulfur dioxide level below 50 ug/m^3.

To achieve 86 percent control, the next acid plant would have to be approximately 60 percent larger than the present acid plant. (The two plants would operate in parallel.) The capital cost of the new plant and the necessary duct work and stack would be approximately $21 million.

The market for sulfuric acid is very limited in Montana and neighboring states. Anaconda has already found it necessary to ship its acid product into the midwest. It may be necessary to neutralize the acid if additional markets cannot be found. The neutralization of sulfuric acid is an energy intensive operation. Depending on the amount of neutralization required, the annual operating and maintenance (O and M) costs associated with the additional acid plant would be between $2.0 and $8.5 million. For purposes of this analysis it is assumed that compliance with the 80 ug/m^3 and 50 ug/m^3 standards can be achieved with capital expenditures of $21.0 million and an annual O and M cost of $5.5 million.[2]

CENEX Petroleum Refinery

The CENEX Petroleum Refinery in Laurel is the third largest

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[2]The assumed O and M costs at the copper smelter are treated as a sensitivity parameter in the summary of control costs.

contributor to ambient sulfur dioxide in Montana. Despite having the lowest production capacity of the three Billings area refineries, use of high sulfur Wyoming crude makes CENEX the largest contributor to ambient sulfur dioxide in the Yellowstone Valley.

Several sulfur dioxide control options are available to CENEX. It has recently agreed to make numerous modifications to process equipment in order to comply with the 80 ug/m^3 standard. These modifications would cost around $8 million ($.5 million O and M). To achieve a 50 ug/m^3 standard, one available option is to raise the height of six exhaust stacks to 199 feet (they now are 90 to 100 feet high). The cost associated with increased stack heights would be roughly $2 million.

As with the Anaconda Copper Smelter, the limiting standard for CENEX is the one-hour standard. Compliance with the NAAQS one-hour standard may require both the process equipment modifications and the increased stack heights. In order to comply with the one-hour MAAQS, an available option is fuel oil desulfurization. The cost of this technique would be $80 million.

In light of the expected sulfur dioxide dispersion resulting from the higher exhaust stacks, compliance with the 80 ug/m^3 and 50 ug/m^3 standards will cost roughly $8 million and $10 million ($.5 million O and M), respectively.

Corrett Coal-fired Power Plant

The Montana Power Company's Corrett Power Plant in Billings produces about 180 MW of electrical power. It burns low sulfur subbituminous coal from the strip mines at Colstrip. About seven percent of the sulfur in the coal is presently removed by crushing and screening. This could be doubled to about 15 percent by a flotation process (Cavallaro et al. 1976). This coal-beneficiation process also loses about 5 percent of the coal heating value. It would cost about $3 million to construct a coal cleaning facility and about $.3 million per year to operate it (including the value of the lost coal) (EPA 1978b).

There is no sufficient meteorological or monitoring information to know if the Corrett plant contributes significantly to the high readings at some of the monitors in the Billings area. If additional control were necessary it would be possible to treat the exhaust gas stream in a lime scrubber to remove the sulfur dioxide. Based on the cost of a similar unit at the Colstrip power complex, the sulfur dioxide emissions could be reduced by about 30 percent for a capital expense of approximately $6.7 million and an annual operating cost of about $.5 million. Because of the constraints of the site, it may be necessary to utilize holding tanks and a sludge dewatering facility before transferring the scrubber residue to tank trucks for transport to settling ponds. If water for the scrubber must be obtained from an adjacent river, a clarifier may also be necessary to clean the water before use in the scrubber. These items could add an additional $1.2 million in capital costs (Bloom et al. 1978).

Since the Lockwood ambient monitoring site averages less than 80 ug/m^3 sulfur dioxide, the Corrett Power Plant, Exxon and CONOCO refineries, and Montana Sulphur and Chemical Company would require modifications only if the 50 ug/m^3 standard were enforced. It is conservatively assumed in this analysis that the $6.7 million lime scrubber modification ($.5 million O and M) would be required for compliance with the 50 ug/m^3 standard. It may well be the case that the $3 million coal cleaning facility, in conjunction with reductions at the other sources, would result in sufficient control of sulfur dioxide emissions.

Exxon Petroleum Refinery

The Exxon Petroleum Refinery, along with the Corrett Power Plant, is probably a significant contributor to the annual sulfur dioxide level of 77.2 ug/m^3 measured at the Lockwood School site. There is no sufficient meteorological or monitoring information to know which, if any, of the many stacks at the Exxon refinery do contribute significantly to this reading.

An approximate 30 percent decrease in sulfur dioxide emissions could be obtained by installing a wet gas scrubber on the Coker-Co boiler exhaust stream. This would require a capital expense of about $4.5 million and an annual operating cost of approximately $.2 million per year. Other means of reducing the sulfur dioxide emissions may be available to Exxon at less cost, e.g., treatment of sulfur-containing water or refinery fuel oil. Again, opting for the conservative approach, it is assumed that the wet gas scrubber, at a cost of $4.5 million, would be required by the 50 ug/m^3 standard.

CONOCO Petroleum Refinery

Despite comparative production capacity CONOCO emits one third the amount of sulfur dioxide, as do the other two refineries, primarily because of the low sulfur Canadian crude used by CONOCO. The CONOCO refinery does not appear to contribute significantly to elevated sulfur dioxide levels in the Billings area. It is not anticipated that additional controls would be required on this facility to achieve the 50 ug/m³ standard. This assumption is reasonable considering the projected 30 percent reduction in the Corrett Power Plant's and Exxon Petroleum Refinery's sulfur dioxide emissions associated with compliance with the 50 ug/m³ standard.

Montana Sulphur and Chemical Company

The Montana Sulphur and Chemical Company is essentially one big air pollution control facility. It receives sulfur-laden natural gas and hydrogen sulfide from the CONOCO and Exxon refineries. It removes sulfur from the gas and returns the clean gas to be burned in the refinery for the process heat. The hydrogen sulfide is converted to liquid hydrogen sulfide, sulfur, and other chemicals. In most refineries this plant would be an integral part of the refinery rather than a separately owned company.

Despite the sophisticated equipment, Montana Sulphur releases one of the most concentrated sulfur dioxide streams in the state. It recently decided to raise its exhaust stack to approximately 100 meters to increase dispersion. There is no sufficient meteorological information to know how much sulfur dioxide Montana Sulphur is contributing to the high readings at the Lockwood cite ambient monitor or if the stack will reduce the ambient sulfur dioxide concentration.

The present emissions could be reduced to approximately one third their present levels with tail-gas cleaning equipment designed specifically for installations like Montana Sulphur. The capital cost for such equipment would be about $1.5 million. The annual operating costs of this unit would be approximately $.4 million.

In conclusion, as with the CONOCO refinery, it is assumed in this analysis that, in light of the projected sulfur dioxide reductions at the Exxon and Corrett sources, the exhaust stack extension would be sufficient to achieve compliance with the 50 ug/m^3 standard.

ASARCO Lead Smelter

ASARCO recently completed the installation of a new sulfuric acid plant to control sulfur dioxide emissions from a portion of the lead sinter machine. ASARCO also announced plans to raise the height of the stack on its blast furnace. These modifications in ASARCO's operations are predicted to bring the sulfur dioxide emissions into compliance with both the 80 ug/m^3 and the 50 ug/m^3 standards. Because there have been no sustained periods of adverse weather conditions since the new acid plant began normal operation, it is impossible to know if this will actually be achieved. If additional controls are required, it is expected that it will not require substantial reconstruction at the Lead Smelter

ASARCO has rebuilt the lead sinter machine so that most of the sulfur dioxide gas generated is recirculated through the machine until it reaches a high enough concentration to be treated in the acid plant. This results in an overall control level for the plant of about 75 percent. One vent and a hood at the end of the sinter machine are not recirculated, they are routed to the stack without any treatment for control of sulfur dioxide. If additional control is required, the remaining vent could be recirculated through the sinter machine also. This may not require any significant modifications to the sinter machine design or metalurgy; however, excess heat generated by the recirculating process could require modifying the lead sinter machine.

