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Methods of Valuing Air Pollution and Estimated Monetary Values of Air Pollutants in Various U.S. Regions

by M.Q. Wang, D.J. Santini, and S.A. Warinner

Center for Transportation Research, Energy Systems Division,
Argonne National Laboratory, 9700 South Cass Avenue, Argonne, Illinois 60439

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December 1994

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NOTATION

AFV	alternative-fueled vehicles
AQVM	air quality valuation model
BACT	best available control technology
BARCT	best available retrofitted control technology
BPA	Bonneville Power Administration
CAA	Clean Air Act
CAAA	Clean Air Act Amendments
CARB	California Air Resources Board
CEC	California Energy Commission
CFC	chlorofluorocarbon
CH_4	methane
CNG	compressed natural gas
CO	carbon monoxide
CO_2	carbon dioxide
COI	cost of illness
CV	contingent valuation
EKMA	environmental kinetic modeling approach
EPA	U.S. Environmental Protection Agency
EV	electric vehicle
FFV	flexible fuel vehicle
FHWA	Federal Highway Administration
GV	gasoline vehicle
IEP	Independent Energy Producers
I/M	inspection and maintenance
LEV	low-emission vehicle
MDPU	Massachusetts Department of Public Utilities
MSA	metropolitan statistical area
MW	megawatt
N_2O	nitrous oxide
NAAQS	national ambient air quality standards
NERA	National Economic Research Associates
NMHC	nonmethane hydrocarbon
NMOG	nonmethane organic gases
NO_2	nitrogen dioxide
NO_x	nitrogen oxides
NPC	National Petroleum Council
NYSEO	New York State Energy Office
OBD	on-board diagnostic system
O_3	ozone
O&M	operation and maintenance
OPUC	Oregon Public Utility Commission
ORVR	on-board refueling vapor recovery system
OTA	Office of Technology Assessment
PM	particulate matter
PM_{10}	particulate matter (less than 10 micrometers in diameter)
pop	population
PSCN	Public Service Commission of Nevada

PUC	public utility commission
PV	present value
RER	Regional Economic Research, Inc.
RFG	reformulated gasoline
ROG	reactive organic gases
RVP	Reid vapor pressure
SCAQMD	South Coast Air Quality Management District
SCE	Southern California Edison Company
SO_2	sulfur dioxide
SO_x	sulfur oxides
SRI	Sierra Research, Inc.
TCM	transportation control measure
TLEV	transitional low-emission vehicle
TSP	total suspended particulate matter
ULEV	ultra low-emission vehicle
VMT	vehicle miles traveled
VOC	volatile organic compound
WTA	willingness to accept
WTP	willingness to pay
ZEV	zero-emission vehicle

ACKNOWLEDGMENTS

This work is sponsored primarily by the U.S. Department of Energy, Assistant Secretary for Policy, Office of Environmental Analysis and Sustainable Development, under contract W-31-109-ENG-38. Additional funding for publishing this report was provided by the U.S. Department of Energy, Assistant Secretary for Energy Efficiency and Renewable Energy, Office of Alternative Fuels.

We are grateful to the following individuals for providing information regarding various studies: S. Bernow of Tellus Institute; S. Buchanan of Bonneville Power Administration; P. Carver of Oregon Public Utility Commission; E. Caverhill of Resource Insight; E. Chang and S. Lieu of South Coast Air Quality Management District of California; R. Buell, S. Chaudry, and J. Diamond of California Energy Commission; S. Chaitkin of California Public Utility Commission; M. Prichard of Houston Lighting & Power; G. Schilberg of JBS Energy, Inc.; G. Fry of Massachusetts Department of Public Utilities; T. Henderson of Public Service Commission of Nevada; and S. Putta of New York State Department of Public Service.

We sincerely thank the following individuals for making comments on and suggestions about the draft of this report: M.F. Lawrence of Jack Faucett Associates; S. Lieu of South Coast Air Quality Management District; J.M. Loyer of California Energy Commission; D. Rote of the Center for Transportation Research, Argonne National Laboratory; and J.H. Suhrbier of Cambridge Systematics, Inc. We are also grateful to M. Fitzpatrick for her skillful editing of this report.

We are solely responsible for the contents and conclusions of this report.

METHODS OF VALUING AIR POLLUTION AND ESTIMATED MONETARY VALUES OF AIR POLLUTANTS IN VARIOUS U.S. REGIONS

by

M.Q. Wang, D.J. Santini, and S.A. Warinner

ABSTRACT

Air pollutant emission values are used to determine the social costs of various technologies that cause air pollution and to estimate the benefits of emission control technologies. In this report, we present two methods of estimating air pollutant emission values — the damage value method and the control cost method—and review 15 recent studies in which these methods were employed to estimate emission values. The reviewed studies derived emission values for only a limited number of areas; emission value estimates are needed for other U.S. regions. Using the emission values estimated in the reviewed studies, we establish regression relationships between emission values, air pollutant concentrations, and total population exposed, and apply the established relationships to 17 U.S. metropolitan areas to estimate damage-based and control-cost-based emission values for reactive organic gases, nitrogen oxides, particulate matter measuring less than 10 microns, sulfur oxides, and carbon monoxide in these areas. Our estimates show significant variations in emission values across the 17 regions.

SUMMARY

Quantifying the monetary value of air pollutant emissions has become increasingly important because of the need to determine the social costs of various technologies that cause air pollutant emissions and to estimate the monetary benefits of emission control technologies. Although emission values in attainment areas might be treated as being zero (unless the area could fall into nonattainment if emission controls were not maintained), emission values in nonattainment areas cannot be zero. Therefore, emission values *must* be chosen to evaluate the societal costs and benefits of the projects that cause air pollutant emissions. Yet emission value estimates are lacking in many regions; even for the regions where these estimates have been made, they are subject to many uncertainties. In this report, we present methods for estimating emission values, review previous studies on emission valuation, and estimate emission values for regions where these values have not yet been developed.

Two general methods can be used to estimate air pollutant emission values: damage value and control cost. The damage value method, used to estimate the monetary cost of damages caused by air pollutant emissions, involves seven steps: (1) identifying emission

sources, (2) estimating emissions, (3) simulating air pollutant concentrations in the atmosphere, (4) estimating exposure of humans and other objects to air pollutant concentrations, (5) identifying the physical effects of air pollutant concentrations on humans and objects, (6) completing an economic valuation of physical effects, and (7) calculating dollars-per-ton emission values.

The control cost estimating method is based on the presumption that emission standards or air quality standards are established at the ideal level — where the marginal damage of air pollution is equal to the marginal control cost. In this approach, it is assumed that the cost required to meet predetermined air quality standards imposed by legislators "reveals" the value society places on the emissions being controlled. Therefore, the estimated marginal control cost to meet air quality standards represents the marginal damage value of air pollution when air quality standards are met. Two major steps are involved in the control cost method: (1) identifying the marginal control measures required to meet predetermined air quality standards and (2) estimating the dollars-per-ton cost for each identified control measure.

The damage value method, which directly estimates emission values, seems theoretically sound. However, in practice, the method suffers from necessary assumptions and simplifications and from tremendous uncertainties involved in each estimating step. The cumulative effect of these uncertainties is to reduce the accuracy of the estimated damage values. Studies based on the method cannot practically include all potential adverse air pollution effects in estimating damage values; some effects are usually excluded, and consequently, damage values are underestimated. There are some scientific disputes concerning the validity of the method and its reliability. Many analysts outside the discipline of economics are critical of the damage value estimating method — philosophical uneasiness results when economists place dollar values on such intangibles as human life and human discomfort. Also, because of complex methods involved in each of the estimating steps, use of the damage value estimating method is time-consuming and resource-intensive. Consequently, the control cost method has been used more frequently than the damage value method to estimate air pollution emission values.

The control cost method involves fewer estimating steps, assumptions, and resources; can generate cost estimates more quickly; and does not require highly specialized expertise to construct emission value estimates. However, the method is based on the fundamental assumption that legislators and/or regulators establish emission and air quality standards solely on the basis of the marginal damages and the marginal control costs of meeting the standards. In reality, establishment of emission and air quality standards is a highly political process; economic implications are only one of many factors considered. On the basis of strict economic theory, it is improper to treat the estimated marginal control costs as the value for emission damages. Nevertheless, the calculated control cost represents the opportunity cost of meeting the standards. If new, less costly control systems can be developed, the most costly measures can be avoided. It is the "avoided opportunity cost" that this report adopts. However, we do not take the position that emission damage values are accurately represented by the estimated emission control costs. In many cases, emission damage values can differ significantly from control costs. Thus, control costs cannot represent damage values.

In the past several years, various studies were conducted to estimate air pollutant emission values in California, Oregon, Nevada, and the northeastern United States. These studies, conducted using the damage value and/or the control cost estimating method, have frequently been cited and used by various public and private organizations without careful consideration of their methodologies and assumptions. In this report, we review the methodologies, assumptions, and results of past major studies on emission value estimation. Six of the studies were conducted by using the damage value method. When estimating emission values, these studies usually considered current air quality status and added power plant emissions to the study areas — in a sense, estimating emission values under the current air quality status. The six damage-based studies resulted in very large differences in emission values among various regions. The differences are caused by air pollutant concentrations, population exposed, and methods and assumptions used. Differences in methods and assumptions often cause significant differences in the values estimated, contaminating the estimated emission values for comparison purposes.

Nine of the studies we reviewed used the control cost estimating method. Like the damage-based studies, the control-cost-based studies showed wide variations in emission values among various regions. The variations are caused primarily by the marginal control technologies selected, which are determined by air quality status in a region. Emission values estimated by various studies for the same region also vary significantly, because of the different control technologies selected and assumptions regarding the costs and emission reductions of the selected technologies.

We also found large discrepancies between the damage-based and control-cost-based emission values; damage-based estimates are generally, but not always, lower. Because of exclusions of certain air pollutant effects and the simplifying assumptions involved in these studies, we believe that damage-based estimates under-represent actual emission values, rather than that emissions are over-controlled. However, given our perspective (that the damage value method usually underestimates emission values), we believe that a damage-value-based estimate that is higher than a control-cost-based estimate implies under-control by current regulations. Damage-based values for PM_{10} (particulate matter less than 10 micrometers in diameter) are actually higher than control-cost-based values, implying that PM_{10} regulations may not have been strict enough.

The reviewed studies estimated emission values only for a limited number of areas. In adopting or proposing emission values, public and private organizations often apply the emission value estimates for previously studied areas directly to their own areas, without any adjustments to reflect the differences in air quality status and total population between their regions and the previously studied regions. Because of these differences, emission values can and should differ significantly among regions.

In order to generate region-specific emission values, damage value models should ideally be run for a particular region to estimate damage values, and emission control costs should be estimated by taking into account the control measures and their costs applied to the region. However, limited resources may prevent such detailed, accurate estimates for individual regions.

In this report, using the emission values estimated in previous original studies, we established regression relationships between emission values, air pollutant concentrations, and total population. Our estimated regression relationships generally take logarithmic forms. We applied the established regression relationships to 17 U.S. metropolitan areas to estimate emission values for these areas. Although our regression-based values may not be as accurate as the estimates made by using the damage value or the control cost method for a particular area, they are more accurate than direct application of the emission value estimates for other areas to the study area.

To allow the flexibility of choosing between damage-based and control-cost-based emission values, we established two sets of regression relationships — one for estimating damage-based values and the other for estimating control-cost-based values. We estimated damage-based values for nitrogen oxides (NO_x), reactive organic gases (ROG), PM_{10}, and sulfur oxides (SO_x), and control-cost-based values for NO_x, ROG, PM_{10}, SO_x, and carbon monoxide (CO). Table S.1 presents our regression-estimated emission values for the 17 metropolitan regions. Note that original estimates of emission values are already available for seven of the regions (Boston, Las Vegas, Los Angeles, New York, Sacramento, San Diego, and the San Francisco Bay area). Our purpose in developing estimates for these areas is to compare our regression-based estimates with the original estimates.

Regression-estimated emission values vary significantly across the 17 areas. In particular, per-ton damage-based emission values vary from \$910 to \$9,800 for NO_x, \$320 to \$5,110 for ROG, \$2,450 to \$17,200 for PM_{10}, and \$2,190 to \$3,970 for SO_x. Control cost-based emission values vary from \$5,220 to \$21,850 for NO_x, \$5,100 to \$19,250 for ROG, \$2,400 to \$6,060 for PM_{10}, \$3,130 to \$13,480 for SO_x, and \$1,410 to \$4,840 for CO. Emission values in Los Angeles are always high, while those in Las Vegas are usually low. Estimated damage-based values are lower than estimated control-cost-based values for each pollutant except PM_{10} — probably because of underestimation of damage values in previous original studies, in which not all air pollution effects were considered. However, when the differences are extremely large, it is certainly possible that control cost estimates are too high.

We also reviewed past studies estimating greenhouse gas emission values. For these studies, researchers generally estimated emission values for carbon dioxide (CO_2) by considering various options for controlling CO_2 emissions. Emission values for other greenhouse gases were usually calculated on the basis of the estimated CO_2 value and the global warming potentials of other greenhouse gases relative to that of CO_2. On the basis of these studies, we suggest a per-ton value of \$15 for CO_2, \$150 for methane (CH_4), \$2,700 for nitrogen oxide (N_2O), \$33 for CO (as a greenhouse gas), \$105 for nonmethane organic gases (NMOG) (as a greenhouse gas), \$210 for NO_x (as a greenhouse gas), \$19,500 for chlorofluorocarbon (CFC)-11, and \$55,500 for CFC-12.

TABLE S.1 Estimated Emission Values for 17 U.S. Regions

Area	Emission Value ($/ton, 1989 dollars)				
	NO_x	ROG	PM_{10}	SO_x	CO
	Damage-Based				
Atlanta	4,330	2,150	5,170	2,720	N/A
Baltimore	4,430	2,210	4,520	2,620	N/A
Boston	4,120	2,030	5,090	2,820	N/A
Chicago	5,380	2,700	10,840	3,600	N/A
Denver	2,840	1,350	3,390	2,330	N/A
Houston	6,890	3,540	5,190	2,910	N/A
Las Vegas	910	320	2,450	N/A[a]	N/A
Los Angeles	9,800	5,110	17,200	3,970	N/A
Milwaukee	3,890	1,930	2,960	2,210	N/A
New Orleans	3,880	1,910	3,600	2,471	N/A
New York	7,130	3,650	15,130	4,030	N/A
Philadelphia	5,940	3,010	8,360	3,340	N/A
Sacramento	3,870	1,920	3,150	2,190	N/A
San Diego	5,510	2,800	4,800	2,600	N/A
San Francisco Area	3,730	1,810	5,970	2,970	N/A
San Joaquin Valley	4,490	2,240	6,550	2,610	N/A
Wash., D.C.	4,900	2,450	6,260	3,070	N/A
	Control-Cost-Based				
Atlanta	9,190	8,780	3,460	6,420	2,280
Baltimore	10,310	9,620	3,170	5,600	2,490
Boston	7,980	7,850	3,120	5,060	1,610
Chicago	7,990	8,150	4,660	9,120	2,440
Denver	6,660	6,590	2,790	4,900	2,960
Houston	17,150	15,160	2,780	3,590	2,680
Las Vegas	5,220	5,100	4,190	11,650	2,770
Los Angeles	21,850	19,250	6,060	13,480	4,840
Milwaukee	11,350	10,250	2,560	4,380	1,590
New Orleans	9,190	8,670	2,400	3,130	1,410
New York	12,340	11,720	5,390	11,090	3,910
Philadelphia	11,360	10,730	4,040	7,330	3,160
Sacramento	11,350	10,240	2,950	5,800	3,040
San Diego	14,110	12,630	3,460	6,640	2,740
San Francisco Area	5,230	5,760	3,200	4,900	2,460
San Joaquin Valley	10,310	9,630	5,110	12,480	2,750
Wash., D.C.	9,190	8,910	3,340	5,320	3010

[a] N/A = not available.

Emission value estimates made in past studies were primarily for stationary source emissions, so the regression estimates based on past studies are applicable to stationary source emissions. Application of emission values estimated in this report to mobile source emissions may result in underestimation of the true values of mobile source emissions, simply because the highest mobile source emission concentrations generally occur in metropolitan areas, where population exposure is high, while stationary source emissions (power plants and manufacturing plants) often occur in less populated areas. The key exception is emission values of greenhouse gases, for which no differences between stationary and mobile source values should be observed.

Because our regression relationships rely on original estimates, we recommend that original estimated emission values be used for relevant areas when available. Our purpose here is not to supplant a more careful study, but to provide working values until studies using the damage value method or the control cost estimating method are completed in the regions for which no estimates have yet been prepared.

1 INTRODUCTION

Since passage of the Clean Air Act Amendments (CAAA) in 1970, U.S. legislators and regulatory agencies have made continuous efforts to reduce air pollution. Consequently, the air in many U.S. urban areas is much cleaner than it would otherwise be. Still, 96 U.S. metropolitan areas violate the federal ambient ozone standard, and 41 violate the federal ambient carbon monoxide (CO) standard (U.S. Environmental Protection Agency [EPA] 1993). Various control measures and strategies are proposed to reduce air pollutant emissions to meet air quality standards in these areas.

In selecting emission control measures or strategies, we need to estimate and compare the benefits and costs of various measures to ensure that those that will achieve the greatest net benefits are implemented first. The costs of control measures can be calculated by taking into account capital, operation, maintenance, and other cost components. Benefits can be calculated on the basis of emission reductions and dollar values per unit of emissions. In order to complete an economic cost-benefit analysis of various control measures, the cost of air pollutant emissions must be quantified.

Estimating air pollution values is also essential in determining the social costs of various technologies. Some technologies may have higher private costs (costs paid by private users) but lower social costs, which include private costs and such externalities as the cost of air pollution. Society should promote technologies that have lower social costs. For example, in recent years, various state public utility commissions (PUCs) began to incorporate environmental externalities in their calculation of the cost of electric power production. The calculated social costs are used in PUCs' resource planning and acquisition process for promoting the use of clean technologies and renewable energy sources. In the transportation sector, various clean transportation technologies to reduce air pollution have been proposed. These technologies usually have high private costs, but their lower social costs may justify their use. To evaluate various transportation technologies from a more complete social-cost-accounting point of view, the externality costs of air pollution must be considered.

Two general methods have been developed to estimate emission values: damage value and control cost. Both methods have been used in past studies to estimate emission values. However, because of a lack of understanding regarding the theoretical background of the methods, there are some conceptual confusions about each. Without a complete understanding of the advantages and disadvantages of each method, people sometimes make premature judgments concerning the use of one method over the other. In Section 2, we present the theoretical background of emission damages and emission control costs and discuss the procedures, assumptions, and uncertainties associated with each of the two methods. The theoretical discussion is intended to eliminate some conceptual confusion regarding emission value estimation and to provide sufficient information to assist researchers in choosing one method or the other.

In recent years, a number of studies have been conducted to estimate air pollutant emission values in some U.S. regions. Each study used the damage value and/or the control cost estimating method, made assumptions in the estimation process, and had its limitations. Results of some studies have been widely cited without carefully considering the studies' assumptions and limitations. This is probably occuring because the need for emission values is urgent, yet few original estimates of emission values have been made. In Section 3, we review past studies on emission value estimation, present major assumptions involved in each reviewed study, and present study results. Our review of past studies also provides historical background on emission value estimation.

Despite past efforts to estimate emission values, estimates are still lacking for many U.S. metropolitan areas. As Section 2 will show, completing original estimates of emission values by using either the damage value or the emission control cost estimating method is not an easy task. Consequently, emission values estimated for one region are commonly applied to different areas, with no adjustment for differing air pollution levels, populations, and other objects affected. In Section 4, using the emission values estimated in previous studies, we establish emission values as regression functions of air pollution levels and total population exposed. We use the established functions to estimate emission values for some U.S. metropolitan areas where emission values have not been developed. In this way, emission values estimated in one area are adjusted for use in another area on the basis of air pollution levels and population. Section 5 presents our conclusions.

2 BACKGROUND AND ESTIMATING METHODS

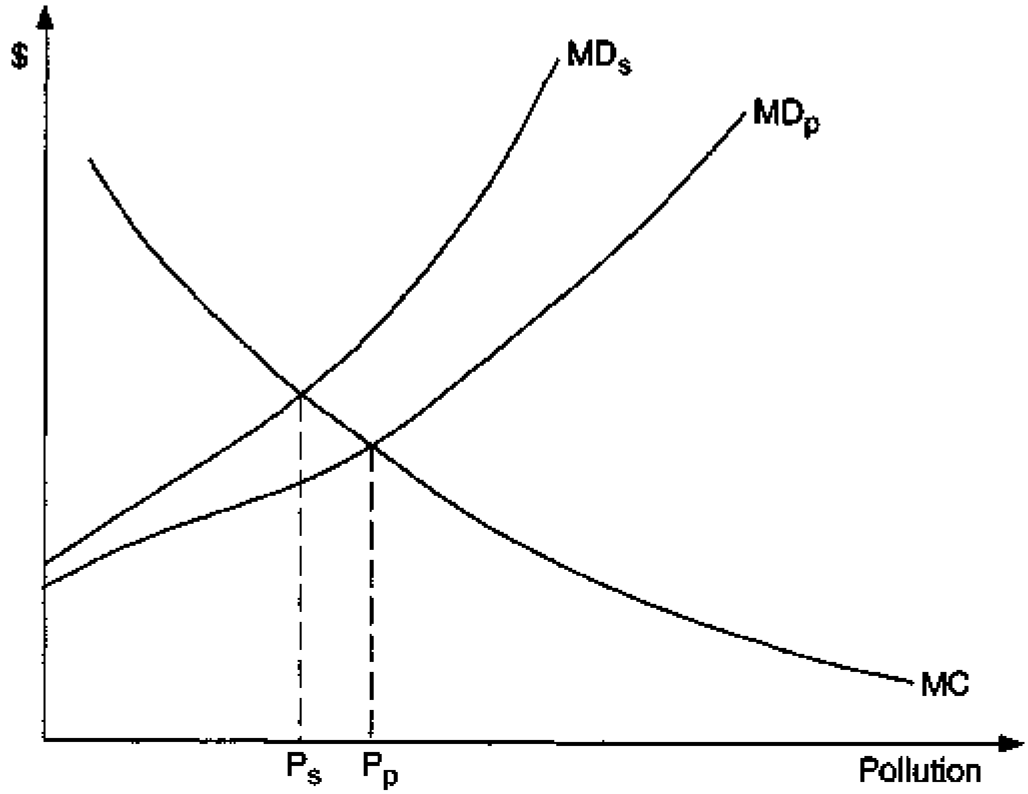
2.1 MARGINAL DAMAGE VALUES AND MARGINAL CONTROL COSTS

Air pollution, created in association with the activities of industries or individuals, causes damages to human health, agricultural crops, ecosystems (e.g., forests and lakes), materials (e.g., buildings and houses), and natural scenery (e.g., visibility). In a classic example of externality, the full costs of air pollution are not borne by those who generate the pollution. On the other hand, the full benefits of reducing air pollution are not exclusive to those who make efforts to reduce it. Without government intervention, the firms or individuals who cause air pollution may not want to install control measures. In the absence of control measures, air pollution persists, and society as a whole bears its damages, directly or indirectly. Because of the difference in air pollution damages to polluting parties and to society, the polluting parties tend to produce air pollution at the private optimal pollution level, which exceeds the socially optimal pollution level (Figure 1).

Two general approaches can be taken to reduce the private optimal air pollution levels to more closely match the socially optimal levels. One approach requires parties to meet air pollution standards that are established to be close to the socially optimal level. This approach is known as the command and control approach. The Clean Air Act (CAA), which is the legal basis of U.S. air pollution control policies, relies primarily on this approach. However, although the CAA considers health and property damages in establishing standards, it generally does not consider the costs of achieving the standards. Thus, the command and control standards in the CAA are generally not established at the socially optimal levels of air pollution.

The other approach involves applying charges so that polluting parties expend additional cost and effort to control pollution beyond their private optimal levels. A critical prerequisite of this approach is estimation of the external costs of pollution, which is a difficult task. Using this approach, if charges were set at the cost of air pollution, polluting parties would encounter the social marginal damage of air pollution. This approach, which is often referred to as the economic approach, has been advocated in the economic community. Researchers and economists have constructed estimates of the dollar value of damages from individual pollutants. Because it imposes air pollution charges, the economic approach is often unpopular with industry; environmentalists may also oppose the approach because, in most cases, it will result in smaller total emission reductions.

Several concepts regarding emission control costs and emission damages need to be discussed in order to put the emission value estimates into proper perspective. Figure 2 illustrates the total damage of air pollutant emissions and the total cost of achieving a given emission level. In the figure, E_0 represents the emission level without emission control; E_C represents the current emission level; and E_S represents the ideal emission standard. Although E_S is intended to be set at the level at which marginal cost equals marginal



MC: Marginal control cost of air pollution (cost, to polluting parties and society of controlling one unit of air pollution at a given air pollution level)

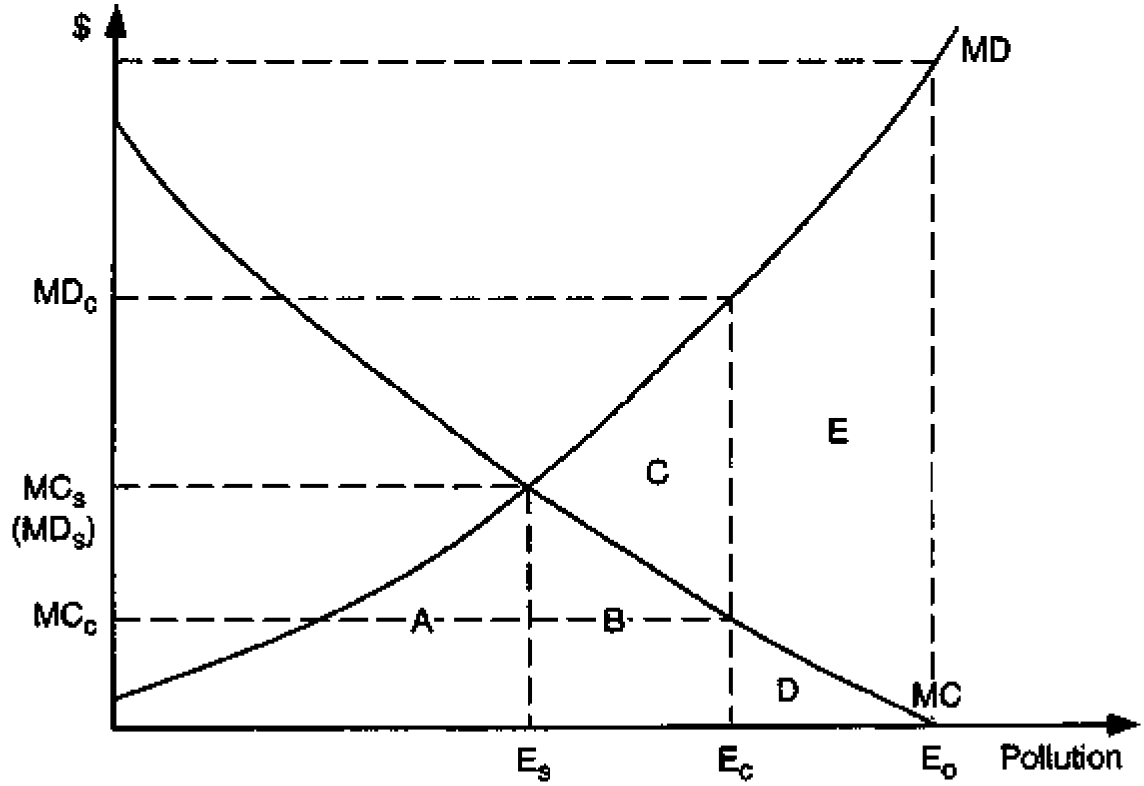
MD_p: Private marginal damage of air pollution (damage, to polluting parties, of one unit of air pollution at a given air pollution level)

MD_s: Social marginal damage of air pollution (damage, to society, of one unit of air pollution at a given air pollution level)

P_p: Private optimal level of air pollution

P_s: Socially optimal level of air pollution

FIGURE 1 Socially Optimal vs. Private Optimal Pollution Levels



E_c: Current emission level

E_s: Ideal emission standard

MC: Marginal control cost curve

MD: Marginal damage value curve

MC_c: Marginal control cost at current emission level

Md_c: Marginal damage value at current emission level

MC_S: Marginal control cost at ideal emission standard

Mds: Marginal damage value at ideal emission standard

FIGURE 2 Control-Cost-Based and Damage-Based Estimates at Three Emission Levels

damage, in practice, standards seldom fall at this level, because the authors of the CAA did not generally consider air pollution control costs when establishing standards and because many uncertainties are involved in estimating costs and damages.

By definition, at a given level of emissions, the marginal control cost is the cost of controlling one additional unit of emissions; the marginal damage is the damage of one additional unit of emissions; the average control cost is the total control cost spent divided by the total amount of emissions reduced; and the average damage is the total damage caused by emissions divided by the total amount of emissions. Specifically, in Figure 2, at the current emission level (E_C), the marginal control cost is MC_C; the marginal damage is MD_C; the average control cost is area D divided by (E_0 - E_C); and the average damage is the sum of areas A, B, and C divided by E_C. At the ideal emission standard, the marginal control cost (MC_S) equals the marginal damage (MD_S); the average control cost is the sum of areas B and D divided by (E_0 - E_S); and the average damage is area A divided by E_S.

Usually, less expensive control measures are implemented first, resulting in upward marginal control cost curves as more emissions are controlled. Because of this, the marginal control cost always exceeds the average control cost. The marginal damage usually increases as the amount of emissions to the atmosphere increases; consequently, the marginal damage always exceeds the average damage. If the emission level is reduced from the current level (E_C) to the ideal emission standard (E_S), the marginal control cost is increased from MC_C to MC_S and, consequently, the average cost is increased. However, at the same time, the marginal damage is decreased from MD_C to MD_S, and consequently, the average damage is decreased. Therefore, to examine emission values estimated in a study, it is important to identify the level of emissions at which emission control costs and emission damage values are estimated. As the figure shows, if the standard is not ideal, or if the standard is ideal but emissions are not yet equal to the standard, marginal costs and marginal damages will not be equal.

For illustration purposes, let us assume that, at the emission standard of E_S, the marginal damage of emissions equals the marginal emission control cost (as we implied, this assumption itself is questionable). First, at the uncontrolled emission level (E_0), industries do not expend any resources for emission control, so the total emission control cost is zero. The total damage caused by emissions is the area under the marginal damage curve and left of E_0 (the sum of areas A, B, C, D, and E). The sum of both the emission control cost and emission damage is the sum of areas A, B, C, D, and E.

Second, at the current emission level (E_C), the total emission control cost is area D. The total emission damage is the sum of areas A, B, and C. The sum of the emission control cost and emission damage is the sum of areas A, B, C, and D. The net savings to society from reducing the uncontrolled emission level (E_0) to the current emission level (E_C) is area E.

Third, at the emission standard (E_S), the total emission control cost is the sum of areas D and B. The total emission damage is area A. The sum of the emission control cost and emission damage is the sum of areas A, B, and D. The net savings to society from reducing the current emission level (E_C) to the emission standard (E_S) is area C, and the net savings from reducing the uncontrolled emission level (E_0) to the emission standard (E_S) is the sum of areas C and E.

As the figure illustrates, even if the emission standard is met, emission damage is not completely eliminated. Rather, at the emission standard, the sum of the emission control cost and emission damage is minimized. Note that further controlling emissions to below the ideal emission standard causes increases in the sum of the emission control cost and emission damage (again, assuming the marginal control cost equals the marginal damage at the emission standard — i.e., the standard is set at the "ideal" value).

The above illustration indicates some important consequences of estimating emission control costs and emission damages. The estimated marginal control cost at the current emission level is lower than that at the emission standard, while the estimated marginal damage at the current emission level is higher than that at the emission standard. The estimated average control cost at the current emission level is lower than that at the emission standard, while the estimated average damage at the current emission level is higher than that at the emission standard. The average emission control cost is always lower than the marginal emission control cost, and the average emission damage is always lower than the marginal emission damage. Therefore, it is important in an emissions study to know what items are estimated (control costs or damage values, and marginal or average) and at what emission level (the current emission level or the emission standard).

2.2 METHODS OF ESTIMATING AIR POLLUTANT EMISSION VALUES

Two general methods can be used to estimate air pollutant emission values: damage value and control cost. The procedures and assumptions involved in each of the two methods are described in detail below.

2.2.1 Damage Value Method

The damage value estimating method is used to estimate the monetary value of damages caused by air pollutant emissions. These estimated values directly represent the value of emission reductions by certain control measures. The method involves the following seven steps: (1) identification of emission sources, (2) estimation of emissions, (3) simulation of air pollutant concentrations in the atmosphere, (4) estimation of exposure of humans and objects to air pollutant concentrations, (5) identification of physical effects of air pollutant concentration on humans and objects, (6) economic valuation of physical effects, and

(7) calculation of dollars- per-ton emission values (Regional Economic Research, Inc. [RER], 1990).[1] These seven steps are discussed in detail below.

(1) Identification of Emission Sources. Emissions of air pollutants are produced through anthropogenic and natural processes (e.g., biogenic processes for volatile organic compound emissions, and storm and wind for particulate matter emissions). Anthropogenic emission sources — the major focus of past efforts on emission estimation — can be classified into point sources (where single points such as smoke stacks can be identified as emission sources) or area sources (where single point sources cannot be identified). By convention, all point sources are stationary sources. Examples include utility power plants, petroleum refineries, and industrial manufacturing facilities. Area sources can be either stationary or mobile. Stationary sources treated as area sources in emission models include residential, commercial, and small industrial sources. Mobile sources include highway motor vehicles, off-highway motor vehicles, locomotives, aircraft, and marine vessels. Emission sources in an air basin are usually identified by the air quality control authority in the basin.

(2) Estimation of the Amount of Emissions. Emissions from stationary sources are generally estimated using one of two methods. The first is based on reports from source operators who periodically sample emissions from their sources and report the data to air quality control authorities, which maintain the reported emission data. The second method involves estimating emissions on the basis of emission factors and activity levels. EPA has developed emission factors for many major emission sources; these are presented in its AP-42 documents (EPA 1992a). In developing emission factors, EPA relies on sampled emission data and applies emission reduction potentials for various control technologies. The agency periodically updates its emission factors, but because of limited emission sampling data and variations in source operating activities, emissions estimated using either method may not accurately represent actual emissions from stationary sources.

In estimating stationary source emissions, certain control technologies may be assumed based on the emission standards applicable to the sources. Emissions can be estimated at the current level, or can be projected for the future. In projecting future emissions, air quality standards that will be applicable in the future must be considered. Assumptions regarding what emission standards will apply, and whether and when the standards will be met are critical in projecting future emission levels.

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[1] Some other classifications of estimating steps may be used for the damage value method. For example, Hall et al. (1989) classified the method into four steps: (1) estimation of air quality, (2) estimation of human dosage of air pollutants, (3) estimation of human response to various levels of pollution, and (4) economic valuation of human response. Note that this definition leaves out effects on property, crops, and other plants and animals. Harrison et al. (1992) classified the method into five steps: (1) estimation of emissions, (2) simulation of air pollutant concentration, (3) estimation of exposure to air pollution, (4) identification of physical effects from exposure, and (5) valuation of effects. The classification developed by RER is used here because it is complete. The studies cited here did not include the last step, calculation of dollars-per-ton emission values; we added this step for our report.

Emissions from mobile sources are usually estimated using grams-per-mile emission factors of motor vehicles and their activity levels. EPA has developed a model (Mobile) to estimate vehicle emission factors. The most recent version of the model is Mobile 5 (EPA 1992b). The California Air Resources Board (CARB) has developed a counterpart model called EMFAC to estimate vehicle emission factors in California. The most recent version of the EMFAC model is EMFAC7G. Because limited amounts of vehicle emission testing results were available to establish the relationships used in Mobile and EMFAC, and because certain assumptions had to be made in the models, both Mobile and EMFAC are believed to underestimate actual on-road motor vehicle emission factors (National Research Council 1991). Because of the disparate nature of travel behaviors by individuals, projections of vehicle activities such as vehicle miles traveled (VMT) are often not accurate. Inaccurate estimates of both emission factors and vehicle activity levels make developing accurate projections of mobile source emissions difficult.

In summary, although various models and data sources are available for estimating air pollutant emissions, the accuracy of emission estimates is often questionable. Inaccurate estimated emission inventories are one of the reasons for discrepancies between predicted and observed air quality.

(3) Simulation of Air Pollutant Concentrations. Air pollutant concentrations in the atmosphere result from various processes, including pollutant dispersion, reaction, and residence, which are complicated by meteorology and topography. The products of the processes are primary and secondary air pollutants in the atmosphere. Primary pollutants are those that involve virtually no chemical reactions; they are produced directly from dispersion of air pollutants emitted from sources. Primary pollutants include CO, nitrogen dioxide (NO_2), sulfur dioxide (SO_2), and particulate matter (PM) from emitted particulates. Secondary pollutants are products of chemical reactions among the various air pollutants emitted into the atmosphere. Secondary pollutants include ozone (from the photochemical reaction of volatile organic compounds [VOCs] and nitrogen oxides [NO_x]), acid deposition (acids resulting from sulfur oxides [SO_x] and NO_x), and PM (resulting from SO_x, NO_x, and ozone).

The complicated processes of dispersion, reaction, and residence of air pollutants in the atmosphere result in non-linear relationships between emissions and air pollutant concentrations. Therefore, simulation of air pollutant concentrations involves sophisticated computer modeling, and requires detailed input data such as spatial and temporal distribution of emissions, meteorological and topographic data, and background pollutant concentration data, among other information. Unfortunately, accurate, detailed input data are often not available.

A common example of complex air quality modeling is the simulation of ozone concentration in the atmosphere. Ozone formation is determined not only by the amount, but also by the ratio, of VOCs to NO_x in the atmosphere. Depending on the ratio, control of either VOCs or NO_x could lead to a decrease or an increase in ozone concentration. To further complicate the simulation process, the VOC/NO_x ratio can vary significantly within an air basin, depending on the location of the emission sources, timing of emissions, and meteorological conditions (Finlayson-Pitts and Pitts 1993). To accurately predict ozone concentration, an air basin must be divided into many small grids, and emissions and meteorological parameter data must be collected for each grid. Estimation of grid-specific emissions is subject to many uncertainties.

Simulating the concentrations of primary pollutants can also be difficult. A typical example is simulation of CO concentrations. CO concentrations at a major intersection are often many times higher than at nearby locations. Simulation of CO concentration over a short period of time (e.g., one hour) predicts CO concentration values that are very different from those produced from simulation over a long period (e.g., eight hours). An accurate prediction of CO concentration (especially at the hourly level) requires micro-scale, short-time-period simulation, which, in turn, requires micro-scale, short-time-period input data. Accurate estimates of traffic flow changes in a micro scale, as well as micro-scale meteorology, are difficult.

Because of the uncertainties involved in emission inventory estimation and simulation of air pollutant concentrations, there are large discrepancies between projected and monitored air pollutant concentrations. Pollutant concentrations in many air basins that, in the past, were projected to meet the federal air quality standards now violate the standards.

(4) Estimation of Exposure by Humans and Objects to Air Pollution. Exposure models are used to predict human exposure to air pollution. These models usually track air pollution concentrations and outdoor, indoor, and in-transit human activities by demographic group. The models are generally based on the assumption that an individual's time-integrated exposure is the product of (1) the air pollutant concentrations in a specific set of micro-environments and (2) the time spent by the individual in those micro-environments. To make the models manageable, all members of a population are assumed to follow a limited number of daily activity patterns and receive almost identical exposures. Such aggregate exposure models are designed to quantify some major tendencies of exposure without considering the details of individual activities in various environments. Even using these simplifications, an exposure model can be complicated. For example, Hall et al. (1989) developed a regional human exposure model that includes 1,000 time-activity patterns for nine demographic groups. Input data to the model include population data by exposure district, mobility data, air quality distribution by hour of day and by exposure district, micro-environment air quality factors, time-activity patterns and frequency of occurrence by the demographic group, and other factors.