Assuming the recirculation process is required and that the sinter machine would not require modification, the cost of complying with the 80 ug/m^3 and 50 ug/m^3 standards would be approximately $6 million with an annual O and M of less than $.4 million.

Summary of Control Costs

Table 17 provides a summary of the estimated control costs associated with compliance with the alternative 80 ug/m^3 and 50 ug/m^3 annual sulfur dioxide standards. Listed are estimates of the expenditures required for air pollution control equipment and installation and the annual operating and maintenance costs associated with the equipment.

In order to compare these control cost estimates with the estimated annual benefits, the equipment costs were annualized into equivalent annual values and added to the operating and maintenance costs. The resulting annualized costs were found to be dominated by the annual operating and maintenance costs (i.e., acid neutralization) at the Anaconda Copper Smelter. Since it is only known that the O and M at Anaconda will be between $2.0 and $8.5 million, depending on the amount

Table 17

Estimated Air Pollution Control Costs Resulting from Alternative Ambient Sulfur Dioxide Standards

Industrial point source	80 ug/m^3 ($ millions)		50 ug/m^3 ($ millions)	
	Equip-ment costs	Annual O & M costs	Equip-ment costs	Annual O & M costs
Anaconda Copper Smelter	21.0	5.5	21.0	5.5
CENEX Petroleum Refinery	8.0	.5	10.0	.5
Corrett Power Plant	0	0	6.7	.5
Exxon Petroleum Refinery	0	0	4.5	.2
CONOCO Petroleum Refinery	0	0	0	0
Montana Sulphur and Chemical Company	0	0	0	0
ASARCO Lead Smelter	6.0	.4	6.0	.4
TOTALS	35.0	6.4	48.2	7.1

of acid neutralization required, and that only an analysis of the sulfuric market can reveal the amount of neutralization required, the lower and upper estimates of Anaconda's O and M costs were used to arrive at low and high estimates of total annual control costs.

The annualization of capital expenditures requires two values--a discount rate and a time period over which the capital cost is distributed. In general, the length of the time period and the average annual air pollution control costs are inversely related. A higher discount rate has the opposite effect: amortized costs will be greater. Appendix D provides estimates of the time period, discount rate, and resulting capitalization factors. It was found that the true life of air pollution control equipment is probably bounded by 20- to 40-year time periods. For mixed public/private investments such as air pollution control, it was found that an appropriate discount rate is probably between four percent and eight percent.

Due to the sensitivity of the resulting annualized costs to these parameters, the downward effect on control costs associated with use of the 40-year time period and four percent discount rate were used to generate the low estimate of control costs and the 20-year time period and eight percent rate were used for the high estimate.

Table 18 provides the estimated total annualized control costs associated with compliance with the 80 ug/m³ and the 50 ug/m³ ambient annual sulfur dioxide standards. Listed are totals for each area for the high and low estimates. An inspection of Table 18 reveals the sensitivity of the estimated annual control costs to the amount of acid

Table 18

Summary of Annual Control Costs Resulting from Alternative Sulfur Dioxide Standards in Montana

Area	80 ug/m ($ millions)		50 ug/m ($ millions)	
	High est.*	Low est.†	High est.*	Low est.†
Anaconda	10.6	3.1	10.6	3.1
Billings	1.3	.9	3.4	2.3
Helena	1.0	.7	1.0	.7
TOTALS	12.9	4.7	15.1	6.0

*Annualized over 20 years at eight percent discount rate including an annual O and M at Anaconda of $2.0 million.

†Annualized over 40 years at four percent discount rate including an annual O and M at Anaconda of $8.5 million.

neutralization required at the Anaconda Copper Smelter and the assumed annualization parameters. The high estimate of annual control cost required to achieve compliance with the 80 ug/m^3 standard ($12.9 million) is reduced 51 percent (to $6.5 million) using the low estimate of O and M at Anaconda; it is reduced 14 percent (to $11 million) using the 40-year time horizon and four percent discount rate. It is reduced 64 percent using both factors (the low estimate).

The high estimates of annual control cost associated with the alternative standards are dominated by the cost of compliance at Anaconda (82 percent and 70 percent for the 80 ug/m^3 and 50 ug/m^3 standards, respectively). The low estimates are distributed more evenly among the three areas, but the copper smelter in Anaconda still accounts for the lion's share (64 percent and 52 percent).

Chapter 4

ANALYSIS AND DISCUSSION

The information developed in Chapters 2 and 3 is used in this Chapter to examine the economic case for alternative annual ambient sulfur dioxide standards and the resulting policy implications.

Analysis of Alternative Standards

Tables 14 and 15 (pp. 47 and 48) provide a summary of the benefits resulting from compliance with the NAAQS (80 ug/m^3) and proposed MAAQS (50 ug/m^3) sulfur dioxide standards. The estimated health, vegetation, materials, and visibility annual benefits resulting from compliance with an 80 ug/m^3 standard are between \$5.7 and \$2.6 million. The benefits resulting from compliance with a 50 ug/m^3 standard are between \$8.6 and \$3.4 million. The site specific benefits range from a low estimate of \$.3 million for compliance with the 80 ug/m^3 standard in Helena to a high estimate of \$5.8 million for the 50 ug/m^3 standard in Billings.

The variation in the high and low estimates is due to two factors: (1) alternative dose-response coefficients relating ambient sulfur dioxide to mortality rates (38 percent of the variation), and (2) the alternative values attached to a sulfur dioxide induced premature death (62 percent of the variation).

Table 18 (p. 61) provides a summary of the annual air pollution control costs associated with compliance with the alternative standards. The annual cost of achieving statewide compliance with an 80 ug/m^3 standard is estimated to be between $12.9 and $4.7 million; the annual cost is estimated to be between $15.1 and $6.0 million for the 50 ug/m^3 standard. The site specific costs range from a high estimate of $10.6 million annually in Anaconda for compliance with both standards to $.7 million in Helena for compliance with both standards.

The variation in the annual control costs are also due to two factors: (1) the time period and discount rate used to annualize the control equipment costs (22 percent of the variation), and (2) the amount of sulfuric acid neutralization required in conjunction with an additional acid plant in Anaconda (78 percent of the variation).

Table 19 provides the range of net benefits (costs) resulting from compliance with the alternative standards in Montana as well as each area. Evident from the table is the sensitivity of the

Table 19

Estimated Annual Net Benefits Resulting from Alternative Ambient Sulfur Dioxide Standards in Montana

Area	80 ug/m^3 ($ millions)		50 ug/m^3 ($ millions)	
	High benefits low costs	Low benefits high costs	High benefits low costs	Low benefits high costs
Anaconda	-2.0	- 9.8	-1.6	- 9.7
Billings	2.9	.1	3.6	- 1.3
Helena	0	- .7	.6	- .5
TOTALS	.9	-10.3	2.6	-11.6

resulting net benefits to the acceptance of the low or high estimates of benefits and costs. If one were to accept the low estimate of benefits and high estimate of costs, the resulting total net benefits (costs) would be -$10.3 and -$11.6 million for the 80 ug/m^3 and 50 ug/m^3 standards, respectively. If one were to alternatively accept the high estimate of benefits and the low estimate of costs, the net social benefits would be $.9 and $2.6 million for the 80 ug/m^3 and 50 ug/m^3 standards, respectively. The site specific annual net benefits (costs) range from -$9.8 million in Anaconda (low benefits, high costs, and 80 ug/m^3) to $3.6 million in Billings (high benefits, low costs, and 50 ug/m^3).

The critical factor is where the true benefits and costs lie in the low to high range of estimates. As previously stated, the range of estimated control benefits is due to alternative factors used in deriving the health benefits--the dose-response relationship and, primarily, the value assigned to reductions in the occurrence of premature death.

The dose-response relationship between various air pollutants and health effects is of considerable debate. The primary argument against any relationship is that the statistical evidence fails to prove causality. Although this is true, it is erroneous to dismiss the statistical evidence as being spurious. Haveman (1979, p. 143), in his rather neutral review of the Lave and Seskin analysis, argues this point.

> While any individual estimate can be criticized (indeed, ridiculed), it is not possible to dismiss the persistence of positive and significant air pollution coefficients estimated with a wide variety of data and model specifications.

The generated low estimates of health benefits were based on a theoretically more appealing regression routine--the adjusted coefficient and mortality rate; however, in this regression the effect of sulfates on mortality rates was not measured and a vast majority of ambient sulfate is derived from sulfur dioxide emissions. This suggests that the true coefficient relating sulfur dioxide to mortality rates may be higher than the high unadjusted coefficient used to derive the health benefits.

The magnitude of this underestimate can be judged by inspecting the equations reported in Tables A.1 and A.2 (pp. 83 and 84). The sum of the SO_2 and sulfate elasticities (when both pollutants are in the same equation) is about the same as when only one of the pollutants is included. Since the sulfate is derived from SO_2 it is plausible to assume that the average of the SO_2 elasticities, *when sulfate is not in the equation,* is the better high estimate. For the unadjusted this is a 4.83 percent elasticity (equations 7.8-5 and 7.8-7); for the adjusted this is a 3.29 percent elasticity (equations 7.9-5 and 7.9-7) or linear coefficients of about 1.33 and .96, respectively. These coefficients are 32 percent and 47 percent higher than the high and low used in the benefit calculations derived above.

On the other hand, Lave and Seskin have not corrected for possible bias due to omitted variables such as smoking habits. Haveman (1979, p. 143) points out this shortcoming.

> And while Lave and Seskin have addressed several of the estimation problems in their analysis, others remain. The absence of any control for differential smoking, nutritional, or health care usage patterns is a serious problem. To what extent are these patterns correlated with other socioeconomic variables or air pollution,

hence, confounding the results?

The potential significance of the omitted variable bias is evident from inspecting the explanatory power of the adjusted regressions. The R^2 values range from .497, when only SO_2 is regressed on the adjusted mortality rates, to .728 when all five of the pollutants considered are included in the regression. This implies that 30 percent to 50 percent of the variation in mortality rates remain unexplained and could lead to an omitted variable bias.

The second factor upon which alternatives were used to generate health benefits was the value assigned to reduced occurrence of premature death. The low estimate is based on the foregone earnings concept. The only argument for using this approach is that a person's ability to pay is constrained by a budget constraint--represented by foregone earnings.

There is no evidence, however, that there exists any correlation between foregone earnings and the social disutility resulting from a premature death. Freeman (1979, p. 11) argues that

> The technique and estimating procedures must be based analytically and empirically on individual behavior and preference. . . . Some measures of the value of reduced mortality have been based upon the earnings of affected individuals. Such measures do not meet this criterion, since there is no known relationship between willingness to pay on the one hand and earnings on the other. The most obvious limitation of the lost earnings measure is that it places no value on the lives of those who are not working because of their age, sex or other factors.

Although the foregone earnings concept is commonly used in the literature (including Lave and Seskin), it represents nothing more than an arbitrarily chosen measure incorrectly used to represent the value of human life. There exists absolutely no connection between earnings potential and welfare, neither on an individual basis nor a collective basis.

On the other hand, it has been argued that the difference in age distribution between the susceptible population and Thaler and Rosen's (1976) relatively young occupational sample presents an upward bias when applying the risk aversion measures to air pollution induced mortality. The rationale for this argument is that the Thaler and Rosen estimate approximates the value of a relatively young person's life while air pollution affects primarily the elderly and, perhaps, idle population that would value their lives less. This argument can be rebutted on several grounds. First, Thaler and Rosen's occupational sample of industrial workers was not composed primarily of young people. The sample was distributed among all working ages. Second, an inspection of Mendolsohn and Orcutt's (1978) predicted mortality rates reveals that not only old, retired people are subjected to premature death, but middle-aged and younger populations are also subjected to premature death.

The most convincing argument against the Thaler and Rosen risk aversion measures suggests that the measures are *underestimates* of the true value of reduced occurrence of premature death. Gregor (1977, p. 59) argues that

> Thaler and Rosen's sample was composed of relatively risky occupations. Therefore, the individuals included in the sample were relatively less risk averse than the general population. This downward bias coupled with the fact that these risk premiums only approximate the individual's own willingness to pay for risk reductions and not those of family, friends and society insure that any benefit estimates based on the Thaler and Rosen results will unequivocally represent a lower bound estimate.

In light of this argument and the lack of measurement of health benefits resulting from reduced levels of sulfate, it is probably the case that the true health benefits are better represented by the high estimate.

Accepting the estimated vegetation and materials benefits as best estimates, one can also say that the rest of the benefits estimated are surely biased downward. Among the possibly substantial aesthetic and amenity damages unaccounted for in the analysis are soil erosion, disturbance to aquatic ecosystems, climatic variations, and the general irreversible imbalance imposed upon the natural environment.

In light of this evidence it appears safe to conclude that a higher level of credence is attached to the high estimates of control benefits. It is surely the case that the true benefits of the alternative standards are best represented by the high estimates of control benefits and they probably exceed those estimates.

Whereas the control benefits are inherently biased downward, the control costs are inherently biased upward. Although nothing can be said regarding the specific factors that comprise the high and low estimates of control costs (time period, discount rate, and acid neutralization), the control cost estimates, as a whole, warrant discussion.

Several factors usually result in inflated control cost estimates. Lave and Seskin (1978, p. 212) point out two such factors.

> The most common way of measuring the costs of abatement is to estimate the cost of adding devices that would reduce emissions from existing facilities. But adding an abatement device to an industrial smokestack is often economically inefficient, since it may also decrease the operating efficiency of the firm. In the long-run, it is often more economical to construct a new plant designed to control emissions even if the initial investment is

quite high. Furthermore, present cost estimates are usually based on current technology; at best, the analyst can make only educated guesses as to the effects of advancing technology. Because such guesses are subject to considerable uncertainty, they are seldom used in cost estimates. Consequently, cost estimates of pollution abatement are likely to be overestimated, since one expects major technological advances in the efficiency of control systems.

In addition to these technology and reconstruction factors is the absence of an examination of the costs associated with plant closure or production cutback control techniques. These are viable alternatives that have not been investigated.

Atkinson and Lewis (1974) also found that the true least cost combination of control equipment among multiple point sources was much less costly than the sum of the estimated costs for each source. Although this is applicable only to areas with multiple sources such as Billings, it implies that it may be possible, based on a marginal analysis of each source, to achieve emission reductions at a fraction of the estimated cost by controlling the emissions from those sources where it is least costly.

Thus it is probably the case that the true control costs are best represented by the low estimate, but sufficient evidence is not available to confirm it. The true control costs associated with achieving compliance with the 80 ug/m^3 and 50 ug/m^3 standards remain obscure in light of the crudeness of the estimates and an uncertain technology.

Because of this uncertainty regarding the true control costs associated with compliance with the alternative standards, it is not possible to ascertain the true net benefits resulting from the standards. It thus is not possible to identify the preferred standard. The analysis does, however, reveal several policy implications discussed below.

Conclusion and Policy Implications

The economic examination of alternative ambient sulfur dioxide standards revealed that substantial benefits would result from abatement of sulfur dioxide air pollution in Montana. The estimated annual health, vegetation, materials, and visibility benefits resulting from compliance with the NAAQS 80 ug/m^3 annual standard and the proposed MAAQS 50 ug/m^3 standard are probably in excess of \$5.7 and \$8.6 million, respectively. These estimates require acceptance of the higher of two alternative dose-response coefficients relating ambient sulfur dioxide to mortality and the higher of two alternative values assigned to reductions in the occurrence of premature death. As discussed above, evidence from the literature suggests that acceptance of the first premise is valid given the inability to include the benefits of reductions in sulfate-induced mortality. The latter premise is correct given that it represents the full willingness to pay value associated with premature death.