Choosing a method to simulate human exposure involves balancing the greater efficiency of rough, aggregated models against the accuracy of detailed models. Detailed models (such as the Hall et al. model) require tremendous amounts of input data on population, human activities, and air pollutant concentrations. These data may not be available in many regions. On the other hand, because of the complexity of human activities, rough, aggregated models may not accurately represent human exposure.

(5) Identification of Air Pollution Effects on Humans and Objects. The various types of air pollution damages include human health effects, materials damages, agricultural damages/effects, visibility effects, and physical/aesthetic effects. Human health effects include both human mortality and morbidity. To determine air pollution effects on human health, most researchers use risk assessment methods to generate dose-response relationships. The risk assessments are based on results from laboratory testing of animals' responses to pollutants, human clinical experiments, and epidemiological studies.

Animal studies can provide background data and hypotheses for human health effects. Results of animal studies may not be very useful to measure human health effects directly, because there may be no link between air pollution-induced effects on animals and those on humans. Moreover, animal studies often focus on fatal, acute diseases such as cancers, but ignore chronic diseases. Animal laboratory tests usually apply high pollutant concentrations to tested animals for a short period of time. Accuracy is questionable when the dose-response functions generated in this way are applied to humans who may be exposed to low pollutant concentrations for a long period of time. Most past air pollution damage studies have not used laboratory animal testing results.

In clinical studies, a particular group is separated into a control sub-group and a testing sub-group. The testing sub-group is exposed to various levels of air pollution. Lung function and functions of other human systems are measured and correlated with human symptoms. Air pollution effects can also be identified during the clinical studies by direct inquiries regarding the prevalence of certain symptoms such as cough, headache, sore throat, chest tightness, and eye irritation. Because only a very small number of individuals can be included in a clinical study, it is problematic to generalize results from a small studied group to the general population.

In epidemiological studies, a number of population samples are taken from environments with varying air pollution problems. These studies evaluate relationships between human symptoms and air pollution using the observed data on human symptoms and measured air pollution concentrations in different environments. A critical assumption in epidemiological studies is that people's activity patterns and activity levels are virtually the same in various environments. Designers of epidemiological studies also encounter difficulty in controlling factors other than air pollution to develop statistical relationships between human symptoms and air pollution.

Air pollution damages to vegetation (agriculture, forestry, and ornamental plants) include foliar injury, reduced yield, and slowed growth. Statistical methods are used to establish relationships between actual yield or growth of vegetation and air pollutant concentrations. Effects may be positive in some instances.

The types of materials subject to air pollution damages include construction metals, exterior paints, stone, masonry, concrete, textile, leather, paper, etc. Although laboratory and field studies can be conducted to estimate relationships between material damages and air pollution, in practice it is difficult to establish reliable relationships because moisture, sunlight, and other environmental factors complicate the process.

Because of the difficulty in identifying and quantifying air pollution effects, past air pollution damage studies do not include all applicable effects. Exclusion of some air pollution effects causes underestimation of actual air pollution damage values.

(6) Economic Valuation of Air Pollution Effects. Determining the monetary values that individuals place on adverse air pollution effects is a key element in estimating air pollutant values. The costs of adverse health effects are related to medical expenses, loss of work, discomfort, and inconvenience. Two general methods have been developed to estimate the monetary value of health effects: the cost of illness (COI) method and the willingness to pay (WTP) or willingness to accept (WTA) method (Hall et al. 1989). The COI method uses available data regarding actual health care costs and wages lost to estimate the direct costs of adverse health effects. Although these data may be readily available, it is difficult to accurately determine what portion of these aggregate costs is attributable to the adverse health effects caused by air pollution. Researchers disagree about what wage rates should be used and how premature death should be valued. Furthermore, the COI method cannot estimate values of non-market goods, such as discomfort and inconvenience, and it cannot measure diminished productivity.

WTP and WTA methods are derived from studies based on the hedonic approach or the contingent valuation (CV) approach. The hedonic approach uses observed behaviors to estimate values for the economic consequences of those behaviors. For example, if people accept a lower wage or pay a higher cost for a lower probability of exposure to the adverse health effects caused by air pollution, or demand higher pay to accept a higher probability of health effects, this provides a measure of how much air quality is worth to them. A critical assumption of the hedonic approach is that people are well aware of the health risk associated with the job that they select and that they are free, to a great extent, to select occupants from a variety of choices.

The CV approach relies on surveying a population sample, through a series of hypothetical questions, to determine the amount of money people are willing to pay (WTP) to avoid the risk of given adverse health effects or the number of dollars people are willing to accept (WTA) for taking the risk. One problem with the CV approach is that hypothetical questions generally elicit hypothetical answers that may not accurately represent what people actually value. This problem is particularly true when people do not face the economic constraints in answering survey questions that they face in reality.

In estimating the cost of adverse effects on vegetation, market dollar values of agricultural products can be directly assigned to the loss of agricultural production due to air pollution, because agricultural products are exchanged in the commodity market. Valuation of effects on non-agricultural vegetation is a little complicated. For forests, commercial timber value may be estimated from the timber market. The value of recreational use of a forest can be estimated using the CV method or the cost of travelling to an alternate recreational site.

Dollar values of material damages can be estimated from costs of increased maintenance and replacement of materials due to air pollution, or from costs of preventing or averting expected pollution effects.

Aesthetic consequences of air pollution are primarily visibility effects. Values of reduced visibility can be estimated using the CV method or the hedonic price approach.

(7) Calculation of Dollars-per-Ton Emission Damage Values. The above six steps are used to calculate total damage values for given air pollutant concentrations. To calculate dollars-per-ton emission damage values, we need to determine the total emissions of various pollutants contributing to the given air pollutant concentrations. Total emissions are estimated during Step 2, described above. While the damage value for a primary pollutant's concentration can be directly allocated to emissions of the contributing pollutant, the damage value for a secondary pollutant needs to be divided among the contributing primary pollutants. For example, the damage value for ozone concentration must be divided between VOC and NO_x, and the damage value for PM concentration must be divided among PM, SO_x, and NO_x. Usually, Step 4 (air quality simulation) provides information regarding the contribution of each primary pollutant to a secondary air pollutant concentration. Such information can be used to divide total damage values among different pollutants.

Estimation of absolute levels of air pollutant emissions and air pollutant concentrations is subject to many uncertainties that affect the accuracy of damage value estimates. To reduce these uncertainties, many past damage value studies have estimated damage values for changes in air pollutant concentrations — in other words, relative, rather than absolute, levels of air pollutant emissions and concentrations. Air pollution damage values are then estimated based on relative changes in air pollutant concentrations. For example, Hall et al. (1989) and Harrison et al. (1992) estimated total health damage values for ozone and PM from the ambient standards to the current actual concentrations. Hall et al. did not estimate dollars-per-ton emission damage values in their study. Harrison et al. estimated dollars-per-ton damage values from the estimated total damage values and the amount of emissions that needed to be reduced to meet air quality standards. RER (1992a) estimated total dollar values of air pollution caused by adding a 50-megawatt (MW) power plant to an air basin using the current air quality level. Dollars-per-ton damage values were calculated from the total damage values and total emissions from the hypothetical power plant.

In summary, the damage value estimating method seems theoretically sound. However, in practice, the method suffers from necessary assumptions and simplifications and from tremendous uncertainties involved in each estimating step. The cumulative effect of the uncertainties is a decrease in the accuracy of the estimated damage values. Studies using the method cannot practically include all potential adverse air pollution effects; some effects are excluded, and consequently, damage values are underestimated. Some scientists dispute the reliability of the methods that are applied to air quality modeling and economic valuation of air pollution effects. Outside the discipline of economics, philosophical uneasiness results when economists attempt to place dollar values on such intangibles as human life and human discomfort. Because of complex methods involved in each of the estimating steps, use of the damage value method is time-consuming and resource-intensive. These drawbacks have caused many organizations to use the control cost rather than the damage value estimating method.

2.2.2 Control Cost Method

The control cost estimating method is based on the presumption that emission standards or air quality standards are established at the ideal level — where the marginal damage of air pollution is equal to the marginal control cost (see Figure 2). In this approach, it is assumed that the cost required to meet predetermined air quality standards imposed by legislators "reveals" the value society places on the emissions being controlled (the method is sometimes called the "revealed preference method"). Therefore, the estimated marginal control cost to meet air quality standards represents the marginal damage value of air pollution when air quality standards are met. Two major steps are involved in the control cost estimating method: (1) identification of the marginal control measures required to meet predetermined air quality standards and (2) estimation of the dollars-per-ton control cost for each identified control measure. Each of these steps is described in detail below.

(1) Identification of Marginal Control Measures. Individual states are required to prepare state implementation plans and air quality control districts are required to prepare air quality management plans that indicate how and when ambient air quality standards are to be met. Usually, these plans identify certain control measures to be implemented.

States and local air quality districts provide a list of control measures to be implemented on the basis of air quality status, needed emission reductions, EPA-required control measures, and emission control cost effectiveness. The least expensive measures are usually recommended for implementation first, and the most expensive measures last. The last control measure (measures) to be implemented is (are) the marginal control measure (measures). Because current air quality status varies among air basins, different control measures may be required to meet uniform national ambient air quality standards (NAAQS). Therefore, to the extent that the list of control measures to be implemented is available for various air basins, the control cost method can be used to estimate marginal control costs for different air basins.

For some air basins, a list of control measures may not be available. Generic control measures for a larger region, rather than specific control measures for each basin, may have to be considered in determining marginal control measures for those air basins. Such generic control measures include best available control technology (BACT) and best available retrofitted control technology (BARCT), both of which are specified by EPA. While these technologies are believed to be the most stringent control technologies, they do not necessarily represent the marginal control measures for each air basin. It is often a difficult, subjective task to identify the marginal control measures for those air basins; the measures selected will have a significant impact on the estimated marginal control costs.

For some pollutants, marginal control measures may be selected from measures to mitigate emissions rather than directly controlling them. For example, control of carbon dioxide (CO_2) emissions from fossil fuel combustion sources could be extremely expensive. Rather than controlling the CO_2 emissions from these sources, the emissions may be reduced through other means such as CO_2 absorption by trees. Therefore, control costs for CO_2 emissions can be estimated from the cost of planting trees.

In some cases, market mechanisms have been adopted for air pollution control. For instance, in 1986, EPA adopted an emission trading program for controlling emissions from stationary sources (EPA 1986). The program included both offsetting and trading of emissions. Offsetting allows a major new source or a major modification of an existing source in a nonattainment area as long as emission increases from the new source or modification are offset by emission reductions from existing sources. Emission trading allows companies to buy emission reduction credits from other companies to meet emission requirements. The 1990 CAAAs adopted provisions to allow utility companies to trade SO_X emission allowances to meet SO_X requirements (EPA 1990a). In the South Coast Air Basin, the South Coast Air Quality Management District (SCAQMD) has recently adopted a program called RECLAIM (SCAQMD 1993) that allows emission trading for stationary SO_X and NO_X emissions. Use of these market mechanisms for emission control is expected to help reduce total emission control costs.

Market mechanisms also affect marginal control cost estimates — marginal control measures for emission control under market mechanisms should be less expensive than under a strict command and control mechanism. For example, through emission trading, a source can produce emissions exceeding air quality standards as long as the amount exceeded can be offset by some other source or sources. Therefore, expensive marginal control measures can be avoided for the more expensive emission source. Consequently, the control measures actually employed for the source are less expensive than those that would be required under inflexible rules. The actual control measures employed, not the theoretical marginal control measures, should be used to calculate actual marginal control costs.

Under the SO_X emission trading program allowed by the 1990 CAAAs, a cost for SO_X emissions will be determined in a newly established emission trading market. This market-determined SO_X price represents the opportunity cost for controlling one ton of SO_X emissions, and may be used as the marginal control cost for SO_X emissions.

(2) Calculation of Dollars-per-Ton Costs for Control Measures. Calculation of control costs in dollars per ton of emissions controlled requires information on the cost and emission reduction of the control measure over its lifetime. Cost estimation must include initial capital cost, operation and maintenance costs, and other pollutant-specific cost components. Estimates of emission reductions need to account for emission control deterioration over the lifetime of the equipment. If a control measure reduces emissions of more than on pollutant, the cost of the technology needs to be allocated among the reduced pollutants to obtain a dollars-per-ton cost for each pollutant. Obtaining the detailed information necessary for control cost estimates can be resource intensive. Assumptions often have to be made for certain components.

Discount rates must be applied in calculating lifetime costs and lifetime emission reductions. The magnitude of the discount rates differs among the past studies. Although it is generally agreed that discounting needs to be applied to the cost estimates, researchers dispute whether discounting should be applied to emission estimates, and treatment varies among studies.

Depending on whether discounting rates are applied to emissions and whether the lifetime of the control technologies is considered, different techniques can be used in calculating control costs. Table 1 illustrates four techniques, each based on units per year or per lifetime and each incorporating such variables as levelized costs, lifetime present value of costs, levelized tons, discounted tons, or straight total tons of emissions.

Application of each of the above techniques to the same control technology results in different cost estimates with different meanings. Appendix A illustrates these differences by means of examples for emission control costs of an electric vehicle and a stationary source. As illustrated, application of the different techniques may prohibit comparison between studies. Techniques 1 and 4, which measure the costs and emissions reductions over different periods, incorporate an inconsistent time variable, so that control costs calculated using either technique for control measures with various lifetimes cannot be directly compared. Techniques 1 and 4 should not be used for calculating per-unit emission control costs.

Table 2 presents the calculated emission control cost-effectiveness of electric vehicles. Note that for technique 1 or 2, the difference between cases a and b is only significant if the annual emission reductions vary over the lifetime. Furthermore, the difference between the two cases is larger if a greater discount rate is applied. Also note that the results using technique 2a are the same as those using technique 3a. Either of these two methods may be applied to calculate what are regarded as correct control cost estimates in this study. We believe that these two methods properly apply economic rules to develop control cost values that are meaningful in a cost-benefit analysis.

Many studies to estimate emission control costs have been completed using technique 2b or 3b. That is, control costs have been calculated in dollars per ton of emissions reduced, and discounting has been applied to costs, but not to emissions. To calculate control costs for use in social evaluation of the costs and benefits of control measures, we recommend that discounting be applied to both costs and emission reductions because, while cost estimates represent the cost side, emission reductions represent the benefit side. Therefore, we recommend that technique 2a or 3a be used in calculating emission control costs.

In summary, the control cost estimating method does not involve as many as steps and assumptions as the damage value method, can generate costs more quickly using fewer resources, and does not require highly specialized expertise to construct cost estimates.

TABLE 1 Emission Control Cost Calculating Techniques

Case	Calculation Method	Unit	Meaning
Technique 1: Lifetime Costs Divided by Annual Emission Reductions			
Case a: discounting both costs and emissions	(lifetime present value of cost)/ (levelized tons reduced per year)	($/lifetime)/ (ton/year)	Cost to reduce one ton each year throughout lifetime
Case b: discounting costs only	(lifetime present value of cost)/ (straight average of tons reduced per year)		
Technique 2: Annual Costs Divided by Annual Emission Reductions			
Case a: discounting both costs and emissions	(levelized costs per year)/ (levelized tons reduced per year)	$/ton	Cost to reduce one ton
Case b: discounting costs only	(levelized costs per year)/ (straight average of tons reduced per year)		
Technique 3: Lifetime Costs Divided by Lifetime Emission Reductions			
Case a: discounting both costs and emissions	(lifetime present value of costs)/ (lifetime present value of tons reduced)	$/ton	Cost to reduce one ton
Case b: discounting costs only	(lifetime present value of costs)/ (straight sum of lifetime tons reduced)		
Technique 4: Annual Costs Divided by Lifetime Emission Reductions			
Case a: discounting both costs and emissions	(levelized costs per year)/ (lifetime present value of tons reduced)	($/year)/ (ton/lifetime)	Annual cost throughout lifetime to reduce one ton
Case b: discounting costs only	(levelized costs per year)/ (straight sum of lifetime tons reduced)		

TABLE 2 Emission Control Cost-Effectiveness of Electric Vehicles

Technique	Case	Discount Rate[a] 0.04	0.06	0.08	Unit	Meaning
1	a	81,436	80,026	79,082	($/lifetime)/	Cost to reduce one ton
	b	76,086	72,250	68,965	(ton/yr)	each year throughout lifetime
2	a	8,677	9,545	10,494	$/ton	Cost to reduce
	b	8,107	8,618	9,154		one ton
3	a	8,677	9,545	10,494	$/ton	Cost to reduce
	b	6,341	6,021	5,749		one ton
4	a	925	1,139	1,392	($/year)/	Annual cost through-
	b	676	718	763	(ton/lifetime)	out lifetime to reduce one ton

[a] For detailed assumptions and calculating procedures, see Appendix A.

Chernick and Caverhill (1991) argue that, considering the uncertainties involved in the damage value method, the control cost method is superior for estimating emission values. However, the method suffers from the fundamental assumption that legislators and/or regulators establish emission and air quality standards solely on the basis of marginal damages and the marginal costs of meeting the standards. In reality, establishing emission and air quality standards is a highly political process in which economic implications are only one of many factors considered. Proponents of the control cost method argue that the method assumes a composite control cost that represents economic, political, and social implications. However, such a composite cost implies that political and social effects can be interpreted in the economic sphere — a philosophy that troubles some people.

The method implies that legislators or regulators have perfect information on marginal damages and the marginal control costs required to meet the standards, which is highly unlikely. Emission and air quality standards are not usually set at a level where marginal damage is equal to marginal control cost; the legal basis for the standards is generally a level at which health effects are minimized. On the basis of strict economic theory, it is improper to treat the estimated marginal control cost as the value for emission damages. Nevertheless, the calculated control cost represents the opportunity cost of meeting the standards. The engineering discipline sets standards based on liberal safety margins, "good practice," and human comfort. If some other control measures are implemented, the most costly measures can be avoided. It is this "avoided opportunity cost" that this report examines. However, we do not take the position that emission damage values are accurately represented by the estimated emission control costs. In many cases, damage values will differ significantly from control costs, and control costs cannot represent damage values.

3 REVIEW OF PREVIOUS STUDIES

Various studies have been conducted using the damage value or the control cost method to estimate air pollutant emission values. These studies have often been cited and used by various public and private organizations without careful consideration of the methodologies and assumptions used. This section presents a review of the methodologies, assumptions, results, and applications of past major studies on emission value estimation. The reviewed studies are divided into subsections of damage-based and control-cost-based studies. Within each subsection, studies are presented in chronological order.

3.1 DAMAGE-BASED STUDIES

3.1.1 Bonneville Power Administration — 1986, 1987, 1991

In 1986, ECO Northwest conducted a study for the Bonneville Power Administration (BPA) to estimate values of environmental pollution for five renewable power plant types (i.e., co-generation by biomass, co-generation by municipal solid waste, geothermal, solar central stations, and wind) (ECO Northwest 1986). The ECO Northwest study was the first to estimate damage values of environmental pollution from power plants.

The 1986 study included effects on human health, visibility, materials, crops and forests, water quality, land use, solid waste disposal, endangered species, aesthetic quality, and cultural value. ECO Northwest implicitly assumed that the five power plant types evaluated would be located in the Pacific Northwest, and estimated environmental pollution values in mills per kilowatt-hour (kWh) of electricity produced. The study did not present dollars-per-ton emission values for air pollutants.

In 1987, ECO Northwest completed a separate study for BPA to estimate damage values of air pollution for a generic, 1,300-MW-capacity coal power plant (ECO Northwest, 1987). The study assumed six sites for the generic coal plant: two were west of the summit of the Cascade mountain range inside BPA's service area (one site near a large city and the other near a medium-size city); three were east of the summit of the Cascade Range inside BPA's service area (these sites were assumed to be near a large-, medium-, or small-size city); and one was in eastern Montana near a small city outside BPA's service area. The study included air pollution damages to human health (mortality and morbidity), agricultural crops, materials, visibility, ecosystems (forest and lakes), livestock, and timber. Damage values for three air pollutants (NO_x, SO_x, and PM) were estimated for the generic coal plant in each of the six sites. The estimated values were presented in mills per kilowatt-hour of electricity produced. Dollars-per-ton emission damage values were not presented.