Although these estimated benefits are substantial, the estimates of air pollution control costs associated with the alternative standards were found to be insufficiently refined to allow the selection of a preferred standard. The control costs required for compliance with the alternative standards are obscure in light of the limited knowledge concerning applicable tecnhologies and the cost of those technologies.

In addition to the estimated benefits mentioned above, the analysis revealed several other policy implications. Whereas the cost of sulfur dioxide abatement in Montana was found to be dominated by the

Anaconda copper smelter, the benefits were found to be dominated by the urban Billings population. This finding presents an appealing case for site specific standards as opposed to a uniform statewide standard.

The remaining major policy implication evident from the analysis is the shortage of data requirements from which the analytical techniques available can select a preferred standard. Among the most critical information needs are sufficiently accurate ambient sulfate concentrations. Sulfate monitoring is a necessary input into the decision-making process. This includes both urban and rural monitoring. The state of Florida's economic analysis (SRI, 1978) of ambient sulfur oxide standards focused on the sulfate induced health and visibility effects. Despite the presence of sensitive citrus fruits and a high relative humidity, both vegetation, and material damages were found to be relatively negligible in light of the magnitude of the sulfate induced health and visibility benefits.

Given the availability of sufficient monitoring, a second information need is the dose-response relationship between the receptors and ambient concentrations of sulfur dioxide. Given the natural environment characteristic of Montana, this is especially critical for all vegetation species (i.e., not only consumable agricultural crops).

The third and most limiting information shortage revealed by the analysis is on the cost side. The information currently available regarding abatement of sulfur dioxide is not sufficient for economically based policy decisions.

The findings reported in this analysis are based on the limited Montana specific information available plus an extensive review of the air pollution literature. There is no question that extensive further Montana specific research into damages, dispersion, monitoring, and control alternatives could provide the basis for more extended economically-based conclusions.

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APPENDIX A

CALCULATION OF THE LINEAR COEFFICIENTS AND MORTALITY RATES USED IN THE HEALTH BENEFIT ESTIMATION

This appendix provides an examination of the Lave and Seskin regression results and the two linear coefficients and mortality rates used in deriving the high and low estimates of health benefits. Tables A.1 through A.3 and Exhibit A.1 provide the Lave and Seskin multiple regression results; the definition, mean value, and standard deviation of the variables used; and the rationale and procedure for adjusting the mortality rates.

As evident from a comparison of Tables A.1 and A.2, the statistical properties of the sulfur dioxide coefficients of the unadjusted regression are superior to those from the adjusted regression. Whereas only one of the four mean sulfur dioxide coefficients of the adjusted regression are significant at the 10 percent level, three of the four coefficients of the unadjusted regression are significant at the 10 percent level, and the fourth is significant at the 15 percent level. An inspection of all of the sulfur dioxide coefficients reveals that the minimum and mean readings are generally significant. The maximum sulfur dioxide coefficients are all negative and, in one case (equation 7.9-9), it is significantly negative. This is not surprising, however, considering the variation in maximum readings observed (see Table A.3).

The lower R^2 values associated with the adjusted regression reflect the loss of explanatory power caused by differences in the age-race make-up of the population. They also reflect the lower variation

Table A.1

Unadjusted Total Mortality Rates (the Effects of Additional Air Pollutants), 1969

	7.8-1	7.8-2	7.8-3	7.8-4	7.8-5	7.8-6	7.8-7	7.8-8	7.8-9	7.8-10
R^2	.848	.842	.792	.815	.843	.869	.875	.915	.914	.924
Constant	570.798	567.612	526.489	753.474	702.037	612.791	699.055	743.408	675.191	808.081
Air pollution variables										
Min S	−.821					−1.201		−1.761	−1.834	−1.949
	(−1.44)					(−2.08)		(−3.26)	(−3.40)	(−3.61)
Mean S	1.116					1.098		1.142	1.061	1.030
	(3.48)					(3.21)		(3.51)	(3.28)	(3.18)
Max S	−.080					−.124		−.132	−.092	−.101
	(−.91)					(−1.40)		(−1.67)	(−1.15)	(−1.28)
Sum S elasticities	8.51					5.47		3.61	3.58	2.45
Min P		.230				.019	.259	−.300	.140	−.103
		(.35)				(.03)	(.40)	(−.51)	(.22)	(−.16)
Mean P		.656				.044	.314	.018	−.010	−.145
		(1.69)				(.10)	(.79)	(.05)	(−.02)	(−.35)
Max P		.136				.153	.139	.223	.181	.226
		(2.00)				(2.34)	(2.02)	(3.79)	(2.73)	(3.37)
Sum P elasticities		12.00				5.07	8.47	5.71	5.74	4.71
Min NO_3			2.478					2.027	−.031	1.245
			(.63)					(.66)	(−.01)	(.41)
Mean NO_3			−.509					−1.395	−.203	−.746
			(−.31)					(−1.12)	(−.16)	(−.59)
Max NO_3			−.290					−.463	−.507	−.464
			(−.76)					(−1.64)	(−1.77)	(−1.66)
Sum NO_3 elasticities			−1.98					−5.48	−3.93	−4.32
Min NO_2				−.853				−.317		−.376
				(−1.97)				(−.91)		(−1.10)
Mean NO_2				.994				.674		.553
				(2.60)				(2.13)		(1.76)
Max NO_2				−.141				−.008		−.007
				(−.94)				(−.07)		(−.06)
Sum NO_2 elasticities				6.72				8.45		6.41
Min SO_2					5.973		3.703		2.749	1.709
					(2.66)		(1.62)		(1.31)	(.80)
Mean SO_2					1.134		1.188		1.397	1.158
					(1.49)		(1.66)		(2.08)	(1.76)
Max SO_2					−1.27		−.156		−.285	−.227
					(−.60)		(−.80)		(−1.51)	(−1.23)
Sum SO_2 elasticities					5.48		4.17		2.81	2.16
Socioeconomic variables										
P/M^2	.151	.162	.136	.153	.168	.171	.170	.156	.178	.172
	(3.28)	(3.42)	(2.55)	(2.82)	(3.56)	(3.75)	(3.73)	(3.67)	(4.34)	(4.03)
≥ 65	5.616	6.172	5.998	6.167	5.677	5.767	5.887	5.845	5.456	5.700
	(14.70)	(16.34)	(13.55)	(14.74)	(14.87)	(15.44)	(16.55)	(16.47)	(16.21)	(16.16)
NW	.488	.575	.484	.526	.493	.584	.535	.605	.525	.589
	(4.02)	(4.80)	(3.46)	(3.99)	(4.15)	(4.81)	(4.81)	(5.44)	(4.70)	(5.21)
Poor	−.035	−.134	.034	.062	.199	−.207	.088	−.090	−.057	−.045
	(−.11)	(−.43)	(.09)	(.18)	(.62)	(−.67)	(.30)	(−.33)	(−.21)	(−.17)
Sum SE elasticities	64.33	70.28	68.60	71.34	67.46	65.77	69.11	67.82	63.34	66.75
Log Pop	−.552	−.693	−.380	−.945	−.777	−.678	−.909	−.960	−.710	−1.039
	(−2.30)	(−2.88)	(−1.30)	(−3.06)	(−3.06)	(−2.87)	(−3.85)	(−3.66)	(−2.97)	(−3.88)

Note: All regressions are based on data for sixty-nine SMSAs (see table C.1, pp. 317-320). The numbers in parentheses below the regression coefficients are *t* statistics.

Source:

Lester B. Lave and Eugene P. Seskin, *Air Pollution and Human Health* (Baltimore: Johns Hopkins University Press, 1977), pp. 146-147.