On the basis of ECO Northwest-estimated damage values, BPA (1991) adopted emission values for NO_x, SO_x, and PM, and used the adopted values to calculate environmental externality costs in its least-cost resource planning process. These costs were added to other cost items to calculate total costs per unit of electricity generated. The calculated total costs were used to rank the costs associated with the power plant types proposed for construction. However, the externality costs were not considered in estimating the final contract price for electricity.

BPA adopted different damage values for areas west and east of the Cascade Range. The two general areas correspond to two diverse areas in Oregon: the west coastal areas (which are highly populated) and the east high plateaus and deserts (which are less populated). BPA did not explicitly distinguish between non-attainment and attainment areas, although it could be argued that values in non-attainment areas could be approximated to the west values and values in attainment areas to the east values.

Although BPA generally preferred to use damage-based emission values, in adopting the value for SO_x, it relied on the estimated value of the allowance to be traded in the SO_x emission trading market allowed by the 1990 CAAAs. The SO_x allowance value was based on SO_x control costs from various power plants. The BPA-adopted dollars-per-ton emission values are listed in Table 3.

3.1.2 South Coast Air Quality Management District — 1989

Hall et al. conducted a comprehensive study for SCAQMD to estimate damage values of ozone and PM_{10} air pollution in the South Coast Air Basin (Hall et al. 1989 and 1992). In their estimates, the researchers included air pollution damages to human health only; damages to agriculture, materials, ecosystems, and visual aesthetics were ignored. Even health effects such as increased asthma attacks, days of hospitalization, and increased cancer risks were left out. Therefore, the estimates made by Hall et al. probably understate actual air pollution damages in the South Coast Air Basin.

When conducting their study, Hall and his colleagues considered the differences between the 1987 air quality levels and applicable air quality standards (both federal and state standards); in this way, they avoided air quality modeling in their study. The researchers established a comprehensive model to estimate the distribution of pollutant exposure and the dose received by human receptors. The study relied on previous studies that included human clinical experiments and epidemiological statistics for the dose-response functions. Economic valuation of air pollution effects was based on either the market-measured cost of effects (such as cost of illness) or costs estimated using the hedonic pricing approach for non-market effects such as discomfort.

Hall et al. estimated substantial health damage values for violation of ozone and PM_{10} standards: an annual damage value of \$1-5.5 billion for violation of the federal ozone standard in the South Coast Air Basin, \$2.7-13.9 billion for violation of the federal PM_{10} standard, \$1.2-6 billion for violation of the California ozone standard, and \$4.7-23.5 billion for violation of the California PM_{10} standard. The study did not estimate dollars-per-ton emission values.

TABLE 3 Damage-Based Emission Values Estimated in Previous Studies

		Emission Value ($/ton, 1989 dollars)[a]				
Study	Region	NO_x	ROG	CO	PM_{10}	SO_X
BPA 1991	West of Cascade Range	849	N/A[b]	N/A	1,480[c]	1,441[d]
	East of Cascade Range	66	N/A	N/A	160[c]	1,441[d]
Pace 1991	Northeastern United States	1,640	N/A	N/A	2,360[c]	4,060
SCE 1992	South Coast: 1989	4,713	2,671	N/A	13,217	504
	South Coast: 2011	8,589	4,821	N/A	24,393	504
CEC 1993	South Coast	14,483	6,911	3	47,620	7,425
	Venture County	1,647	286	0	4,108	1,500
	San Francisco Bay Area	7,345	90	1	24,398	3,482
	San Diego	5,559	98	1	14,228	2,676
	San Joaquin Valley	6,473	3,711	0	3,762	1,500
	Sacramento Valley	6,089	4,129	0	2,178	1,500
	North Coast	791	467	0	651	1,500
	North Central Coast	1,959	803	0	2,876	1,500
	South Central Coast	1,647	286	0	4,108	1,500
	Southeast Desert	439	157	0	680	1,500
Nevada Power Co. 1993	Las Vegas Valley: 1990	211	0	N/A	1,364	288
	Las Vegas Valley: 2010	423	0	N/A	2,729	0
	Outside Las Vegas Valley: 1990	173	0	N/A	192	77
	Outside Las Vegas Valley: 2010	327	0	N/A	404	0

[a] Emission values in some studies were expressed in dollars other than 1989 dollars. The consumer price index was used to convert these values into 1989 dollars.

[b] N/A = not available.

[c] Value for total PM.

[d] Based on the estimated market value of SO_X allowance in the SO_X trading market, which is allowed by the 1990 CAAAs. Thus, the value is control-cost-based.

3.1.3 Pace University — 1991

In 1990, the Center for Environmental Legal Studies at Pace University in New York conducted a study for the New York State Energy Research and Development Authority and the U.S. Department of Energy to review and analyze existing studies on air pollutant externality values (Ottinger et al. 1991). The study, which included a thorough review of existing literature, discussed in detail the important issues involved in estimating externality costs associated with generation of electricity. Such issues include use of discount rates and the statistical value of human life. The study summarized the actions taken by individual state PUCs to incorporate environmental externality costs in utility resource planning and resource bidding evaluation.

The study presented various methods of estimating emission values. Of the presented methods, the Pace University research team preferred the damage value method. They insisted that damage estimates must include health effects, visibility effects, material damages, and vegetation damages (forest-related and agricultural). The Pace University researchers did not conduct their own estimates to obtain values for these effects; instead, they assessed air pollution values estimated in a variety of damage-based studies dating from 1980 to 1990, and selected a "starting value" for each identified effect. Although a wide range of studies pertaining to different years and locations were cited, the "starting point" values in the Pace University study were derived largely from ECO Northwest's 1987 study for BPA. Damage values for CO_2, SO_2, NO_x, PM, land use, and water pollution were proposed in the Pace study.

Because of the nature of their study, the Pace researchers clearly stated that their values represented "starting point values" — a set of basic damage values for use in further research but not for direct use as damage estimates. Nevertheless, to demonstrate the usefulness of emission values, the Pace University researchers applied dollars-per-ton damage values and the amount of emissions from six power plant types (coal-fired, oil-fired, natural gas-fired, nuclear, renewable, waste-to-energy) and calculated environmental externality costs in mills per kilowatt-hour of electricity produced for each of the power plant types.

3.1.4 California Energy Commission — 1989, 1990, 1991, 1992, 1993

The California Energy Commission (CEC) has estimated values of power plant air pollutant emissions since 1989. CEC intended to use the emission values to estimate air pollution externality costs, which it would employ in evaluating and selecting power plant sites. We present CEC's historical development of the damage value and the control cost method together here.

In 1989, CEC used the control cost method to estimate emission values for NO_x, ROG, SO_x, PM_{10}, and CO_2 in the South Coast Air Basin, the remainder of the state, and out-of-state areas (CEC 1989). In estimating emission values for the South Coast Air Basin, CEC relied on the Tier 1 emission control measures presented in SCAQMD's 1989 Air Quality Management Plan. CEC used the high-end cost ranges to approximate marginal control costs, and assumed the same emission values for the South Coast Air Basin as for the rest of the state. For out-of-state regions that were generally in attainment, CEC simply assumed 10% of the values in the South Coast Air Basin (except CO, for which CEC assumed the same value for California as for out-of-state areas). In its 1990 electricity report, CEC revised its 1989 emission value estimates to reflect comments from utility companies; the revisions included substantial reductions in the 1989 NO_x estimates. While some emission values were applied to both the South Coast Air Basin and the rest of the state, no values were assigned to emissions out of the state (except CO_2, for which a value of $7 per ton was assigned to both California and out-of-California areas) (CEC 1990).

In preparing its 1992 electricity report, CEC continued to work on emission value estimation. Concerned about the fundamental problems of the control cost method, CEC began to consider both the control cost and the damage value estimating methods.

CEC's concerns regarding the cost control method included its belief that "taking the highest marginal control costs from any sector may not represent the public's true willingness to pay for additional emission reductions in the electricity sector." (p.6, Buell et al. 1991). In addition, CEC stated that the marginal cost for a source classification may often be overestimated. Based on these reasons, CEC decided to eliminate the measures costing over $100,000 per ton when selecting marginal control measures. CEC used the average costs of *marginal* control measures as the emission values for NO_x or ROG, but the average costs of *all* control measures as emission values for SO_x, PM_{10}, or CO, because data regarding the marginal control measures for these three pollutants were not available.

CEC estimated emission values for each of the California air basins and for various other states on the basis of information regarding emission control measures and their costs; this information was obtained from the air quality control district in each basin.

In using control costs generated by individual air quality districts, CEC made some adjustments and corrections. For example, in adopting SCAQMD's cost estimates, CEC corrected SCAQMD's per-ton control costs by applying discounting to emissions (which SCAQMD ignored). We accept CEC's position that physical emissions should be discounted. CEC also adjusted costs expressed in various current dollar terms to 1989 constant dollars by using a series of inflation rates.

In evaluating the damage value estimating method, CEC used the air quality valuation model (AQVM) developed for the commission by RER (Tanton et al. 1992). The AQVM includes emission estimation, air quality simulation, estimation of physical effects of air pollution, and valuation of air pollution effects.

CEC contracted RER to conduct a comprehensive study to estimate emission damage values using the damage value method. RER researchers reviewed previous studies on dose-response functions and valuation of air pollution effects and, based on these studies, adopted a set of dose-response functions and a set of values for air pollution effects (RER 1990). RER has frequently updated and revised its adopted dose-response functions and effects values since 1990, based on newly available information.

In estimating emission values, RER included adverse air pollution effects on human health (mortality and acute morbidity, but not chronic diseases), agricultural crops, non-agricultural vegetation, materials, and visual aesthetics.

RER used simplified air quality models including the screen model for non-reactive pollutants such as CO and PM and the environmental kinetic modeling approach (EKMA) model for ozone to simulate air quality effects caused by emissions. The screen model is recommended by EPA for preliminary evaluation of potential air pollution problems. The model employs a set of standard meteorological conditions (designed by EPA to represent the most likely worst-case concentration) to produce downwind concentrations versus distance. The EKMA model, which requires less input data than some advanced models (such as the urban airshed model), is recommended for determining the level of VOC control needed to meet the ozone standard. EPA prefers the urban airshed model, which requires intensive, detailed input data, for ozone simulation.

RER has produced a computer-based AQVM for the CEC to estimate emission damage values. Using input data on emissions, background air pollutant concentrations, meteorological conditions, and other factors, the AQVM can produce dollars-per-ton emission values for different air basins in California. Emissions from a hypothetical power plant in a specific region are fed into the model, and dollars-per-ton emission values are calculated.

RER pointed out that calculation of dollars-per-ton emission values by the model is problematic because allocating total air pollution damage values to given primary air pollutants is difficult. RER recommends that the dollars-per-ton calculation be eliminated (RER 1992b), but CEC has been using the model to estimate dollars-per-ton emission damage values for various California air basins.

In its estimates, CEC uses the emission reductions projected by the air quality control district in each air basin, and assumes linear relationships between emissions and air quality. Although this assumption is questionable, it is used in virtually all of the damage-based studies reviewed in this section, because of the high costs and inaccuracies involved in estimating the "true" nonlinear relationships.

In its 1992 electricity report, CEC established emission values for various pollutants in each of the California air basins and in other states that provide electricity for California (CEC 1993a and 1993b). CEC-estimated emission values are listed in Table 3.

After the 1992 electricity report, CEC began to use damage-based rather than control cost-based values. CEC continues to refine the AQVM to produce accurate emission value estimates. The commission is currently working on generating new emission value estimates for its 1994 electricity report.

3.1.5 Southern California Edison Company — 1992

In 1992, National Economic Research Associates (NERA) conducted a study for the Southern California Edison Company (SCE) to estimate damage values of air pollutants in the South Coast Air Basin (Harrison et al. 1992). Starting with the current air quality in the basin, the study estimated the damage values of ozone and PM_{10} between the current air quality levels and federal standards. Assuming that PM_{10} pollution was caused solely by PM_{10} emissions, NERA allocated the total cost of PM_{10} air pollution to PM_{10} emissions, and divided the total cost of ozone air pollution between ROG and NO_x according to their contributions to ozone formation in the South Coast Air Basin. Dollar-per-ton damage values for each pollutant were then calculated by dividing the total cost of emission reductions by the total reduction necessary to meet federal air quality standards. Therefore, the estimated dollar-per-ton value was the cost of moving from the current air quality concentrations to attainment of air quality standards. Damage values estimated included human health effects (mortality and morbidity), visibility effects, materials damages, forest-related aesthetic damages, and agricultural damages.

In its study, NERA relied on the results of various studies to complete certain estimating steps. For example, NERA adopted the dose-response functions and the air pollution impact values outlined in previous studies.

In preparing its testimony before the California PUC regarding the values of emission reductions by electric vehicles, SCE used the NERA-estimated emission values for NO_x, ROG, and PM_{10} (SCE 1992); the SO_x emissions value estimated by RER for CEC ($560 per ton of SO_x); and the out-of-state emission values estimated by CEC for out-of-basin emissions.

3.1.6 Nevada Power Company — 1993

NERA recently completed a study for the Nevada Power Company to estimate damage values of PM_{10}, NO_x, SO_x, and VOC in Southern Nevada. The changes in concentrations of PM_{10}, NO_2, SO_2, and ozone were simulated based on emissions of PM_{10}, NO_x, SO_x, and VOC generated from a hypothetical power plant in and out of the Las Vegas valley (where concentrations exceed federal PM_{10} and CO standards). Assuming that emissions from the hypothetical power plant affected an area of measuring 100 km by 100 km, NERA estimated values for various air pollution effects within the area. Researchers divided the affected area into 2.5-km by 2.5-km grids, and established exposure factors (increase in population exposure per ton of emissions) for each grid. Using the established exposure factors and the air pollution values cited from other studies, NERA estimated the per-ton damage value of each pollutant emitted from the hypothetical power plant.

The 1993 study included air pollution effects of human mortality and morbidity, visibility, material and agricultural damages, and acid deposition damages to ecosystems (e.g., lakes, forests, and agriculture). NERA estimated damage values on the basis of 1990 baseline emissions, concentrations, and population data, then extrapolated the estimated values to the years 2000 and 2010 by considering changes in population, per-household income, and application of SO_x emission trading allowed by the 1990 CAAAs. These estimated values have been used by the Nevada Power Company in planning its power plant resources.

3.1.7 Summary: Damage-Based Emission Values Estimated in the Reviewed Studies

Table 3 presents damage-based emission values estimated in the reviewed studies. As the table shows, there are wide variations in emission values among various regions. The variations are caused by differences in air pollution concentrations, population exposed, and methods and assumptions used in the studies. The latter are most significant because these differences contaminate the estimated emission values. For example, for the South Coast Air Basin, CEC-estimated values are much higher than SCE-estimated values, because of the different assumptions and methods used in the two studies.

In most California air basins, SO_x emissions have the same value ($1,500). Although CEC did not state how the value was estimated, CEC likely used the SO_x allowance value in the SO_x trading market that was estimated in some other studies. So the CEC-estimated SO_x value may be based on the control cost estimating method.

When estimating damage-based emission values, the above studies generally evaluated the current air quality status, then added power plant emissions to the studied areas. Because most of the studied regions are nonattainment areas, the estimated values are generally for nonattainment areas.

3.2 CONTROL-COST-BASED STUDIES

3.2.1 South Coast Air Quality Management District — 1988 and 1991

SCAQMD generated an initial list of control measures for its 1989 Air Quality Management Plan in 1988 (SCAQMD 1988a and 1988b), and modified the list for its 1991 revision of the plan (SCAQMD 1991). In its list, SCAQMD presents emission control cost-effectiveness in dollars per ton of emissions reduced for a number of control measures.

The SCAQMD control measure list includes measures that meet the District's Tier I criteria for currently available technologies to achieve compliance with the California Clean Air Act. SCAQMD includes in its list the most stringent control technologies to be employed in new sources. Such technologies include the BACT, which represents the most stringent control technologies required for emissions reductions applied to new sources in nonattainment areas.

SCAQMD used two methods for calculating the cost-effectiveness of various emission control measures: (1) the levelized cash flow method, and (2) the discounted cash flow method (SCAQMD 1988c). These methods correspond to techniques 2b and 3b in Table 1, in which discounting was applied to cost estimates but not to emission estimates. SCAQMD included capital, operating, and maintenance costs; assumed an economic lifetime of 10 years for most control technologies; and used a real discount rate of 4% in its estimates.

SCAQMD's control measure list has served as the basis of several control-cost-based studies. For example, the Tellus Institute (Section 3.2.4) and the CEC (Section 3.1.4) studies used SCAQMD's list to develop emission values for southern California.

SCAQMD has recently adopted an emissions trading program to control SO_X and NO_x emissions from all large sources in the South Coast Air Basin (SCAQMD 1993). Dollars-per-ton emission values for SO_x and NO_x can now be determined based on the market values of SO_x and NO_x in the emission trading market.

3.2.2 New York State Energy Office — 1989

In 1989, the New York State Energy Office (NYSEO) estimated emission values for NO_x, SO_2, and CO_2 and provided the estimated emission values to the state's utility companies for use in calculating air pollution externality costs, which were used to select least-cost power plant types.

In developing the estimates, NYSEO applied the average costs of low-cost technologies (mixed control technologies) and high-cost technologies (advanced control technologies) to power plants. NYSEO maintained that low-cost technologies reflected control costs in attainment areas, while high-cost technologies reflected control costs in nonattainment areas.

In determining marginal control technologies, NYSEO considered the control technologies necessary to meet power plant emission standards in both attainment and nonattainment areas. Using input parameters for a 200-MW coal-fired power plant and identified marginal control technologies for the plant, NYSEO estimated dollars-per-ton emission values. NYSEO's cost estimates included the private costs of installing and operating control equipment (i.e., capital, operating, and maintenance costs) and the governmental costs of implementing and monitoring power plant emission regulations. In calculating control cost-effectiveness, NYSEO assumed a lifetime of 20 years for all control technologies installed in power plants, and used a nominal discount rate of 10%. Discounting was not applied to emissions. NYSEO-estimated emission values are presented in Table 4.

3.2.3 Independent Energy Producers of California — 1989

In 1989, JBS Energy, Inc. conducted a study for the Independent Energy Producers (IEP) of California to estimate emission values of ROG, NO_x, SO_x, CO_2, methane (CH_4), and N_2O in southern California, the remainder of California, and areas outside of California

TABLE 4 Control-Cost-Based Emission Values Estimated in Previous Studies

Study	Region	Emission Values (\$/ton, 1989 dollars)[a]				
		NO_x	ROG	CO	PM_{10}	SO_x
NYSEO 1989	State of New York	2,460	N/A[b]	N/A	N/A	603
IEP 1989	Southern California	24,500	17,500	N/A	N/A	18,300
	Remainder of California	18,800	1,130	N/A	N/A	1,800
	Outside California	2,700	565	N/A	N/A	1,000
Tellus Institute 1990	Southern California	262,000	29,000	820	44,000	75,000
	Northeastern United States	6,500	5,300[c]	820	4,000[d]	1,500
PSCN 1991	State of Nevada	6,297	1,093[c]	852	3,871[d]	1,445
NYSEO 1991	State of New York: average costs	933	N/A	N/A	N/A	181
	State of New York: marginal costs	4,039	N/A	N/A	N/A	824
CEC 1993	South Coast	26,400	18,900	9,300	5,700	19,800
	Venture County	16,500	21,100	0	1,800	6,200
	San Francisco Bay Area	10,400	10,200	2,200	2,600	8,900
	San Diego	18,300	17,500	1,100	1,000	3,600
	San Joaquin Valley	9,100	9,100	3,200	5,200	17,800
	Sacramento Valley	9,100	9,100	5,000	2,800	9,600
	North Coast	6,000	3,500	0	900	3,000
	North Central Coast	9,100	9,100	0	900	3,000
	South Central Coast	9,100	9,100	0	900	3,000
	Southeast Desert	6,000	3,500	2,900	5,700	19,700
	O_3 Attainment but PM_{10} Violation Areas	6,000	3,500	0	900	3,000
OPUC 1993	State of Oregon[e]	3,363	N/A	N/A	2,882[d]	N/A

[a] Emission values in some studies were expressed in dollars other than 1989 dollars. Consumer price index was used to convert these values into 1989 dollars.

[b] N/A = not available.

[c] Value is for VOCs.

[d] Value is for total PM.

[e] OPUC adopted a range of costs for each pollutant. The middle value of the range is presented here.