Table A.2

Age-Sex-Race-adjusted Total Mortality Rates (the Effects of Additional Air Pollutants), 1969

	7.9-1	7.9-2	7.9-3	7.9-4	7.9-5	7.9-6	7.9-7	7.9-8	7.9-9	7.9-10
R^2	.516	.531	.355	.434	.497	.594	.607	.689	.702	.728
Constant	1067.027	1057.043	1014.537	1205.630	1166.708	1102.833	1154.284	1193.226	1110.729	1205.511
Air pollution variables										
Min S	−.761 (−1.45)					−1.062 (−2.03)		−1.486 (−2.80)	−1.466 (−2.85)	−1.501 (−2.85)
Mean S	.979 (3.32)					.889 (2.87)		.901 (2.82)	.920 (2.98)	.827 (2.62)
Max S	−.078 (−.96)					−.118 (−1.48)		−.122 (−1.56)	−.094 (−1.23)	−.082 (−1.06)
Sum S elasticities	6.71					3.33		1.84	2.98	2.08
Min P		.298 (.52)				.094 (.16)	.086 (.14)	−.201 (−.34)	−.134 (−.22)	−.281 (−.45)
Mean P		.604 (1.76)				.182 (.46)	.516 (1.43)	.155 (.41)	.317 (.78)	.218 (.54)
Max P		.126 (2.09)				.142 (2.40)	.099 (1.58)	.187 (3.24)	.120 (1.90)	.143 (2.19)
Sum P elasticities		10.78				6.19	8.42	6.17	6.20	5.32
Min NO_3			3.156 (.89)					2.400 (.80)	1.362 (.47)	1.865 (.64)
Mean NO_3			−.998 (−.68)					−1.486 (−1.21)	−1.065 (−.89)	−1.134 (−.92)
Max NO_3			−.030 (−.09)					−.212 (−.76)	−.263 (−.96)	−.269 (−.99)
Sum NO_3 elasticities			−1.00					−3.61	−3.47	−3.43
Min NO_2				−.888 (−2.27)				−.417 (−1.22)		−.522 (−1.57)
Mean NO_2				.923 (2.69)				.658 (2.12)		.544 (1.78)
Max NO_2				−.170 (−1.26)				−.066 (−.58)		−.071 (−.64)
Sum NO_2 elasticities				4.40				5.64		3.45
Min SO_4					6.201 (3.00)		4.329 (2.09)		3.548 (1.77)	3.141 (1.52)
Mean SO_4					.928 (1.39)		.963 (1.48)		1.079 (1.68)	.976 (1.52)
Max SO_4					−.194 (−.99)		−.225 (−1.27)		−.335 (−1.85)	−.319 (−1.77)
Sum SO_4 elasticities					3.95		2.64		1.38	.99
Socioeconomic variables										
P/M^2	.110 (2.61)	.118 (2.81)	.100 (2.08)	.117 (2.39)	.133 (3.08)	.129 (3.12)	.139 (3.36)	.126 (3.01)	.146 (3.711)	.150 (3.60)
≥65	−.295 (−.84)	.198 (.59)	.087 (.22)	.143 (.38)	−.194 (−.55)	−.119 (−.35)	−.004 (−.01)	−.047 (−.13)	−.354 (−1.10)	−.206 (−.60)
NW	.329 (2.95)	.399 (3.75)	.334 (2.64)	.337 (2.84)	.335 (3.06)	.424 (3.85)	.377 (3.73)	.431 (3.94)	.364 (3.42)	.385 (3.49)
Poor	.167 (.57)	.088 (.31)	.229 (.69)	.260 (.84)	.320 (1.08)	−.007 (−.03)	.195 (.72)	.087 (.33)	.084 (.32)	.095 (.37)
Sum SE elasticities	4.30	9.27	8.46	9.47	6.97	5.84	8.26	7.46	3.81	5.64
Log Pop	−.363 (−1.65)	−.494 (−2.32)	−.215 (−.81)	−.646 (−2.32)	−.532 (−2.27)	−.496 (−2.32)	−.648 (−3.01)	−.685 (−2.65)	−.436 (−1.91)	−.649 (−2.48)

Note: All regressions are based on data for sixty-nine SMSAs. The numbers in parentheses below the regression coefficients are *t* statistics.

Source:

Lester B. Lave and Eugene P. Seskin, *Air Pollution and Human Health* (Baltimore: Johns Hopkins University Press, 1977), pp. 150-151.

Table A.3

Variable	Description	1969 (69 SMSAs) Mean	Standard deviation
Air Pollution	Min S: Smallest biweekly sulfate reading (μg per cubic meter × 10)	34.768	18.608
	Mean S: Arithmetic mean of biweekly sulfate readings (μg per cubic meter × 10)	115.652	47.258
	Max S: Largest biweekly sulfate reading (μg per cubic meter × 10)	286.435	142.852
	Min P: Smallest biweekly suspended particulate reading (μg per cubic meter)	32.290	14.691
	Mean P: Arithmetic mean of biweekly suspended particulate readings (μg per cubic meter)	99.507	27.436
	Max P: Largest biweekly suspended particulate reading (μg per cubic meter)	268.768	149.779
	Min NO_3: smallest biweekly nitrate reading (μg per cubic meter × 10)	4.377	3.469
	Mean NO_3: arithmetic mean of biweekly nitrate readings (μg per cubic meter × 10)	21.609	11.774
	Max NO_3: largest biweekly nitrate reading (μg per cubic meter × 10)	61.667	36.029
	Min NO_2: smallest biweekly nitrogen dioxide reading (μg per cubic meter)	40.319	24.885
	Mean NO_2: arithmetic mean of biweekly nitrogen dioxide readings (μg per cubic meter)	136.232	53.882
	Max NO_2: largest biweekly nitrogen dioxide reading (μg per cubic meter)	281.667	119.704
	Min SO_2: smallest biweekly sulfur dioxide reading (μg per cubic meter)	4.536	3.592
	Mean SO_2: arithmetic mean of biweekly sulfur dioxide readings (μg per cubic meter)	33.000	29.355
	Max SO_2: largest biweekly sulfur dioxide reading (μg per cubic meter)	115.812	97.608
Socioeconomic	P/M^2: SMSA population density (per square mile × 0.1)	97.813	160.781
	≥65: Percentage of SMSA population at least sixty-five years old (×10)	90.913	20.487
	NW: Percentage of nonwhites in SMSA population (×10)	129.609	92.209
	Poor: Percentage of SMSA families with incomes below the poverty level (×10)	87.899	35.103
	Log Pop: The logarithm of SMSA population (×100)	586.181	35.907
Mortality	Unadjusted total mortality rate (per 100,000)	910.058	138.238
	Age–sex–race-adjusted total mortality rate (per 100,000)	959.812	71.094

Source:

Lester B. Lave and Eugene P. Seskin, *Air Pollution and Human Health* (Baltimore: Johns Hopkins University Press, 1977), pp. 323, 336.

in mortality rates caused by the adjustment procedure (see Table A.3).

Another major point, evident from the regression results, is the effect on the sulfur dioxide elasticities when sulfate is added to the regression, e.g., the sum of the SO_2 elasticities for equation 7.8-7 (SO_2 and P regressed on mortality) of .0417 percent is reduced to .0281 percent (equation 7.8-9) when SO_4 is added to the regression. The implications of this are discussed in Chapter 4. This indicates that a relatively substantial effect on mortality is achieved by reducing ambient sulfate--a derivative of sulfur dioxide. The result could be a substantial underestimate in health benefits by using an average of the sum of the SO_2 elasticities rather than the one sum of the SO_2 elasticities derived from equation 7.8-5. Ideally, one should consider the effect of decreased sulfate resulting from reductions in sulfur dioxide emissions.

The linear coefficients used to derive a health benefit were derived by averaging the sums of the SO_2 elasticities from each of the two sets of four equations.[1] The average of the unadjusted sum of the SO_2 elasticities is .0366 percent; the adjusted is .0232 percent. By definition these elasticities measure the percent change in mortality associated with a one percent change in ambient sulfur dioxide or

[1]This procedure was personally endorsed by Eugene Seskin (letter to Martin Perga, CENEX, 15 February 1979).

$$\frac{\frac{\Delta M}{M}}{\frac{\Delta SO_2}{SO_2}} = \eta \rightarrow \frac{\Delta M}{\Delta SO_2} = \eta \frac{M}{SO_2} \qquad [A.1]$$

where: M = mean mortality rate

SO_2 = mean sulfur dioxide reading

η = elasticity.

To convert the elasticity to a linear coefficient the known values (see Table A.3) can be substituted into the expression on the right. For the unadjusted linear coefficient, the derivation is as follows:

$$\Delta M = .0366 \left(\frac{910.058}{33.000}\right) = 1.01 \qquad [A.2)$$

A similar procedure, using the adjusted elasticity (.0232) and mortality rate (959.812), results in a low coefficient of .65. These coefficients represent the unit change (rather than percent) in mortality per one ug/m³ (rather than one percent) in ambient sulfur dioxide. The implications of the alternative, constant elasticities, is discussed in the text.

Calculation of the unadjusted Montana mortality rate is straightforward. An average of 1969, 1970, and 1971 mortality per 100,000 1970 population is 968.160.[2] The calculation of an adjusted mortality rate (see Exhibit A.1) is slightly more complicated. The Montana mortality rate is calculated (again a 1969, 1970, and 1971 average) for each of

[2]Mortality figures are derived from *Vital Statistics*, published annually; the population was obtained from the *1970 Census of Population*.

nine age groups[3] for both white and nonwhite population.[4] These 18 mortality rates were then weighted by percent make-up of the national population. The resulting age-race adjusted Montana mortality rate is 981.34, slightly more than the unadjusted rate. This is primarily due to Montana's relatively small nonwhite population. The age-race adjusted mortality rate corresponds to what Montana's mortality rate would be if its population had the demographic characteristics of the national average.

Since the linear coefficients derived above estimate the unit change in mortality, the effect of using the age-race adjusted mortality would be to slightly reduce the resulting percent reduction. Thus the downward effect of the adjusted mortality corresponds to the downward effect of using the adjusted coefficient. These two factors together are used to generate the low estimate. The unadjusted mortality is used with the unadjusted coefficient for the high estimate.

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[3] < 5, 5-14, 15-24, 25-34, 35-44, 45-54, 55-64, 65-74, ≥ 75.

[4] An inspection of sex proportions in Montana revealed little deviation from the national averages.

Exhibit A.1

ADJUSTMENT METHOD FOR MORTALITY RATES

Let MR_i be the SMSA mortality rate for age-sex-race group i and let P_i be the proportion of the total U. S. population in age-sex-race group i. Then the direct method for calculating the age-sex-race-adjusted total mortality rate (ASR) is $ASR = \sum_i MR_i P_i$.[1]

Since the adjustment procedure uses a single age distribution for adjusting the mortality rate of each SMSA, it permits direct comparison of the same age-adjusted, sex-race mortality rate across areas but does not permit comparison of different age-adjusted, sex-race mortality rates.

An example may clarify the issue. Assume there are two age groups, young (Y) and old (O). Suppose that 80 percent of the total population are young and 20 percent are old, and that their mortality rates (MR_Y and MR_O, respectively) are 2 and 9 per 100, respectively; for example, $MR_Y = 2$ and $MR_O = 9$. This gives rise to a national unadjusted total mortality rate of (0.8 x 2) + (0.2 x 9) = 3.4. Now, let Clean City have a higher proportion of old people than the national average (say, 50 percent) but lower than average mortality rates: MR_Y (Clean City) = 1 and MR_O (Clean City) = 7, and let Dirty City have a much lower proportion of old people (5 percent) but relatively high mortality rates: MR_Y (Dirty City) = 2.5 and MR_O (Dirty City) = 12. With these assumptions, the unadjusted total mortality rate in Clean City would be equal to (0.5 x 1) + (0.5 x 7) = 4, and the unadjusted total mortality rate in Dirty City would be equal to (0.95 x 2.5) + (0.05 x 12) = 2.975. Clean City's relatively high unadjusted total mortality rate (compared with Dirty City's and the national average) carries with it the implication that there are factors in Clean City that make it a less healthy environment, an incorrect conclusion. Similarly, Dirty City's relatively low unadjusted total mortality rate suggest that there are factors in Dirty City that make it a healthy place to live. However, if the total mortality rate in each city is adjusted for age, using the age distribution of the national population, the two mortality rate become: *ASR* (Clean) = (0.8 x 1) + (0.2 x 7) = 2.2, and *ASR* (Dirty) = (0.8 x 2.5) + (0.2 x 12) = 4.4. Thus the adjustment provides for a more valid comparison of the mortality rates.

As a second illustration, assume that 80 percent of whites are young while 90 percent of nonwhites are young (and that 81 percent of the national population is young). Now let MR_Y (White) = 2, MR_O (White) = 9, MR_Y (Nonwhite) = 2.4, and MR_O (Nonwhite) = 10. Using the single age distribution of the national population, the two adjusted mortality rates would be *ASR*(White) = (0.81 x 2) + (0.19 x 9) = 3.33, and *ASR*(Nonwhite) = (0.81 x 2.4) + (0.19 x 10) = 3.844. These two rates are directly comparable and indicate that the adjusted nonwhite mortality rate is approximately 15 percent higher than the adjusted white mortality rate. If the individual age distributions for the two races were used to calculate the adjusted mortality rates, the results would be *ASR*(White) = (0.2 x 2) + (0.2 x 9) = 3.4, and *ASR*(Nonwhite) = (0.9 x 2.4) + (0.1 x 10) = 3.16. These two rates are *not* directly comparable; they would imply that the nonwhite death rate was lower than the white death rate.

[1]Note that $\sum_i P_i = 1$, and that for the entire nation the unadjusted total mortality rate (UMR) and the age-sex-race-adjusted total mortality (*ASR*) are equal. The last equality can be seen from the following relationship:

$$ASR = \sum_i \frac{D_i}{P_i} \times \frac{P_i}{U.S.\ Pop} = \frac{1}{U.S.\ Pop} \sum D_i = UMR$$

where D_i is the number of deaths in age-sex-race group *i*, and *U.S. Pop* is the total population of the United States.

Source:

Lester B. Lave and Eugene P. Seskin, *Air Pollution and Human Health* (Baltimore: Johns Hopkins University Press, 1977), Appendix 3, pp. 356-357.

APPENDIX B

UNIFORM EMISSION ROLLBACK PROCEDURE

This appendix provides the uniform emission rollback as applied to the three analysis areas of Montana. Uniform emission rollback assumes a linear relationship between point source emissions and ambient concentrations. The derivation takes into account background levels of ambient sulfur dioxide concentrations.

In the case of areas with only one major point source, such as Anaconda and Helena, the procedure is straightforward. When an area contains multiple sources, such as Billings, the calculation is slightly more complex. Each of these two situations are handled separately.