(Schilberg et al. 1989). JBS adopted SCAQMD's control measure list and the SCAQMD-estimated control costs to calculate emission values, and selected the measures necessary to meet BACT standards in Southern California as marginal control measures. In the rest of California, the researchers chose control technologies to meet less stringent emission standards. For the out-of-state areas, NYSEO-estimated values for NO_x were used. The study also assumed use of scrubbers in new coal-fired power plants to estimate the value for SO_x, and employed a value of half of the ROG for the rest of California as the ROG value for the out-of-state areas. The value for CO_2 was estimated from the cost of reforestation in northern California and in the Pacific Northwest. Values for CH_4 and N_2O were estimated from the value for CO_2 and the global warming potentials of the three pollutants.

3.2.4 Tellus Institute — 1990

In 1990, the Tellus Institute of Boston conducted a study to estimate emission values of air pollutants using the control cost method (Bernow and Marron 1990). Although Tellus researchers maintained their preference for the damage-based method, they did not believe that the method was reliable for use in actual policy applications because less reliable relationships and inadequate input data were generally used with the method. Consequently, the Tellus researchers suggested that the control cost estimates be used as surrogates for emission damage values, and they used the control cost method to estimate emission values.

The Tellus researchers developed estimated emission values for NO_x, SO_x, VOC, PM, CO, CO_2, CH_4, and N_2O in Southern California and in the northeastern United States. In determining marginal control measures, Tellus researchers used the measures with the highest control costs necessary to comply with emission and air quality standards imposed by the BACT, the NAAQS, or the 1990 CAAAs. For Southern California, the most expensive control measures proposed by SCAQMD were selected. For the northeastern United States, marginal control measures were determined from a variety of sources.

In its estimates, the Tellus study included capital, operating, and maintenance costs. Lifetime costs of control measures were calculated assuming a lifetime of 30 years (the average lifetime of a power plant) for control technologies and a real discount rate of 7%. Because the Tellus estimates reflected values in Southern California and the northeastern United States, both of which are nonattainment areas, the Tellus emission values are for nonattainment areas. The Tellus estimates have been widely used by state PUCs in utility resource planning and acquisition.

3.2.5 Public Service Commission of Nevada — 1991

In 1989, the Public Service Commission of Nevada (PSCN) proposed to incorporate environmental externality costs of power plant operations into the utility resource planning process. In 1991, PSCN adopted emission factors for various power plant types and dollars-per-ton emission values for nine air pollutants (PSCN 1991). PSCN requires that all utility companies in Nevada use its adopted values as the basis for calculating the air pollution externality costs of power plants.

Emission values adopted by PSCN were based primarily on the Tellus Institute's estimates, with some adjustments. PSCN lowered Tellus' estimated values for NO_x, VOCs, and CO to reflect ozone and CO attainment status in Nevada.

To address uncertainties involved in its adopted emission values, PSCN allowed utility companies to use either PSCN-adopted values or to generate their own estimated values. Two utility companies (Nevada Power Company and Sierra Pacific Power) opted to estimate emission values using the damage value method. Their estimated emission values are lower than the PSCN-adopted values — consistent with many other studies that have generally shown that damage-based values are lower than control-cost-based values. The Nevada Power Company has already published its damage-estimated values; the Sierra Pacific Power Company is in the process of completing its estimates. Meanwhile, PSCN is currently revising its rule to possibly mandate use of the PSCN-adopted emission values.

3.2.6 New York State Energy Office — 1991

In 1991, NYSEO conducted a study to estimate emission tax rates for achieving a given level of emission reductions. For the study, NYSEO designed two tax schemes: a general revenue tax and a trust fund tax. Under the general revenue scheme, the tax rate was equal to the cost of the marginal control measures required to achieve the desired emission reduction. Thus, the general revenue tax reflects the marginal cost for emission control. Under the trust fund scheme, the tax rate was determined from the total cost of achieving a pre-determined emission reduction level divided by the total emissions reduced. The trust fund tax, then, reflects the average cost for emission control. In this way, the 1991 NYSEO study indirectly estimated emission values using the control cost method.

As part of the study, NYSEO estimated dollars-per-ton emission values for SO_2, NO_x, and CO_2. For SO_2, the marginal damage curve and the marginal control cost curve were estimated first. The marginal damage curve was a linear connection between two data points. One point was a damage value of $2,200 per ton for 280,000 tons of emissions per year (the cap in the state of New York according to 1990 CAA requirements). The other point was a damage value of zero at 100,000 tons of annual SO_2 emissions. The marginal control cost curve was based on the retrofit application of SO_2 control measures to achieve system-wide SO_2 reductions. By comparing the marginal damage curve and the marginal control cost curve, NYSEO determined that SO_2 emissions could be cost-effectively reduced an additional 75,000 tons per year beyond the 1990 CAAA requirements in the state. To achieve this reduction in SO_2 emissions, NYSEO calculated a revenue tax of $858 per ton and a trust fund tax of $188 per ton (1990 real dollars).

For NO_x, NYSEO calculated a general revenue tax of $4,204 per ton. The rate was determined to be equal to the highest marginal control cost of meeting NO_x emission reduction requirements in the state of New York according to the 1990 CAAAs. To calculate a trust fund tax rate, NYSEO assumed a reduction of 6,600 tons of NO_x at an average cost of $6,100/ton, resulting in a total control cost of $42.26 million. The NO_x emission reduction of 6,600 tons was equivalent to the increase in NO_x emissions between 2000 and 2010 in the state. By spreading the calculated total cost over total NO_x emissions reduced (the 6,600 tons and NO_x emission reductions already achieved from other control measures), NYSEO calculated a trust fund tax of $971 per ton of NO_x.

For CO_2, NYSEO assumed a 10% reduction in total CO_2 emissions over the 1988 New York State CO_2 emission inventory. On the basis of the total cost of achieving this reduction, NYSEO calculated a general revenue tax of $74 per ton and a trust fund tax of $5.5 per ton.

3.2.7 Massachusetts Department of Public Utilities — 1992

In 1990, the Massachusetts Department of Public Utilities (MDPU) adopted regulations requiring that electric utility companies in Massachusetts take into account environmental externalities in their power plant resource planning (D.P.U. 89-236) (MDPU 1992). MDPU established initial values for certain air pollutants, and allowed utility companies to update emission values on a case-by-case basis. In 1991, when the Massachusetts Electric Company submitted different externality values, MDPU decided to investigate whether the previous values should be updated or revised.

During MDPU's investigation, several utility companies proposed use of damage-based estimates. They argued that the damage value method is more conceptually correct for estimating externality costs because the control cost method is based on the misconception that environmental regulations are determined at the level where marginal benefits are equal to marginal costs, which is an oversimplified view. They further argued that the control cost method violated a fundamental principle of economics, because marginal control costs vary across different sources for the same pollutant, and these source-specific marginal costs cannot be established by society's collective revealed preference for a level of pollution.

On the other hand, MDPU and some other parties that were in favor of the control cost method argued that the methodological correctness and the theoretical appeal of the damage value method must be regarded as flaws that become evident upon examination of how the damage values are actually estimated. For example, some important damage effects are ignored in most damage estimate studies, and the tremendous uncertainty of the method is not addressed.

MDPU therefore established two criteria for accepting an estimating method: comprehensiveness and reliability. To meet the comprehensiveness criterion, estimates should be developed to address all important effects of emissions, including human morbidity, mortality, and genetic effects; material damages; agricultural productivity; and non-priced goods (e.g., cultural, scenic, and recreational value; visibility; and damages to species and natural systems). To meet the reliability criteria, estimates must be defensible. An estimate is defensible if it is accompanied by a clear and explicit presentation of the method, data, calculations, judgments, and assumptions used, and addresses the variability and uncertainty of the results.

MDPU determined that the present stage of the damage value method did not meet the two criteria. Therefore, the department adopted the control cost method in estimating externality costs. MDPU also adopted the values estimated by the Tellus Institute for the northeastern United States As previously discussed, these estimates are among the highest developed values because they are based on using the most expensive control measures.

MDPU's investigation revealed two additional issues. First, the offsetting policy allowing utilities to flexibly meet emission requirements should reduce actual control costs, compared to those assumed by the Tellus Institute. Second, the SO_x emission trading provision allowed by the 1990 CAAAs will internalize SO_x emission costs in utility resource planning, making it unnecessary to require utility companies to incorporate SO_x emission values in their resource planning process. These two issues remain unsettled.

3.2.8 Oregon Public Utility Commission — 1993

In 1989, the Oregon Public Utility Commission (OPUC) adopted a requirement that air pollution externality costs be included in utility resource planning. OPUC provided ranges of emission values for NO_x, total suspended particulate matter (TSP), and CO_2. OPUC did not provide an emission value for SO_x, because the commission believed that the SO_x emission trading provision allowed by the 1990 CAAAs would help internalize the cost of SO_x emissions in utility resource planning.

Emission value ranges proposed by OPUC were based on the costs of marginal control technologies required by the BACT for new power plants in attainment areas in Oregon and Washington. In calculating the costs of marginal control measures, OPUC assumed a lifetime of 30 years for the control technologies. The cost estimates included only equipment costs; operating and maintenance costs were excluded.

When the Oregon Department of Justice maintained that OPUC might not have the authority to require incorporation of externality costs in utility resource acquisition decisions, OPUC decided to provide general guidelines for utilities to consider externality costs in their resource planning process. In its guidelines, OPUC suggested that utility companies select emission values within the ranges provided or estimate their own emission values. Two utility companies (Portland General Electric and Pacific Corporation) suggested that air pollution externality costs used in the resource planning process should be the lowest estimate from among the marginal damage value, marginal control cost, and offset cost. Portland General Electric Company maintained that the upper bounds of the emission value ranges proposed by OPUC should be lowered to reflect the power plants' opportunity to reduce emission control costs by using offsets to comply with air quality standards.

3.2.9 Summary: Control-Cost-Based Emission Values Estimated in the Reviewed Studies

Table 4 presents control-cost-based emission values estimated in the above studies. There are wide variations in emission values among the studied regions. The variations are caused primarily by the marginal control technologies selected, which are determined primarily by air quality status in a region. Note that for each region, estimated emission values vary significantly with different studies. For example, for Southern California, the Tellus study estimated much higher emission values than the CEC study. This is because the marginal control costs estimated by the Tellus Institute were the highest costs of the "last" marginal control measure for the region, while CEC excluded control measures with costs above $100,000 and used the average cost of several marginal control measures. However, the CO value estimated by Tellus is lower than that estimated by CEC because different control technology lists were used in the two studies. These differences demonstrate the importance of the marginal control measures selected in determining control-cost-based emission values.

In reviewing the above studies, we also found that different studies may produce different control cost estimates for the same technology. For example, while the NYSEO study (1989) estimated a control cost of $7,281 per ton of NO_x from selective catalytic reduction, the California IEP study estimated a cost of $18,800 for the same technology (Schilberg et al. 1989). This is probably because different assumptions regarding equipment lifetime, emission reductions, and capital and operating costs were used in the two studies. Neither study presented the detailed assumptions used in the estimates.

When estimating emission values, the above studies were generally based on meeting air quality or emission standards from the current emission or air quality levels in the studied areas. Because most of the studied regions are nonattainment areas, the control-cost-based values were usually applied for nonattainment areas.

3.3 SUMMARY OF APPLICATIONS FOR THE TWO ESTIMATING METHODS

Both the damage value and the control cost method have been widely used in estimating emission values in past studies. However, assumptions and simplifications have to be made with either method; these assumptions, together with differences in air pollutant concentrations and total populations, lead to large variations in estimated emission values among the previous studies.

Large discrepancies in estimated emission values also occur between the damage value and the control cost method. Comparison of Tables 3 and 4 reveals that the damage-based values are generally lower than control-cost-based values. Because of exclusions of certain air pollution damage effects in damage-based studies and because of the many assumptions involved in these studies, we believe that damage-based values under-represent actual emission values, not that the lower damage-based values imply that emissions are over-controlled. Note that for PM_{10}, damage-based values are actually higher than control-cost-based values, which contradicts the argument that the damage estimate method results in lower emission values.

The vital assumption of the control cost method is that marginal control costs are equal to marginal emission damages. If we accept that legislators or regulators determine air quality standards and regulations primarily by considering the cost of achieving air quality goals and the damage caused by air pollution, use of the control cost method to estimate emission values may be preferred.

The damage value estimating method suffers from uncertainties in establishing dose-response functions and valuation of air pollution effects. In the damage-based studies described above, researchers generally adopted generic dose-response functions and generic values for air pollution effects cited from various studies; they often used linear extrapolation when applying pre-determined relationships and values and ignored some air pollution effects in estimating damage values. If we accept these assumptions, use of the damage value method to estimate emission values may be preferred.

The damage-based studies described in this section estimated damage values only for power plant emissions. Because power plants are generally not located in metropolitan areas, their emissions cause less damage than motor vehicle emissions in the core of a metropolitan area. Thus, applying the emission values based on power plant emission damages to motor vehicle emissions will probably lead to underestimates of the true damages of vehicle emissions.

Emission values estimated in the previous studies have been used to calculate the externality costs of air pollution. Koomey (1990) applied emission values estimated in some studies to electric power plant emissions to calculate externality costs of power plant air pollution. Because uncertainties are involved in all the past studies, Koomey did not have any preference regarding which study he used for his work; he used emission values estimated in ten past studies (both damage- and control-cost-based) and estimated ten sets of air pollution externality costs for power plants.

Small and Kazimi (1994) used emission damage results from some previous studies to estimate damage-based dollars-per-ton emission values. By applying the estimated emission values to motor vehicle, they calculated an air pollution externality cost of 3¢ cents per mile for passenger cars. Note that, although they intended to estimate motor vehicle externality costs, Small and Kazimi used air pollution damage results from past studies that were conducted primarily for stationary sources.

Although emission value estimates in the previous studies involve many assumptions and consequent uncertainties, some estimates must be chosen from these studies to complete societal cost and benefit analyses of projects that cause air pollutant emissions. Use of these estimates is particularly crucial because emission values in nonattainment areas cannot be zero, although emission values in attainment areas might be treated as zero. Consequently, many state PUCs have either adopted or proposed incorporatation of emission values into utility resource planning and acquisition. For information regarding state actions to incorporate air pollution externality costs into utility resource planning and acquisition, see Houston Lighting & Power Company (1993), Consumer Energy Council of American Research Foundation (1993), Ottinger et al. (1991), and Cohen et al. (1990).

4 DEVELOPMENT OF EMISSION VALUES FOR DIFFERENT U.S. REGIONS

Emission values must be chosen to evaluate the societal costs and benefits of projects that cause air pollutant emissions. The studies described in Section 3 estimated emission values only for a limited number of regions. In adopting or proposing emission values, many state PUCs and other private and public organizations apply the emission value estimates for those regions directly to their areas without adjustments for the differences in air quality status and total population. These differences can cause emission values to vary significantly among regions. Region-specific emission values have to be estimated to allow accurate calculation of environmental damages in a region.

Ideally, damage estimate models should be run for a particular region to estimate the damage values for that region, and emission control costs should be estimated by taking into account the control measures and their costs applied to the region. However, limited resources may prevent such detailed, accurate estimates for individual regions.

In this report, using the emission values estimated in previous original studies, we establish regression relationships between emission values, air pollutant concentrations, and total population. Air pollutant concentrations affect emission values directly; total population affects emission values more indirectly. For damage-based emission values, total population determines how many people are exposed to air pollution, and therefore determines the magnitude of health damage values — the most significant air pollution damage in most cases. For cost-based emission values, total population partly determines the number of emission sources in the region. A higher population requires more human services and leads to more activities, both of which result in more emission sources. Therefore a region with higher population incurs a higher cost to meet air quality standards than a region with a lower population, everything else being equal.

We apply established relationships to some U.S. metropolitan areas to estimate emission values for those regions. While our regression-based emission values for a metropolitan area may not be as accurate as the estimates developed using the damage value or the control cost method, they are more accurate than direct application of the emission value estimates for other regions to a study region.

To allow the flexibility of choosing between damage-based and control-cost-based emission values, we establish two sets of regression relationships: one for estimating damage-based values and the other for estimating control-cost-based values.

4.1 REGRESSION ANALYSIS

In establishing regression relationships, we used damage-based and control-cost-based emission values for eleven California air basins (estimated by CEC), damage-based emission values for the areas west of the Cascade range in Oregon (estimated by ECO Northwest), damage-based emission values for the northeastern United States (specifically Massachusetts and New York) (developed by Pace University), control-cost-based emission values for the northeastern United States (specifically Massachusetts) (estimated by the Tellus Institute), and damage-based emission values for the Las Vegas valley (estimated by NERA). Emission value estimates for 15 regions, then, were used to establish regression relationships (Appendix B contains the database created for the regression analysis).

In establishing a regression relationship for a particular pollutant, we tried various functional forms. We generally chose the most statistically significant functional form of the variables as the final regression relationship for each pollutant. However, in some cases, theoretical expectations for signs of coefficients caused us to adopt models with less "goodness of fit" (i.e., smaller R^2). During the regression analysis, we found that for some pollutants, the constant term was not significant. In those cases, we forced the constant term to be zero. We also found that some coefficients for air pollutant concentrations and/or populations were not statistically significant. However, we occasionally retained these relatively insignificant coefficients in the regression relationships because simple theory implies that both air pollutant concentrations and population affect emission values. Our established regression relationships for damage-based and control-cost based emission values are presented below.

4.1.1 Damage-Based Emission Value Relationships

Damage-based emission values for each pollutant were regressed against various combinations of and various functional forms of air pollutant concentrations and total population. The regression relationships found between emission damage values, air pollutant concentrations, and total population are given below. Table 5 presents the statistics for these relationships. Note that emission values here are expressed in 1989 constant dollars.

$$
\begin{aligned}
NO_{x,\,damage} &= 1{,}640 \ln(pop) + 4{,}220 \ln(O_3) \qquad (1)\\
ROG_{damage} &= 871 \ln(pop) + 2{,}310 \ln(O_3)\\
\ln(PM_{10,\,damage}) &= 0.764 \ln(pop) + 0.685 \ln(PM_{10})\\
\ln(SO_{x,\,damage}) &= 5.41 + 0.325 \ln(pop) + 0.0138 \ln(SO_2)
\end{aligned}
$$

where:

$NO_{x,\,damage}$ = NO_X damage value (\$/ton)

ROG_{damage} = ROG damage value (\$/ton)

$PM_{10,\,damage}$ = PM_{10} damage value (\$/ton)

$SO_{x,\,damage}$ = SO_X damage value (\$/ton)

pop = total population (in 10^3)

O_3 = highest second daily maximum 1-hr ozone concentration (ppm)

PM_{10} = highest arithmetic mean PM_{10} concentration ($\mu g/m^3$)

SO_2 = highest arithmetic mean SO_2 concentration (ppm)

Damage value estimates for CO are scarcer than those for the above four pollutants. Among the cited original studies, only the CEC study estimated CO damage values for California's air basins. CEC estimated a per-ton value of $3 for the South Coast Air Basin, $1 for both the San Francisco Bay area and San Diego, and $0 for other California air basins. The CEC estimates imply virtually zero damage value for CO, which certainly under-represents the actual damaging effects of CO in most urban areas. The CEC study estimated CO damage values based on power plant emissions. CO disperses rapidly and is not a problem at great distances from the source. While power plants and people are not generally located close together, motor vehicles and people generally are. CO emissions from motor vehicles are probably far more damaging to humans than those from power plants.

TABLE 5 Statistics of Regression Relationships for Damage-Based Values

Variable		NO_x	ROG	PM_{10}	SO_x
Regression	R^2	0.43	0.36	0.30	0.67
	F Value	7.39[a]	4.93[b]	4.55[b]	12.0[a]
Constant	Standard Error	N/A[c]	N/A	N/A	1.33
	t Value	N/A	N/A	N/A	4.05[d]
Population	Standard Error	371	248	0.179	0.0868
	t Value	4.43[d]	3.51[d]	4.27[d]	3.75[d]
Pollutant Concentration	Standard Error	137	881	0.353	0.148
	t Value	3.09[d]	2.62[d]	1.94[b]	0.0934[e]

[a] At the significance level of 99%.

[b] At the significance level of 95%.

[c] N/A = not available.

[d] At the significance level of 97.5%.

[e] Not significant.