Table 1 (p. 11) lists the ambient sulfur dioxide measurements at each site in the three analysis areas. Since Billings has the highest area source sulfur dioxide emissions (Gelhaus et al. 1978) and the monitoring site in downtown Billings measures 5.7 ug/m^3, for purposes of this analysis it is assumed that the three analysis areas in Montana have background concentrations of ambient sulfur dioxide of 5 ug/m^3.[1]

Uniform emission rollback in Anaconda and Helena is simply a matter of calculating the percent reduction in ambient concentration at the worst site required to meet a standard and assume the other

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[1]This assumption is consistent with the average minimum SO_2 reading of Lave and Seskin's SMSA sample (see Table A.3).

sites will experience an equal reduction in the ambient concentration attributed to nonbackground sources. The worst site in Anaconda is the Highway Junction monitor with an ambient reading of 108.7 ug/m^3. In order to comply with the 80 ug/m^3 standard, a reduction of 28.7 ug/m^3 or 26.4 percent must occur at this site. Uniform emission rollback assumes that this 26.4 percent reduction in ambient concentration must be accompanied by an equal reduction in emissions. The question is, what effect will a 26.4 percent reduction in emissions have on the other sites? The calculation for Anaconda follows: the Post Office site has current readings of 31.5 ug/m^3 and 5 ug/m^3 is assumed background level; therefore, the 26.4 percent reduction in emissions will result in a reduction of 7.0 ug/m^3 ([31.5 - 5] x .264). The revised reading would be 24.5 ug/m^3.

For compliance with a 50 ug/m^3 standard, the Highway Junction site would likewise realize a 58.7 ug/m^3 (54.0 percent) reduction in ambient sulfur dioxide. This would result in a reduction at the Post Office site of 14.3 ug/m^3. The uniform emission rollback procedure for Helena is similarly calculated.

Multiple sources slightly complicate the calculation for Billings. The procedure is in three steps: (1) apportion the total area point source emissions to each monitoring site, (2) calculate the ambient reduction required at each site as a percentage of the area's total emissions, and (3) apply the percentage of emission reduction to each monitoring site. The procedure follows. Table 16 (p. 52) lists the five major sulfur dioxide point sources in the Billings area. A clear geographic division is evident. The CENEX refinery is west of Billings

(CENEX farm site); the remainder are east of Billings (Lockwood site). Emissions are apportioned under these conditions with 10,380 tons of sulfur dioxide per year at the CENEX farm site and the remainder, 24,514 tons, to the Lockwood site.

For compliance with an 80 ug/m^3 ambient standard, the CENEX farm site (see Table 1, p. 11) must achieve a 68.7 ug/m^3 or 46.0 percent reduction in ambient sulfur dioxide. The Lockwood site need not achieve a reduction because it is currently below the 80 ug/m^3 level. The uniform emission rollback technique assumes that this reduction at the CENEX farm site must be accompanied by an equal reduction in emissions for a reduction of 4,796 tons per year (10,380 x .460). This emission reduction is equivalent to a 13.7 percent areawide emission reduction (4,796/34,894 x 100). Therefore, the other two sites will realize, from area compliance with an 80 ug/m^3 standard, a 13.7 percent reduction in nonbackground ambient sulfur dioxide. The reduction at the Lockwood site would be 9.9 ug/m^3 ([77.2 - 5] x .137). The reduction at the N. 27th site would be negligible .

Both sites must achieve reductions in ambient levels in order to comply with a 50 ug/m^3 standard: 98.7 ug/m^3 at the CENEX farm site and 27.2 ug/m^3 at the Lockwood site. The reduction at the N. 27th site would again be negligible against the assumed background level of 5 ug/m^3.

APPENDIX C

VEGETATION DAMAGE CLASS MODIFICATION

This appendix provides the rationale for modifying the SRI crop damage classes. Based on the scientific literature available, the SRI analysis grouped crops into damage classes on a scale of A to E with A representing those crops whose economic portion of the plant is highly susceptable to sulfur dioxide fumigations. A damage class of E was given to those crops whose economic portions are totally resistant to sulfur dioxide.

These damage classes require modifications to reflect knowledge put forth by recent publications. Of the eight crops examined, alfalfa and sugar beets remain in the damage classes designated by SRI. The remainder, barley, beans, hay, oats, timber, and wheat, were modified as follows:

Crop	SRI damage class	Modification
Barley	C	B
Beans	C	B
Hay	E	C
Oats	C	B
Timber	C	A
Wheat	C	A

The rationale behind the modification is twofold: (1) the scientific literature published since the original SRI analysis documented higher levels of sensitivity of these crops and (2) the SRI damage classes are relative classifications, e.g., if alfalfa was given an A classification and the literature suggests that wheat is as sensitive or more sensitive to sulfur dioxide, wheat was modified to an A classification.

Guderian (1977) reported that, at ambient concentrations of only 40 ug/m^3 sulfur dioxide averaged over the growing season, wheat suffered a 15 percent yield loss. Materna (1973) found that, at average annual concentrations of 20 to 24 ug/m^3, fir forests experienced a 20 percent loss in growth. He also found that concentrations of 70 to 100 ug/m^3 resulted in the rapid death of whole groups of trees. Based on this evidence the damage classes for wheat and timber were modified to class A--that of alfalfa.

Guderian (1977) also found that, at sulfur dioxide concentrations of only 25 ug/m^3 averaged over the growing season, oats suffered leaf injury. The EPA (1973) in its reevaluation of sulfur dioxide criteria, found that barley and beans are relatively nonresistant to sulfur oxides. It based its conclusion on results documented by Zimmerman (1955) and Middleton et al. (1958). Based on these studies it was felt that a C ranking, that of relatively resistant crops such as sugar beets, was not appropriate; oats, barley, and beans were given damage class B ratings.

The original SRI analysis classified hay as an E crop, totally resistant to *any* concentration of sulfur dioxide. A multitude of evidence (e.g., Wilhour et al. 1978 and EPA 1978a) suggests that rye grass and alfalfa are highly sensitive to ambient sulfur dioxide. Hay was given a damage class of C based on this evidence.

An additional factor in the original SRI damage class designation warrants discussion. As previously mentioned, the SRI damage class designations were based on not only the susceptibility of the plant, but the susceptibility of the economic portion of the plant, e.g., a plant that was found to be highly sensitive to sulfur dioxide, but whose economic portion was relatively resistant, was given a damage class ranking of C. The question is, what is the economic portion of a plant? A review of the SRI analysis reveals that the economic portion is construed to mean the portion harvestable, marketable, eatable, etc. This definition of "economic portion" is, indeed, incomplete because of the omission of nonmarket values, e.g., the mere knowledge of a plant being harmed by air pollutants may pose a significant although unquantified social disutility. This phenomenon probably introduces a significant downward bias in the results. In light of this, even the modified damage classes must be considered a conservative approach in estimating annual vegetation damage.

A P P E N D I X   D

DERIVATION OF THE AMORTIZATION FACTORS USED TO CONVERT CONTROL COSTS INTO ANNUAL COSTS

This appendix provides the derivation of the capital recovery factors used to convert the expenditures on air pollution control equipment into annual costs. The capitalization factors require two inputs: an appropriate real discount rate and a time period over which the equipment cost is distributed.

The desired time period is the life of the air pollution control equipment. Although some of the existing control equipment in Montana has lasted in excess of 50 years, some key components of the equipment can be expected to need replacement within 15 years of the installation date. Replacement of minor components is assumed to be included with operating and maintenance expenditures. The average life of the control equipment is probably bounded by a 20- to 40-year time period. Both of these values are used in this analysis.

Because air pollution abatement involves both public and corporate interests, an appropriate discount rate must consider both the willingness of society to forego present consumption for future rewards and the opportunity cost of corporate air pollution control capital.

Sjaastad and Wisecarver (1977) proposed an equation to estimate a discount rate for mixed public/private investments.

$$\rho = \alpha r + (1 - \alpha)i \qquad [D.1]$$

where: α = fraction of the investment which comes from corporate funding

r = before-tax corporate opportunity cost

i = social interest rate.

The equation merely proposes an average rate weighted by the fraction of the investment which comes from corporate and public $(1 - \alpha)$ funds. Values for each of r, i, and α need to be estimated to utilize this equation.