4.1.2 Control-Cost-Based Emission Value Relationships

Control-cost-based emission values for each pollutant were regressed against various combinations of total population and air pollutant concentrations. The regression relationships found between control-cost-based emission values, population, and air pollutant concentrations are given below. Table 6 presents statistics for these regression relationships. Note that emission values here are expressed in 1989 constant dollars.

$$NO_{x,\,cost} = 40{,}000 + 5.71\ \ln(pop) + 151\ \ln(O_3) \tag{2}$$

$$ROG_{cost} = 30{,}200 + 385\ \ln(pop) + 120\ \ln(O_3)$$

$$PM_{10,\,cost} = -16{,}800 + 793\ \ln(pop) + 3{,}790\ \ln(PM_{10})$$

$$SO_{x,\,cost} = -51{,}100 + 956\ \ln(pop) + 13{,}500\ \ln(PM_{10})$$

$$CO_{cost} = -6{,}390 + 579\ \ln(pop) + 2{,}110\ \ln(CO)$$

where:

$NO_{x,\,cost}$ = NO_x control-cost-based value (\$/ton)

ROG_{cost} = ROG control-cost-based value (\$/ton)

$PM_{10,\,cost}$ = PM_{10} control-cost-based value (\$/ton)

$SO_{x,\,cost}$ = SO_x control-cost-based value (\$/ton)

CO_{cost} = CO control-cost-based value (\$/ton)

pop = total population (in 10^3)

O_3 = highest second daily maximum 1-hr ozone concentration (ppm)

PM_{10} = highest arithmetic mean PM_{10} concentration ($\mu g/m^3$)

CO = highest second maximum nonoverlapping 8-hr CO concentration (ppm)

The regression relationships for both damage-based and control-cost-based emission values take logarithmic forms. Tables 5 and 6 show that population is more significant than pollutant concentration in damage value regression relationships, but pollutant concentration is more significant in control cost regression relationships. This is consistent with the fact that damage values in the past studies were primarily determined by total population exposed, while control costs were primarily determined by air pollutant concentrations.

TABLE 6 Statistics of Regression Relationships for Control-Cost-Based Values

Variable		NO_x	ROG	PM_{10}	SO_x	CO
Regression	R^2	0.42	0.29	0.56	0.32	0.35
	F Value	5.99[a]	3.64[b]	9.73[c]	4.33[a]	4.47[a]
Constant	Standard Error	14900	16600	4800	19900	3000
	t Value	2.69[d]	1.82[a]	-3.49[d]	-2.56[d]	-2.13[a]
Population	Standard Error	1010	1120	254	1050	441
	t Value	0.00564[e]	0.345[e]	3.12[d]	0.907[e]	1.31[f]
Pollutant Concentration	Standard Error	4950	5540	1140	4740	1170
	t Value	3.06[d]	2.16[a]	3.32[d]	2.86[d]	1.81[g]

[a] At the significance level of 95%.

[b] At the significance level of 92.5%.

[c] At the significance level of 99%.

[d] At the significance level of 97.5%.

[e] Not significant.

[f] At the significance level of 85%.

[g] At the significance level of 90%.

4.2 ESTIMATES OF EMISSION VALUES FOR VARIOUS U.S. METROPOLITAN AREAS

4.2.1 Input Data

Based on the regression relationships established in Section 4.1, we have estimated emission values for some U.S. metropolitan areas. Table 7 presents data on air pollutant concentrations and population in 17 U.S. metropolitan areas. We apply these data to the regression relationships to estimate emission values for these metropolitan areas. The 17 regions selected include all nine of the nonattainment areas specified in the 1990 CAAAs for introducing reformulated gasoline (RFG): Baltimore, Chicago, Denver, Houston, Los Angeles, Milwaukee, New York, Philadelphia, and San Diego. Our regression estimates for four other metropolitan areas (Boston, Sacramento, San Francisco, and Las Vegas) are included here for comparison with the estimates in some original studies. The remaining metropolitan areas (Atlanta, New Orleans, San Joaquin Valley of California, and Washington, D.C.) are included because vigorous air pollution control measures are currently proposed in these areas. Our regression relationships can be used to estimate emission values for any target nonattainment area.

TABLE 7 Input Data Used in Regression Relationships[a]

Metropolitan Area	Total Population (10^3)	Ozone (highest 2nd daily maximum 1-hr concentration in ppm)	PM_{10} (highest arithmetic mean concentration in $\mu g/m^3$)	SO_2 (highest arithmetic mean concentration in ppm)	CO (highest 2nd maximum non-overlapping 8-hr concentration in ppm)
Atlanta	2,649	0.13	40	0.006	7
Baltimore	2,329	0.14	36	0.010	8
Boston	2,852	0.12	36	0.013	5
Chicago	6,156	0.12	46	0.018	6
Denver	1,696	0.11	37	0.008	11
Houston	3,258	0.22	32	0.008	8
Las Vegas	647	0.10	65	N/A[b]	13
Los Angeles	8,624	0.30	62	0.006	17
Milwaukee	1,408	0.15	36	0.007	6
New Orleans[c]	2,013	0.13	32	0.005	6
New York	8,535	0.16	52	0.019	11
Philadelphia	4,863	0.15	41	0.015	9
Sacramento	1,384	0.13	40	0.006	12
San Joaquin Valley[d]	2,405	0.14	63	0.004	9
San Diego	2,353	0.18	41	0.005	9
San Francisco Bay[e]	3,601	0.10	35	0.003	7
Washington, D.C.	3,739	0.13	36	0.014	9

[a] Source: EPA 1990b; 1991; 1992c. The data presented here are averages of three years (1989-1991).

[b] N/A = not available.

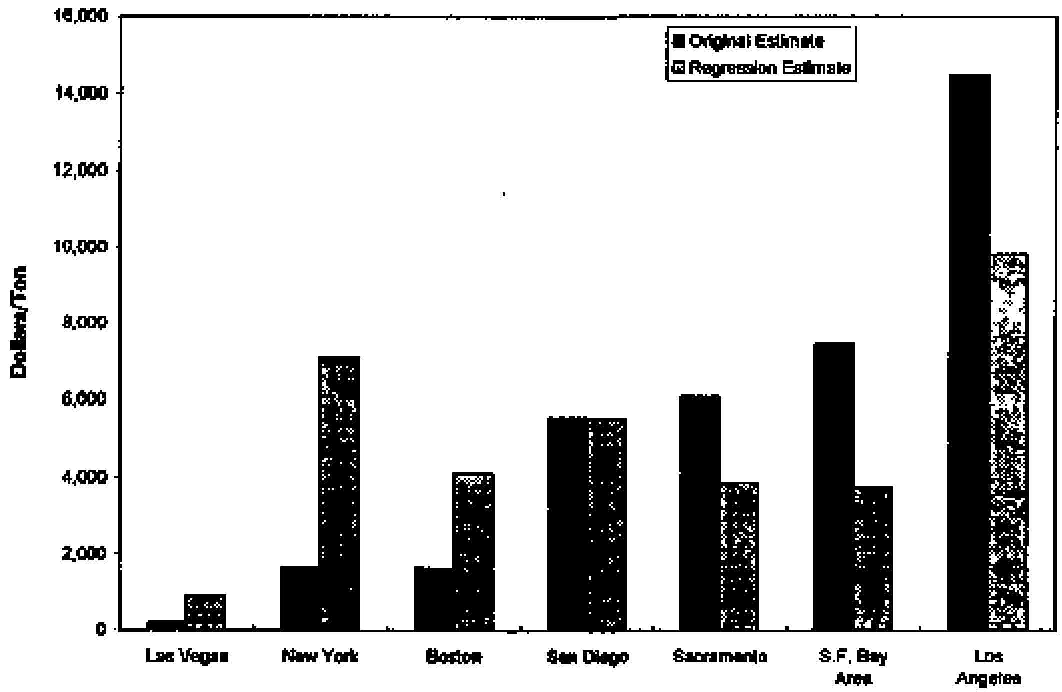
[c] New Orleans includes both the New Orleans and Baton Rouge areas.

[d] The San Joaquin Valley of California includes Bakersfield, Fresno, Merced, Modesto, Stockton, and Visalia-Tulare.

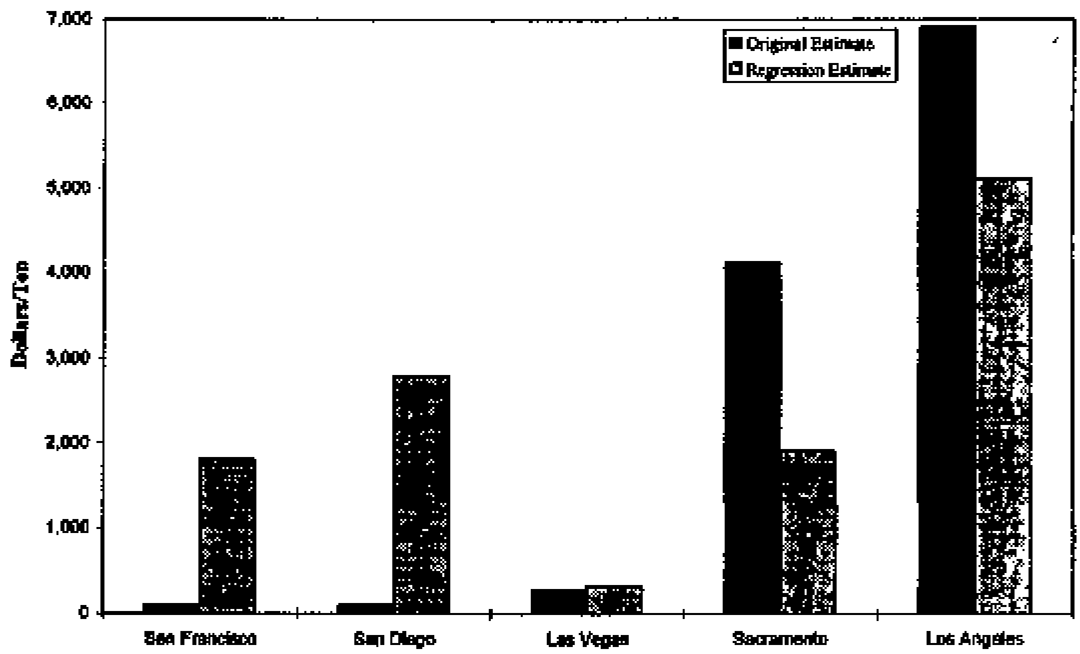
[e] The San Francisco Bay area includes both San Francisco and Oakland.

4.2.2 Estimated Emission Values

Damage-Based Estimates. Figures 3 and 4 present the regression-estimated emission values and original emission value estimates for Boston, Las Vegas, Los Angeles, New York, Sacramento, San Diego, and San Francisco. There are large differences between our estimates and the original damage-based estimates for ROG and PM_{10}. Our estimated SO_x values are close to the original SO_x values. The NO_x regression relationship underestimates the NO_x damage value in Los Angeles, Sacramento, and the San Francisco Bay area, but overestimates the value in Boston, New York, and Las Vegas. In San Diego, the regression-estimated NO_x value is comparable to the original estimate. The ROG regression relationship underestimates the ROG value in Los Angeles and Sacramento, but overestimates the value in San Francisco and San Diego. In Las Vegas, our estimate of ROG value is close to the original estimate. The PM_{10} regression-estimated value is substantially lower than the original estimate in Los Angeles, San Francisco, and San Diego, but higher than the original estimate in New York, Boston, Sacramento, and Las Vegas. The SO_x regression relationship underestimates the SO_x damage value in Los Angeles, San Francisco, and Boston, but overestimates the value in Sacramento. In New York and San Diego, our regression-estimated SO_x value is comparable to the original estimate. Overall, damage-based values in Los Angles are always underestimated by the regression relationships, and damage-based values in Boston and New York are usually overestimated.

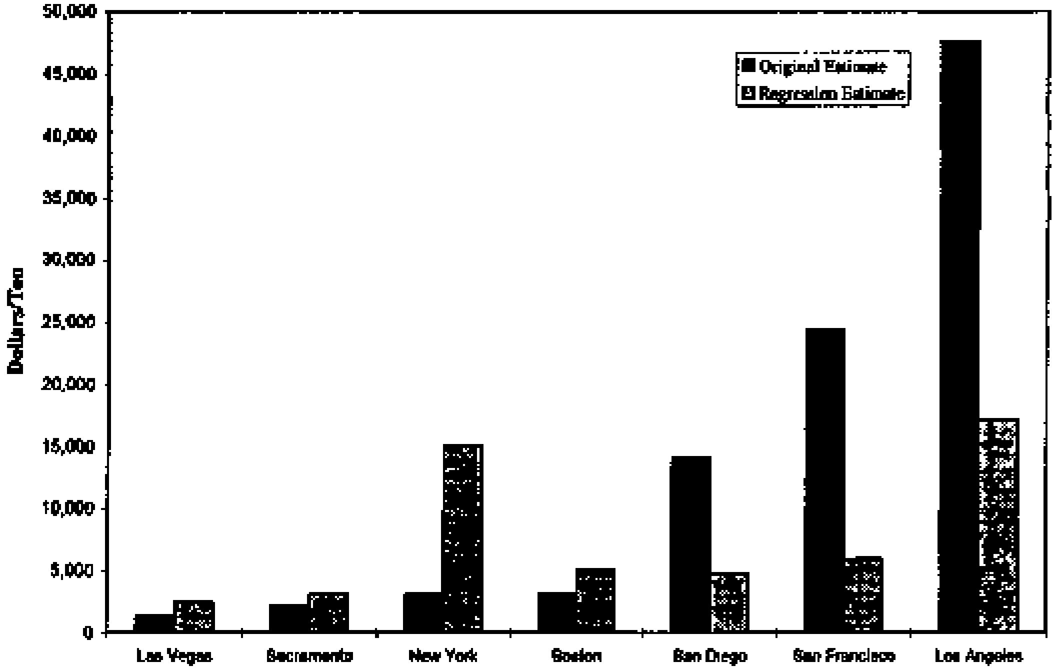


(a) Damage-Based Emission Values for NO_X

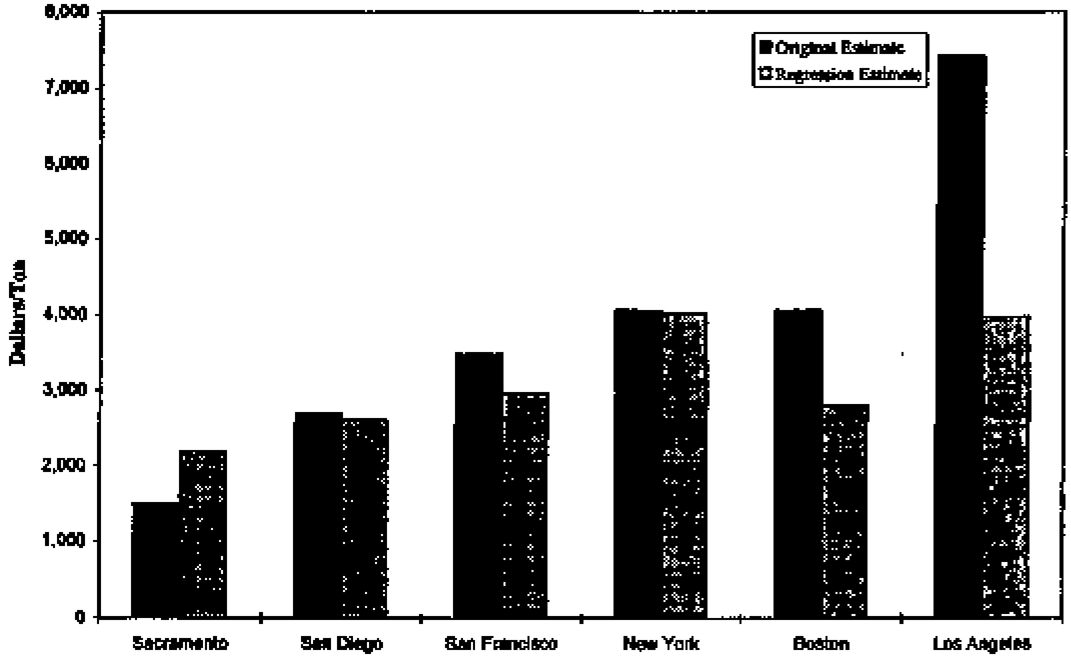


(b) Damage-Based Emission Values for ROG

FIGURE 3 Comparison between Regression Estimates and Original Estimates: Damage-Based Emission Values (1989 constant dollars)

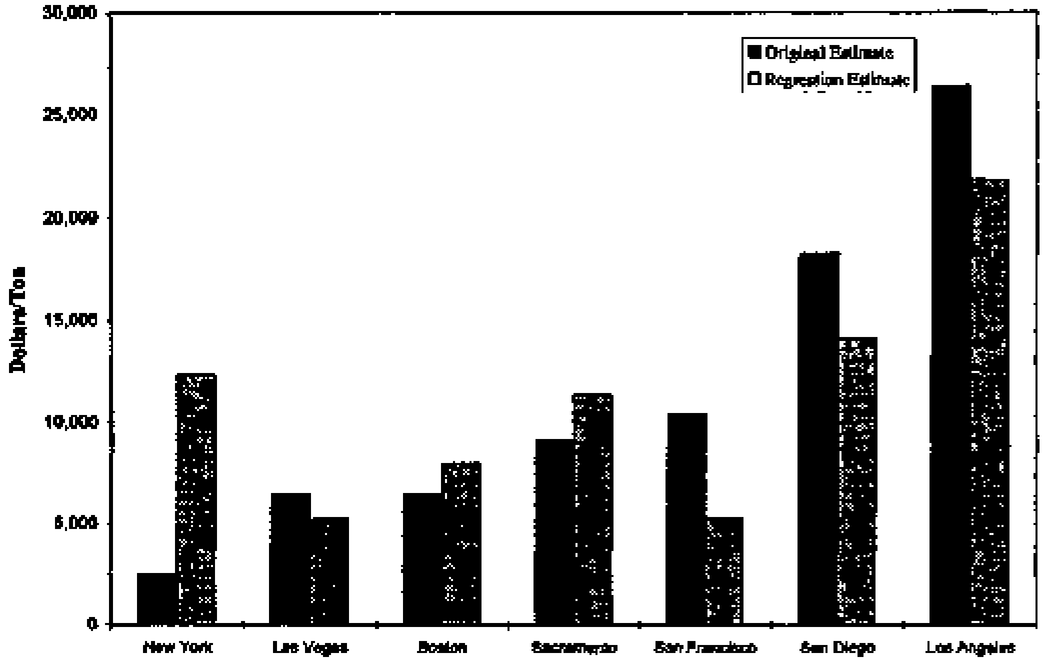


(c) Damage-Based Emission Values for PM_{10}

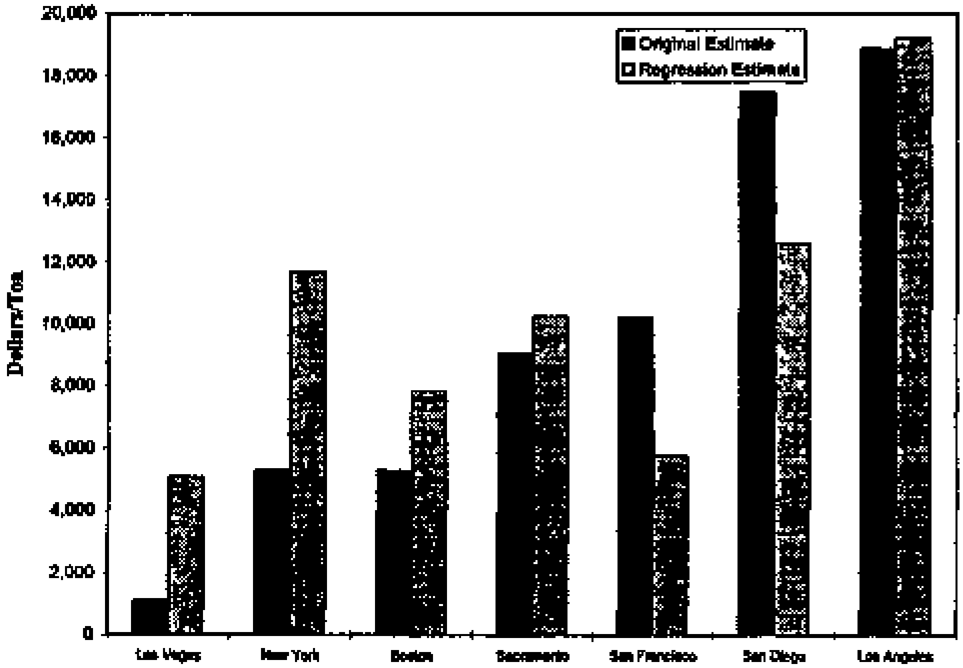


(d) Damage-Based Emission Values for SO_X

FIGURE 3 (Cont.)

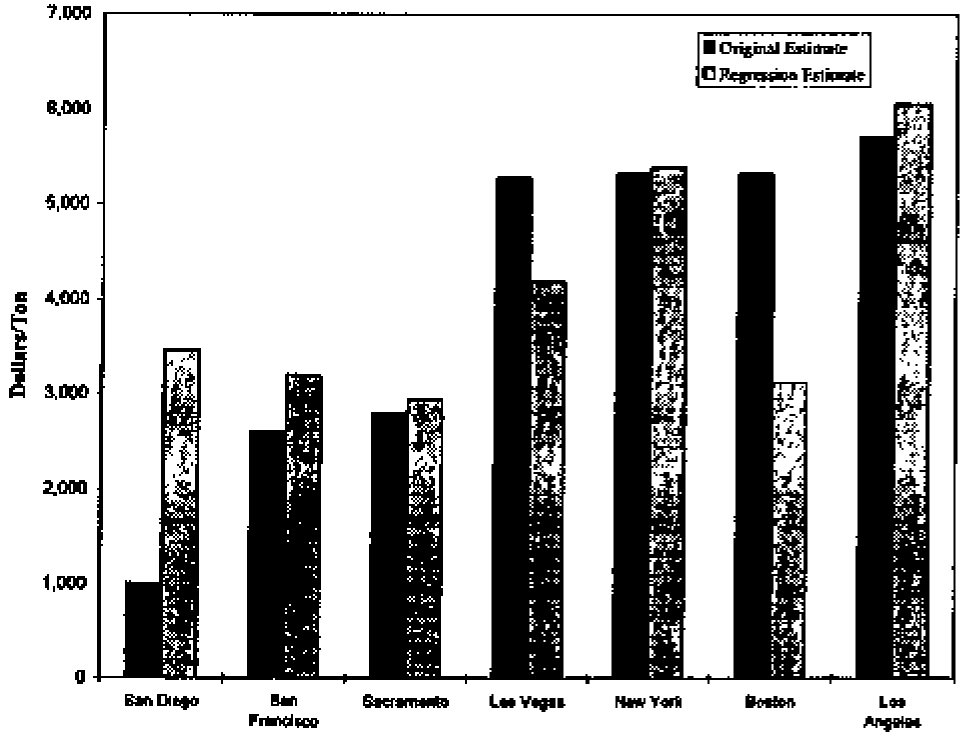


(a) Control-Cost-Based Emission Values for NO_x

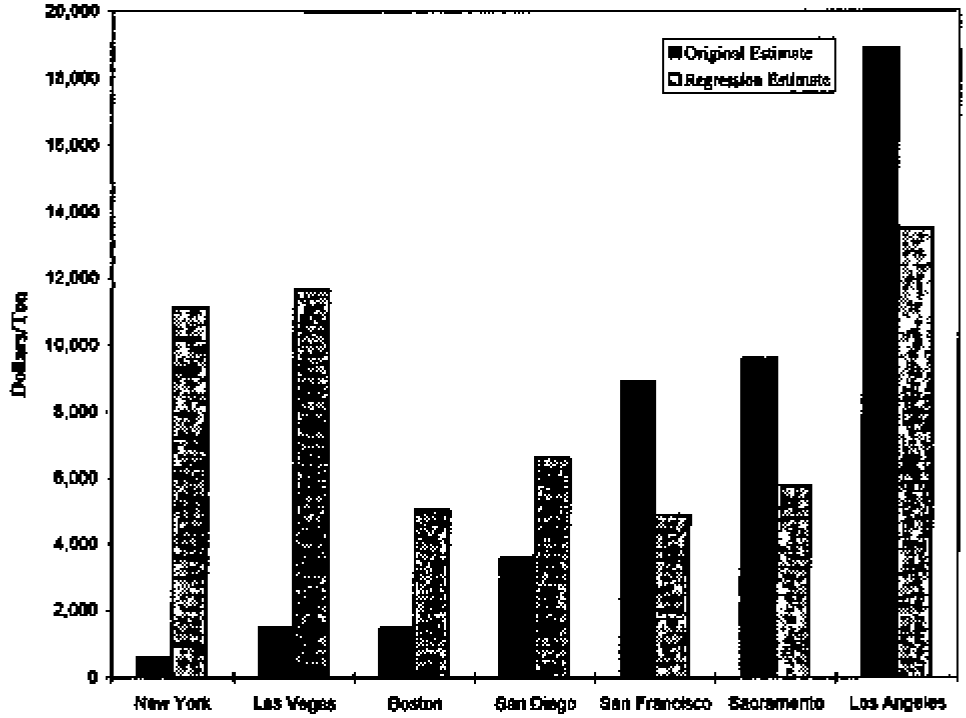


(b) Control-Cost-Based Emission Values for ROG

FIGURE 4 Comparison between Regression Estimates and Original Estimates: Control-Cost-Based Emission Values (in 1989 constant dollars)

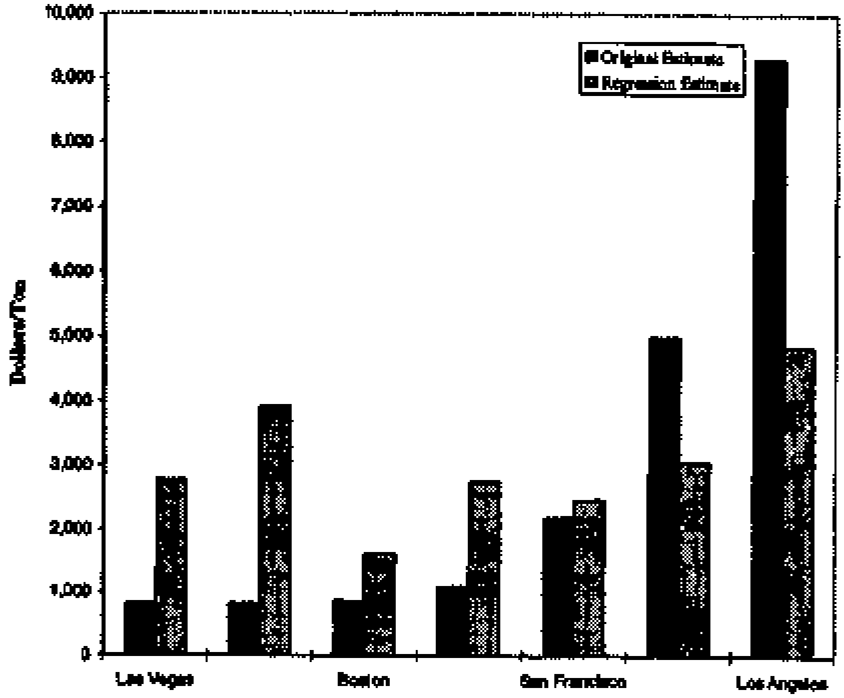


(c) Control-Cost-Based Emission Values for PM_{10}



(d) Control-Cost-Based Emission Values for SO_x

FIGURE 4 (Cont.)



(e) Control-Cost-Based Emission Values for CO

FIGURE 4 (Cont.)

Control-Cost-Based Estimates. Using the regression analyses, the NO_x estimate is lower than the original control-cost-based emission value in Los Angeles, San Francisco, San Diego, and Las Vegas, but higher than the original value in New York, Sacramento, and Boston. The regression-estimated value for ROG is lower than the original estimate in San Francisco and San Diego, but higher than the original estimate in New York, Las Vegas, and Boston. In Los Angeles and Sacramento, our estimate of the ROG emission value is comparable to the original estimate. The regression-estimated value for PM_{10} is close to the original estimate in Los Angeles, New York, and Sacramento. The PM_{10} regression relationship underestimates the PM_{10} value in Boston and Las Vegas, but overestimates the value in San Diego and San Francisco. The SO_x regression relationship underestimates the SO_x value in Los Angeles, Sacramento, and San Francisco, but overestimates the value in New York, Las Vegas, Boston, and San Diego. The CO regression relationship underestimates the CO value in Los Angeles and Sacramento, but overestimates the value in New York, San Diego, Las Vegas, and Boston. Our estimate of the CO value is close to the original estimate in San Francisco. Overall, differences in control-cost-based values between regression estimates and original estimates are smaller for PM_{10} than for any other pollutants. The differences are smaller for control-cost-based values than for damage-based values.

Because our regression relationships rely on original estimates, we recommend that, when available, original estimated emission values be used for relevant areas. Our purpose here is not to supplant a more careful study, but to provide working values until studies are completed for the various locations for which no estimates have been developed.

Table 8 presents emission values estimated by using the established regression relationships for the 17 nonattainment areas. Not surprisingly, there are significant variations in emission values across the 17 areas. Damage-based emission values vary from \$910 to \$9,800 for NO_x, \$320 to \$5,110 for ROG, \$2,450 to \$17,200 for PM_{10}, and \$2,190 to \$4,030 for SO_x. Control-cost-based emission values vary from \$5,220 to \$21,850 for NO_x, \$5,100 to \$19,250 for ROG, \$2,400 to \$6,060 for PM_{10}, \$3,130 to \$13,480 for SO_x, and \$1,410 to \$4,840 for CO. Emission values in Los Angeles are always high, while values in Las Vegas are usually low. Estimated damage-based emission values are generally lower than control-cost-based values for each pollutant except PM_{10} — probably because of underestimation of damage values in previous original studies, in which not all air pollution effects were considered.

4.2.3 Qualifications of the Estimated Emission Values

The above emission value estimates are based on the established regression relationships which, in turn, are based on previously estimated emission values. In the regression analysis, the selection of independent variables (population and air pollutant concentrations) and regression functional forms has affected the final relationships. Compared with original estimates for a given region, regression estimates are rather rough and can indicate only the magnitude that emission values might have for the region. Caution must be taken in using the regression estimates.

Researchers can use either damage-based or control-cost-based emission values — both have their advantages and disadvantages. One should be aware that selection of either type of value could have significant consequences.

Past estimates of emission values were based primarily on stationary source emissions. Therefore, the established regression relationships (based on these past studies) rely on stationary source estimates. Application of these values to mobile source emissions may under-represent the true values of mobile source emissions. Because many major stationary sources are located away from metropolitan areas, while emissions from motor vehicles occur primarily in or near the core of metropolitan areas, damage-based values for mobile source emissions are likely to exceed those for stationary source emissions. This is especially true for mobile source CO emissions, because these emissions in street canyons pose extensive population exposure. With respect to cost-based emission values, very few control measures for mobile source emissions were included in the original studies. Again, the established regression relationships are based primarily on estimated emission control costs for stationary sources. Emission values based on stationary source control costs may be higher or lower than those based on both stationary and mobile source control costs.

TABLE 8 Estimated Emission Values for 17 U.S. Regions

Area	Emission Value ($/ton, 1989 dollars)				
	NO_x	ROG	PM_{10}	SO_x	CO
	Damage-Based				
Atlanta	4,330	2,150	5,170	2,720	N/A[a]
Baltimore	4,430	2,210	4,520	2,620	N/A
Boston	4,120[b]	2,030	5,090[b]	2,820[b]	N/A
Chicago	5,380	2,700	10,840	3,600	N/A
Denver	2,840	1,350	3,390	2,330	N/A
Houston	6,890	3,540	5,190	2,910	N/A
Las Vegas	910[b]	320[b]	2,450[b]	N/A[b]	N/A
Los Angeles	9,800[b]	5,110[b]	17,200[b]	3,970[b]	N/A[b]
Milwaukee	3,890	1,930	2,960	2,210	N/A
New Orleans	3,880	1,910	3,600	2,471	N/A
New York	7,130[b]	3,650	15,130[b]	4,030[b]	N/A
Philadelphia	5,940	3,010	8,360	3,340	N/A
Sacramento	3,870[b]	1,920[b]	3,160[b]	2,190[b]	N/A[b]
San Diego	5,510[b]	2,800[b]	4,800[b]	2,600[b]	N/A[b]
San Francisco Area	3,730[b]	1,810[b]	5,970[b]	2,970[b]	N/A[b]
San Joaquin Valley	4,490	2,240	6,550	2,610	N/A
Washington, D.C.	4,900	2,450	6,260	3,070	N/A
	Control-Cost-Based				
Atlanta	9,190	8,780	3,460	6,420	2,280
Baltimore	10,810	9,620	3,170	5,600	2,490
Boston	7,980[b]	7,850[b]	3,120[b]	5,060[b]	1,610[b]
Chicago	7,990	8,150	4,660	9,120	2,440
Denver	6,660	6,590	2,790	4,900	2,960
Houston	17,150	15,160	2,780	3,590	2,680
Las Vegas	5,220[b]	5,100[b]	4,190[b]	11,650[b]	2,770[b]
Los Angeles	21,850[b]	19,250[b]	6,060[b]	13,480[b]	4,840[b]
Milwaukee	11,350	10,250	2,560	4,380	1,590
New Orleans	9,190	8,670	2,400	3,130	1,410
New York	12,340[b]	11,720[b]	5,390[b]	11,090[b]	3,910[b]
Philadelphia	11,360	10,730	4,040	7,330	3,160
Sacramento	11,350[b]	10,240[b]	2,950[b]	5,800[b]	3,040[b]
San Diego	14,110[b]	12,630[b]	3,460[b]	6,640[b]	2,740[b]
San Francisco Area	5,230[b]	5,760[b]	3,200[b]	4,900[b]	2,460[b]
San Joaquin Valley	10,310	9,630	5,110	12,480	2,750
Washington, D.C.	9,190	8,910	3,340	5,320	3,010

[a] NA = not available.

[b] For these regression estimates, original estimates of emission values are available. They are presented in Table B.1.

4.3 VALUES OF GREENHOUSE GAS EMISSIONS

Some previous studies described in this report presented emission values for greenhouse gases. These studies usually estimated emission values for CO_2 using the control cost estimating method. Emission values for other greenhouse gases are generally calculated using the estimated CO_2 value and the global warming potentials of other greenhouse gases relative to that of CO_2. Table 9 presents previous CO_2 emission value estimates. The table shows large variations in estimated CO_2 emission values. These variations are due partly to the cited studies' assumptions regarding CO_2 control measures and the level of CO_2 emissions controlled. On the basis of estimated CO_2 emission values cited in the studies, we suggest a median value of $15 per ton for CO_2 emissions.

As with past studies, we applied the global warming potentials of other greenhouse gases to the value of $15 per ton of CO_2 to calculate emission values for other greenhouse gases. Global warming potentials and calculated emission values for the greenhouse gases are presented in Table 10. CO_2 emission values estimated in the cited studies are based primarily on the control cost estimating method. To apply the global warming potentials of various greenhouse gases in calculating emission values, researchers would have to implicitly interpret the estimated CO_2 values as damage values, therefore creating an inconsistency between the original control cost estimates and their intended use. Ideally, either control costs should be estimated for other greenhouse gases, or values for CO_2 should be estimated using the damage value method — then the global warming potentials of greenhouse gases can be applied to calculate their damage values.

TABLE 9 CO_2 Emission Values Estimated in Past Studies

Study	Value ($/ton)	Remarks	Control Measure Specified
NYSEO 1989	6		Reforestation
Schilberg et al. 1989	14.7		Reforestation
Bernow and Marron 1990	22		Tree planting
Ottinger et al. 1991	4.6-13.6		Not specified
PSCN 1991	22		Tree planting
NYSEO 1991	74 5.5	Marginal cost Average cost	Based on a revenue tax Based on a trust fund tax
Nordhaus 1992	0.8 1.4	Average cost Marginal cost	10% worldwide reduction in CO_2 over the 1989 level
	13.0 32.7	Average cost Marginal cost	50% worldwide reduction in CO_2 over the 1989 level
NERA 1993	3.2		Not specified
OPUC 1993	10-40		Not specified
CEC 1993	7.6		Not specified
Morris et al. 1993	5.2	Average cost	$15/ton carbon tax
	13.7 90-900	Average cost Marginal cost	U.S. CO_2 stable between 1990 and 2030
	18.8 117-2,102	Average cost Marginal cost	10% reduction in U.S. CO_2 between 1990 and 2030

TABLE 10 Global Warming Potentials and Emission Values of Greenhouse Gases

Greenhouse Gas	Global Warming Potential: Bernow and Marron 1990	DeLuchi 1991[a]	NERA 1993	Our Adopted Value	Emission Value ($/ton, 1989 dollars)[b]
CO_2	1	1	1	1	15
CH_4	10	9	10	10	150
N_2O	180	190	180	180	2,700
CO	2.2	2	2.2	2.2	33
NMOG (C weight)	N/A[c]	7	N/A	7	105
NO_x	N/A	14	N/A	14	210
CFC-11	N/A	N/A	1,300	1,300	19,500
CFC-12	N/A	4,500	3,700	3,700	55,500

[a] DeLuchi's global warming potentials for a 500-year horizon are cited here because of their consistency with the global warming potentials in the other two studies.

[b] We adopted a value of $15 per ton of CO_2 and calculated values for other pollutants by using the adopted CO_2 value and the global warming potentials of the other pollutants.

[c] N/A = not available.

5 CONCLUSIONS

We have reviewed two general methods of estimating the monetary values of air pollutants: the damage value method and the control cost method. Using the damage value method, researchers directly estimate the values of air pollutant emission damages by simulating air quality impacts of a given amount of emissions, estimating health and other welfare impacts of the resulting air quality changes, and calculating estimated monetary values of the health and other welfare impacts. For the control cost method, the marginal emission control cost required to meet given air quality or emission standards is estimated. This cost represents the opportunity cost offset by emission reductions from a given source, and is treated as the value for emissions reduced by the source. Although the damage value method is theoretically sound, many assumptions and uncertainties are involved in its estimating procedures. Consequently, estimated emission values may not accurately represent true emission values. On the other hand, the opportunity cost estimated using the control cost method may or may not represent the value that society places on a given air pollutant.

Numerous studies have been conducted to estimate the values of air pollutant emissions. We have reviewed the major studies in this report. As in our study, many of the reviewed studies are secondary sources; that is, the researchers did not conduct original estimates of emission values. Both the damage value and control cost methods were used in the past studies. By and large, damage-based estimates tended to be incomplete and to be lower than control-cost-based estimates, except for PM_{10}, for which damage-based values are higher than control-cost-based values. Not surprisingly, these original studies used different methods and assumptions, and their results are difficult to reconcile. The studies have revealed wide variations in emission values among regions — region-specific estimates of air pollutant emission values are certainly needed.

Using emission values estimated by some original studies for 15 U.S. air basins, we established relationships between emission values on the one hand and total population and air pollutant concentrations on the other hand. The established regression relationships take logarithmic forms. On the basis of the established relationships, we have estimated both damage-based and control-cost-based emission values for 17 major U.S. urban regions. Our estimates show that emission values vary significantly among regions and among pollutants. Although the regression-estimated emission values may not be as accurate as the estimates conducted using the damage value or the control cost method for a particular region, our estimates are more accurate than direct application of the estimates for other areas to the study region. Ideally, emission values should be estimated for each region.

Our report summarizes greenhouse gas emission values developed during past studies. Based on past estimates, we proposed a per-ton CO_2 emission value of $15 and developed emission values for other greenhouse gases based the proposed CO_2 value and the global warming potentials of various greenhouse gases.

Emission value estimates in past studies are primarily for stationary source emissions. Consequently, our regression estimates (based on past studies) are applicable to stationary source emissions. Estimates of the values for mobile source emissions are lacking; virtually no damage-based estimates have been developed for mobile source emissions. Although researchers have estimated the cost-effectiveness of various mobile source emission control measures, the estimated mobile source control costs have not been used to approximate mobile source emission values. Use of emission values estimated in this report may understate the true value of mobile source emissions. The only exception may be emission values for greenhouse gases, for which the differences in values between mobile sources and stationary sources may not be significant.