The opportunity cost of corporate funds (r) is the foregone return on investment in this sector. The before-tax profits and net interest payments as a percent of assets for all nonfinancial corporations was 15.5 percent, 15.6 percent, and 14 percent in 1972, 1973, and 1974, respectively.[1] In 1974 the return on stockholder equity after tax for the the top 500 corporations was 13.6 percent and, for all manufacturing, 14.9 percent.[2] Fifteen percent is probably a conservative estimate of the foregone return on corporate investment.[3]

A source of funds approach can be used to derive an appropriate social interest rate (i). This approach examines the return in foregone uses of each source of U.S. government funds including corporate funds (r). The foregone returns are weighted according to percent make-up of government funds yielding a social interest rate.

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[1]Derived from the *Federal Reserve Bulletin,* January, 1977.

[2]Ibid.

[3]This approach implies that returns can be invested at this same rate.

Table D.1

Nonfinancial Domestic Flow of Funds

Item	Billions (1975 dollars)*	Percent of total	Rate†	Weighted cost
Direct lending in credit markets:				
1. U.S. Gov. securities	41.0	26.4	6.98	1.84
2. State and local	9.6	6.2	7.05	.44
3. Corporate bonds	7.2	4.6	9.57	.44
4. Commercial paper	2.7	1.7	6.33	.00
Time and savings deposits	94.7	61.0	6.50	3.96
TOTALS	155.2	100.0		6.80

Federal Reserve Bulletin, Table, p. A45.

†Ibid., Table, p. A10.

Duffield (1978) recently utilized this approach to estimate a social interest rate. His analysis is repeated below nearly verbatim.

Combined data for 1975-1976 indicates that 15.8 percent of the U.S. government expenditures was based on borrowing, 24.9 percent was based on corporate income tax, and 59.3 percent was based on personal income tax.[4] The cost of long-term government debt is around 7 percent.

[4]Derived from the *Federal Reserve Bulletin*, January, 1977. Social insurance taxes (social security and unemployment) are assumed

The opportunity cost of corporate funds is as previously discussed (about 15 percent). A calculation for the foregone return on investment by private individuals is more complex. One approach is to look at the flow of funds for private, domestic nonfinancial investors as shown in Table D.1.

The rate on time and savings deposits is a best guess in view of savings rates varying from 4 1/4 percent to 7 3/4 percent depending on the time period of commitment. Over one half of all savings and loans, mutual savings banks, and commercial bank savings are some sort of time deposit arrangement beyond the usual 90-day withdrawal. The above figures give the weighted return on incremental investment for 1975 or a marginal cost; this is more appropriate than the average cost approach (which nonetheless is used for government borrowing and currently imparts a downward bias).

Private investors also face investment alternatives in less liquid assets, primarily real estate, stocks, and consumer durables. A total wealth estimate for 1972 provides the asset shares shown in Table D.2. It is assumed that returns on real estate investment are at least equal to the interest costs homeowners pay on mortgages. The rate given is for 1975. Returns on corporate stock and liquid assets are as previously discussed. From this calculation an estimate of the opportunity cost of foregone investment implied in personal income taxes is 9.8 percent.

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equally shared by corporate and personal; excise and miscellaneous taxes are lumped in personal.

Table D.2
Private Asset Shares

Item	Billions ($)*	Percent	Return (1975)	Weighted cost
Real estate	1,403.7	43.1	8.75†	3.77
Corporate stock	870.9	26.7	15.00	4.01
Liquid assets	984.3	30.2	6.80	2.05
Misc. other	1,096.0			____
TOTALS (excluding misc. other)	3,258.9			9.83

Statistical Abstract of the U.S., 1975. Liquid assets include an insignificant amount of cash.

†*Federal Reserve Bulletin*, p. A41, contract rate on mortgages, 1975.

This estimate is also appropriate for foregone consumption which is theoretically measured as the rate of time preference. The best estimate of marginal consumption returned is that individuals are bribed to save up to a certain point (and no more) by a 9.8 percent return on savings. Returns on consumption must equal or exceed the rate for extra savings. Given that Regulation Q has provided an upper limit to the legal rate on time deposits, this 9.8 percent is probably conservative.

These estimates for the opportunity costs of federal borrowing, and corporate and private income tax can now be used to calculate a weighted average cost of United States funds. The weighted average, 10.6 percent, represents the social interest rate (i).

Source of federal funds (1975-1976 average)	Percent	Rate	Weighted cost
Borrowing	.158	7.0	1.1
Corporate tax	.249	15.0	3.7
Personal tax	.593	9.8	5.8
TOTAL			10.6

The fraction of air pollution control investment that comes from corporate (α) and public ($1 - \alpha$) funds can be estimated by examining federal tax laws. Federal tax laws currently allow various tax credits on air pollution control equipment. Among these credits are (1) a special five-year (rapid) amortization which permits up to 20 percent of the installed cost to be written off each year, (2) a direct tax credit of 10 percent if normal depreciation is used, and (3) a direct five percent tax credit if the five-year rapid amortization is used.[5] These tax credits result in a certain amount of an air pollution control investment being funded by the public in the form of tax forgiveness.

Ruby (1978) has estimated a linear relationship between that portion of the present value of an air pollution control investment that is paid by the public ($1 - \alpha$) and the nominal discount rate (ρ).

$$1 - \alpha = .50 - 1.17\rho \qquad [D.2]$$

or

$$\alpha = .50 + 1.17\rho \qquad [D.3]$$

[5]The *Federal Register*, March 21, 1979, pp. 17310-17342 detail current tax credits for air pollution investments.

This equation assumes a 30-year time horizon,[6] a marginal corporate income tax rate of 48 percent and use of the rapid amortization option.

Substituting this equation into the Sjaastad and Wisecarver (1977) equation, along with the estimated values for the corporate opportunity cost (r = 15 percent) and the social interest rate (i = 10.6 percent) yields a 1976 nominal discount rate (ρ) of 13.4 percent.

This nominal rate includes an inflation premium that must be extracted. The following equation converts the nominal rate to a real discount rate.

$$\lambda = \frac{1 + \rho}{1 + I} - 1 \qquad [D.4]$$

where: I = rate of inflation

λ = real discount rate.

Ideally, the desired I value is the rate of inflation perceived by the public as most likely to occur in the future. One must fall back upon historical averages because inflation is not easily predicted. Table D.3 provides the historical rates of inflation and the corresponding real discount rate resulting from each rate of inflation. Because the nominal rates derived above correspond to 1976, the extracted inflation premium must also be based on the historical rates of inflation as perceived by the public in 1976. A more recent nominal rate was not derived under the assumption that the real-nominal relationship has been stable.

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[6]This corresponds to the intermediate of the time horizons used in this analysis. Relative to the rate of inflation, the sensitivity of a 20- or 40-year horizon in equation D.3 is negligible.

Table D.3

Historical Rates of Inflation and the Resulting Real Discount Rate

Averaging	Rate of inflation* (%)	Real discount rate at each inflation rate (%)
1965-1976	5.2	7.8
1970-1976	6.5	6.5
1974-1976	8.6	4.4

*As measured by the Consumer Price Index in the *Monthly Labor Review*, April, 1977, p. 109.

The sensitivity of the real discount rate to the assumed inflation premium is evident from Table D.3. For purposes of this analysis the extreme values of the reasonable alternatives, four percent and eight percent, are used to annualize air pollution control costs.

Given the discount rate (λ) and the time horizon (n), the air pollution capital costs (C) can be annualized into average annual costs (A) by the following capital recovery formula:

$$A = C\left[\frac{\lambda(1+\lambda)^n}{(1+\lambda)^n - 1}\right]. \qquad [D.5[$$

Since the effect of a lower discount rate and a longer time period is to reduce the annual costs, the four percent and 40-year parameters were used to generate low estimates of annual control costs. The amortization factor in this case is .05052, i.e., the control equipment cost multiplied by .05052 equals the annual cost. For the high estimate (λ = 8.0 percent; n = 20 years) the amortization factor is .10185.

Evident from the amortization factors is the sensitivity of the annual costs to the discount rate and time horizon. A full twofold difference results from variations in these parameters.