Because our regression relationships rely on original emission value estimates, we recommend that these values be used for relevant areas, when available. Our purpose here is not to supplant a more careful study, but to provide working values until studies are completed for the various locations for which no value estimates have been developed. We strongly believe that accurate estimates of emission values using either the damage value or the control cost method are needed for various individual regions and for mobile source emissions.

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APPENDIX A:

SAMPLE CALCULATION OF EMISSION CONTROL COSTS OBTAINED BY USING DIFFERENT CALCULATING TECHNIQUES

APPENDIX A: SAMPLE CALCULATION OF EMISSION CONTROL COSTS OBTAINED BY USING DIFFERENT CALCULATING TECHNIQUES

We have presented four techniques of calculating the cost effectiveness of emission control measures. In this appendix, we calculate control cost effectiveness using each technique for two sample control measures: a hypothetical stationary source emission control technology and electric vehicles as a mobile source emission control measure. Our intention here is to show the differences in the calculated results and their meanings for each different calculation technique.

The four techniques are presented in Table 1 of the main text. Some detailed information regarding the techniques and assumptions used in the following sample calculation is presented in an early report first authored by Warriner (Warriner et al., 1993). For that detailed information, we refer readers to the Warriner report.

A.1 Emission Control Cost-Effectiveness of a Hypothetical Stationary Control Technology

The assumptions for a hypothetical stationary control technology are presented in Table A.1.

Table A.2 presents calculated control cost-effectiveness with three different discount rates using the four techniques. The table shows significantly different costs and meanings calculated using the different techniques. In practice, techniques 2 or 3 should be used in calculating control cost-effectiveness, because the results obtained using these two techniques do not account for the equipment's lifetime, which would allow the cost-effectiveness of various control measures with varying lifetimes to be directly compared. To evaluate costs and benefits of control measures from a societal accounting point of view, we believe that discounting should be applied to both costs and emissions. Therefore, we recommend that case a of both techniques 2 and 3 be used in calculating control cost-effectiveness.

TABLE A.1 Assumptions of Hypothetical Stationary Emission Control Technology

Lifetime of the equipment (year)	10
Annual emission reduction (tons/year)	50
Initial capital cost of the equipment ($)	1,750,000
Annual operating cost of the equipment ($/year)	175,000

TABLE A.2 Control Cost-Effectiveness of Hypothetical Stationary Emission Control Technology

Technique	Case	Discount Rate (%) 4	6	8	Units	Meaning
1	a	63,388	60,760	58,485	(\$/lifetime)/ (ton/yr)	Cost to reduce one ton each year throughout lifetime
	b	63,388	60,760	58,485		
2	a	7,815	8,255	8,716	\$/ton	Cost to reduce one ton
	b	7,815	8,255	8,716		
3	a	7,815	8,255	8,716	\$/ton	Cost to reduce one ton
	b	6,339	6,076	5,849		
4	a	964	1,122	1,299	(\$/year)/	Annual cost to reduce
	b	964	826	872	(ton/lifetime)	one ton throughout lifetime

Table A.2 shows that, for constant annual emission reductions, cases a and b for technique 1 or 2 yield the same control costs. This is because, with constant annual emission reductions, levelized annual emission reductions are the same as the straight average of annual emission reductions. The table also shows that case a of techniques 2 and 3 yield the same results, because under this case, discounting is applied to both costs and emissions for each of the two techniques.

Note that the control costs calculated using techniques 1, 2, and 3 decline as the discount rate increases.

A.2 Emission Control Cost-Effectiveness of Electric Vehicles

We use electric vehicles (EVs) as another example of calculating emission control cost-effectiveness to show control costs calculated using the four techniques under varying annual emission reductions and varying annual operating costs.

We have calculated emission control costs of EVs relative to gasoline vehicles (GVs). Tables A.3 and A.4 present the assumptions for EVs and baseline GVs. Table A5 presents control costs calculated using the four techniques.

TABLE A.3 Assumptions for Electric Vehicles and Baseline Gasoline Vehicles: General Parameters[a]

Increase in initial EV costs ($)	5,000
EV and GV lifetime (yr)	12
Gasoline price ($/gal)	1.20
GV fuel economy (mi/gal)	28
Electricity price (cents/kWh)	4.5
EV electricity consumption (kWh/mi)	0.35
Battery lifetime (yr)	3
Battery cost ($)	3,000
Ratio of EV O&M[b] cost to GV O&M cost	0.6
NMHC[c] reduction by Evs (%)	90
CO reduction by EVs (%)	90
NO_x reduction by EVs (%)	50

[a] These assumptions are based on Wang et al. (1993).

[b] O&M = operation and maintenance.

[c] NMHC = non-methane hydrocarbon.

TABLE A.4 Assumptions for Electric Vehicles and Baseline Gasoline Vehicles: Annual Parameters

Year	Annual VMT[b]	GV Emissions (g/mi)[a]			GV fuel cost ($/yr)	GV O&M cost[b] ($/yr)	EV fuel cost ($/yr)	EV battery cost ($/yr)	EV emission reduction (lb/yr)[c]	EV net cost ($/yr)
		NMHC	CO	NO_x						
1	12,900	0.503	2.623	0.291	553	132	203	0	51.5	-402
2	12,600	0.565	4.026	0.376	540	289	198	0	70.7	-457
3	12,300	0.659	6.140	0.507	527	368	194	0	99.0	-481
4	11,900	0.747	8.087	0.631	510	415	187	3,000	122.7	2,511
5	11,500	0.829	9.912	0.748	493	447	181	0	142.9	-491
6	11,000	1.150	14.052	1.005	471	468	173	0	192.3	-465
7	10,600	1.443	17.825	1.249	454	477	167	3,000	213.3	2,522
8	10,100	1.715	21.299	1.480	433	488	159	0	266.3	-469
9	9,600	1.974	24.574	1.698	411	488	151	0	291.9	-455
10	9,100	2.214	27.595	1.905	390	489	143	3,000	310.6	2,558
11	8,700	2.426	30.222	2.103	373	86	137	0	325.5	-270
12	8,200	2.637	32.798	2.290	351	478	129	0	333.1	-413

[a] From Wang et al. (1993). Emissions of a 1995 GV were estimated by using EPA's Mobile 5a. NMHC emissions here include both exhaust and evaporative emissions.

[b] Vehicle miles traveled, from FHWA (1992).

[c] A composite emission reduction is calculated from emission reductions for NMHC, CO, and NO_x with relative damage values of 1, 0.49, and 1.4 for NMHC, CO, and NO_x, respectively. These relative damage values are from Wang et al. (1993).

TABLE A.5 Control Cost-Effectiveness of Electric Vehicles

		Discount Rate (%)				
Technique	Case	4	6	8	Unit	Meaning
1	a	81,436	80,026	79,082	($/lifetime)/	Cost to reduce one ton
	b	76,086	72,250	68,986	(ton/yr)	each year throughout lifetime
2	a	8,677	9,545	10,494	$/ton	Cost to reduce one ton
	b	8,107	8,618	9,154		
3	a	8,677	9,545	10,494	$/ton	Cost to reduce one ton
	b	6,341	6,021	5,749		
4	a	925	1,139	1,392	($/year)/	Annual cost through-
	b	676	718	763	(ton/lifetime)	out lifetime to reduce one ton

Table A.5 shows that, with varying annual emission reductions and costs, control costs calculated using different techniques are different, except that techniques 2a and 3a result in the same control costs. Under varying annual emission reductions and annual costs, use of a high discount rate in emission control cost calculations could result in an increase or a decrease in control costs, depending on which technique is used.

The two examples presented in this appendix are intended to demonstrate the differences in control costs calculated using different techniques. Although we have tried to use reasonable assumptions for the two examples, the control costs calculated here should be used only to compare various calculating techniques.

APPENDIX B:

DATABASE FOR REGRESSION ANALYSIS BETWEEN EMISSION VALUES AND AIR POLLUTANT CONCENTRATIONS AND POPULATION

TABLE B.1 Database for Regression Analysis between Emission Values and Air Pollutant Concentrations and Population

Region	Damage-Based Emission Values (1989 dollars, $/ton)					Control-Cost-Based Emission Values (1989 dollars, $/ton)					Air Pollutant Concentrations [a]					
	NO_X	ROG	CO	PM_{10}	SO_X	NO_X	ROG	CO	PM_{10}	SO_X	O_3	NO_2	PM_{10}	SO_2	CO	Total Pop. (10^3)[e]
South Coast Basin[b]	14,483	6,911	3	47,620	7,425	26,400	18,900	9,300	5,700	19,800	0.28	0.052	63	0.004	14	13,183
San Joaquin Valley[b]	6,473	3,711	0	3,762	1,500	9,100	9,100	3,200	5,200	17,600	0.14	0.027	63	0.004	9	2,404
San Fransico Area[b,c]	7,435	90	1	24,398	3,482	10,400	10,200	2,200	2,600	8,900	0.11	0.025	35	0.003	8	5,825 1,816
Sacramento Valley[b]	6,089	4,129	0	2,178	1,500	9,100	8,100	5,000	2,800	9,600	0.14	0.023	39	0.008	11	
Ventura Co. [b]	1,647	286	0	4,106	1,500	16,500	21,100	0	1,800	6,200	0.16	0.025	38	0.001	4	642
Santa Barbara[b,d]	1,647	286	0	4,106	1,500	9,100	9,100	0	900	3,000	0.13	0.024	36	0.002	4	351
North Central Coast[b]	1,959	603	0	2,876	1,500	9,100	9,100	0	900	3,000	0.09	0.011	24	0.001	2	572
San Diego[b]	5,559	98	1	14,228	2,676	18,300	17,600	1,100	1,000	3,600	0.18	0.03	41	0.006	9	2,857
North Coast[b,e]	791	467	0	551	1,500	6,000	3,500	0	900	3,000	0.1	0.029	44	0.003	3	222
Southeast Desert[b,e]	439	157	0	680	1,600	6,000	3,500	2,900	5,700	19,700	0.17	0.036	76	0.003	10	225
CA O_3 Attainment and PM_{10} Violation[b,f]	N/A[g]	N/A	N/A	N/A	N/A	6,000	3,500	0	900	3,000	0.11	0.029	60	0.003	7	152
West of Cascade Range[h]	849	N/A	N/A	1,973	N/A	3,363	N/A	N/A	8,843	1,400	0.11	N/A	31	0.006	8	1,677
Eastern Massachusetts[i]	1,640	N/A	N/A	5,152	4,060	6,600	6,800	620	6,333	1,500	0.12	0.032	33	0.012	6	4,408

TABLE B.1 (Cont.)

Region	Damage-Based Emission Values (1989 dollars, $/ton)					Control-Cost-Based Emission Values (1989 dollars, $/ton)					Air Pollutant Concentrations [e]					Total Pop. (10^3)[a]
	NO_x	ROG	CO	PM_{10}	SO_x	NO_x	ROG	CO	PM_{10}	SO_x	O_3	NO_2	PM_{10}	SO_2	CO	
Greater New York Area[j]	1,640	N/A	N/A	3,152	4,060	2,469	6,300	820	6,383	203	0.16	0.043	46	0.017	10	11,417
Las Vegas Valley[k]	211	0	N/A	1,364	286	6,297	1,093	882	6,161	1,445	0.1	0.034	65	NA	13	647

[a] Data on air pollutant concentrations and total population for each MSA are from EPA's air quality and emission trends report (EPA 1990b; 1991; 1992c). The values presented here are average values for 1989-91.

[b] Damage- and control-cost-based emission values for California air basins are from CEC (1993b).

[c] The San Francisco area includes San Francisco metropolitan statistical area (MSA) and Oakland MSA.

[d] Emission values estimated by CEC for the South Central Coast Air Basin were adopted for Santa Barbara.

[e] Two sets of air pollutant concentration measurements are available. One set is EPA's measurements presented in its air quality and emission trends report (EPA 1990b; 1991; 1992c). EPA presents its measurements for each MSA nationwide. The other set is CARB's measurements. CARB presents its measurements for each county in the state (see BER 1992a). In establishing regression relationships, we used EPA's air pollutant concentration measurements. No EPA measurements for the North Coast Air Basin or the Southeast Desert Air Basin are available. We used EPA and CARB measurements available for other California air basins to establish regression relationships between EPA and CARB measurements. We then used the established relationships to estimate EPA measurements from CARB measurements for the North Coast and Southeast Desert Air Basins.

[f] The ozone attainment and PM_{10} violation areas in California include four counties: Mendocino, Siskiyou, Modoc, and Lassen. Portions of Placer and El Dorado Counties belonging to these areas were not considered here. The CEC-estimated emission values for this area were calculated using the control cost method, but not the damage value method. EPA measurements of air pollutant concentrations for this area were estimated using the established relationships between EPA measurements and CARB measurements (see footnote e).

[g] N/A = not available.

[h] The damage-based values are from ECO Northwest's study for the Bonneville Power Administration (1991). The control-cost-based values are from OPUC's estimates (1993). The area includes Portland, Salem, Eugene-Springfield and Medford, Oregon. Air pollutant concentrations are population-weighted concentrations from the four MSAs.

[i] The damage-based values are from Pace University's estimates (Ottinger et al. 1991). The control-cost-based values are from MDPU's estimates (1989). The area includes Boston, Brockton, Fall River, Fitchburg-Leominster, Lowell, New Bedford, Salem-Gloucester, and Worcester. Air pollutant concentrations in the area are population-weighted averages among the eight MSAs.

[j] The damage-based values are from Pace University's estimates (Ottinger et al. 1991). The control-cost-based values are from NYSEO (1989). The area includes New York, Nassau-Suffolk, and Poughkeepsie. Air pollutant concentrations in the area are population-weighted averages among the three MSAs.

[k] The damage-based values are NERA's estimates (1993). The cost-based values are based on PSCN's estimates for the entire state (1991).

APPENDIX C:

EMISSION CONTROL COST-EFFECTIVENESS OF MOBILE SOURCE CONTROL MEASURES

APPENDIX C: EMISSION CONTROL COST-EFFECTIVENESS OF MOBILE SOURCE CONTROL MEASURES

The past studies described in this report estimated emission values based on emissions from stationary sources in general, and from power plants in particular. The primary purpose of those studies was to establish regulations to incorporate air pollution externality costs of power plants in the utility resource planning and acquisition process. Although the study by Small and Kazimi (1994) estimated damage-based emission values for motor vehicles, the fundamental relationships used in their study were developed primarily from studies on stationary source emission values.

Emission value estimates based on emissions from mobile sources should be different from those based on emissions from power plants. Damage-based emission values for mobile source emissions might be higher than those for stationary source emissions mainly because mobile source emissions occur in downtown areas and other activity centers where human exposure is intensive, while stationary source emissions (especially power plant emissions) occur outside human activity centers. The location of emissions is especially important for localized air pollution such as CO pollution. For example, while some past studies (e.g., the CEC study) have estimated virtually zero damage value for CO emissions from power plants, CO emissions from motor vehicles in urban streets certainly cause damages to human health. No studies have yet been conducted to estimate the damage values of motor vehicle CO emissions. Such studies are needed to accurately evaluate the benefit of various mobile source CO emission control measures.

Control-cost-based emission values used in past studies have relied almost entirely on emission control costs estimated for stationary sources. Emission values based on stationary and mobile source control costs would certainly be different. The exclusion of mobile source emission control costs in estimating emission values may be caused by two factors. First, the past studies were intended to estimate emission values for stationary sources, particularly power plants; researchers may have believed that mobile source control costs were irrelevant for such studies. Second, estimated emission control costs for mobile sources were less comprehensive and subject to greater uncertainties. For example, the assumed baseline vehicle emissions and vehicle lifetime are critical in determining the control costs of motor vehicle emissions. Yet, there are many uncertainties regarding both of these factors. The assumptions made in some past studies to estimate mobile source emission control costs were not explicit, which makes it difficult to compare the studies.

Table C.1 summarizes past studies on mobile source emission control costs. As the table shows, because assumptions involved in different studies are different, there are widevariations in the estimated mobile source emission control cost-effectiveness between studies and even within some of the same studies. Nevertheless, the table shows that gasoline Reid vapor pressure (RVP) reduction, Stage II refueling emission control, on-board refueling control, enhanced inspection and maintenance (I/M) programs, old car scrappage, gross emitter repair, and oxygenated fuels are generally cost-effective methods of reducing mobile source emissions. Costs of alternative-fuel vehicles (AFV) vary widely, depending on studies. However, compressed natural gas (CNG) vehicles in general cost less than other vehicle types.

TABLE C.1 Cost-Effectiveness of Mobile Source Emission Control Measures

Control Measure	Cost ($/ton, 1989 dollars)	Pollutant	Source
Gasoline RVP reduction	330 to 730	VOC	OTA 1988
	3,150	VOC	Kinnear 1992
Stage II refueling emission control	1,040	VOC	OTA 1988
	1,040	VOC	SRI 1994
	2,160	VOC	Kinnear 1992
On-board refueling emission control	900	VOC	Kinnear 1992
	1,250 to 1,460	VOC	OTA 1988
	2,260	VOC	SRI 1994
Enhanced I/M program	350	VOC	SRI 1994
	450	VOC	EPA 1992d
	1,260	VOC	Kinnear 1992
	3,330 to 6,660	VOC	OTA 1988
Old car scrappage	3,150	VOC or NOx	Kinnear 1992
	3,320	VOC + NOx	CARB 1993b
	4,250 to 6,730	VOC + NOx + CO	SRI 1994
Gross emitter repair	180	CO	Bishop et al. 1993
Federal Phase 2 gasoline	6,700 to 9,600	VOC + NOx	NPC 1993
	9,190	VOC	Kinnear 1992
	51,860	VOC + NOx	SRI 1991
CA Phase 2 gasoline	8,600	VOC	NPC 1993
	77,790	VOC + NOx	SRI 1991
Oxygenated fuels	-640 to -10	CO	Fraas et al. 1990
Federal Tier 1 standards	2,430	VOC or NOx	Kinnear 1992
	5,080	VOC + NOx + CO	SRI 1994
Federal Tier 2 standards	11,550	VOC + NOx + CO	SRI 1994
	12,160	VOC or NOx	Kinnear 1992
TLEVs	180	VOC + NOx	CARB 1993a
	4,220 to 16,440	VOC	Anderson et al. 1994
	7,820	VOC + NOx + CO	SRI 1994
LEVs	470	VOC + NOx	CARB 1993a
	9,830	VOC + NOx + CO	SRI 1994
	13,970 to 55,210	VOC	Anderson et al. 1994
ULEVs	5,660	VOC + NOx	CARB 1993a
	7,960	VOC + NOx + CO	SRI 1994
	79,410 to 186,410	VOC	Anderson et al. 1994

TABLE C.1 (Cont.)

Control Measure	Cost ($/ton 1989 dollars)	Pollutant	Source
The LEV program	8,610 to 37,830	VOC	
	23,850	VOC + NOx	DRI/McGraw Hill 1994
M85 FFVs	2,400 to 14,410	VOC + NOx + CO	Wang et al. 1993
	3,170 to 27,860	VOC	Fraas et al. 1990
	63,410	VOC	Krupnick et al. 1990
M100 FFVs	2,780 to 15,370	VOC + NOx + CO	Wang et al. 1993
M85 dedicated vehicles	670 to 2,400	VOC + NOx + CO	Wang et al. 1993
	11,530 to 29,790	VOC	Krupnick et al. 1990
M100 dedicated vehicles	-3,750 to 7,490	VOC	Fraas et al. 1990
	960 to 3,340	VOC + NOx + CO	Wang et al. 1993
	12,570	NOx	CARB 1993b
	25,940 to 49,000	VOC	Krupnick et al. 1990
Dual-fuel CNG vehicles	0 to 1,920	VOC + NOx + CO	Wang et al. 1993
Dedicated CNG vehicles	< 0	VOC + NOx + CO	Wang et al. 1993
	-3,650 to 160	VOC	Fraas et al. 1990
	2,420	NOx	CARB 1993b
	16,230	VOC + NOx + CO	SRI 1994
EVs	480 to 11,530	VOC + NOx + CO	Wang et al. 1993
	54,490	VOC + NOx + CO	SRI 1994
	209,330	VOC + NOx	DRI/McGraw Hill 1994

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