Title
Estimating the Air Pollution Costs of Transportation Modes

Author
Small, Kenneth A.

Publication Date
1975-10-01

ESTIMATING THE AIR POLLUTION COSTS
OF TRANSPORTATION MODES

by

Kenneth A. Small

October 1975

Working Paper No. 261

The work in this paper was supported by National Science Foundation grant no. GI-37181. The results and views are the independent products of university research and are not necessarily concurred in by the National Science Foundation.

ESTIMATING THE AIR POLLUTION COSTS

OF TRANSPORTATION MODES

by Kenneth A. Small[1]

Individuals concerned with transport planning policy must use a good deal of judgement to weigh the relatively tangible costs of the various modes against a number of other considerations not easily quantified in money terms: air pollution, land use, noise, and architectural aesthetics, to mention a few. To the extent that these intangibles can be quantified in the more rigorous framework of benefit-cost analysis, the uncertainty and disagreement surrounding planners' decisions may be reduced.

It is the goal of this paper to provide some rough and aggregate measures of the economic costs imposed on society by air pollution from various transportation modes in urban areas. While recognizing that costs vary with many local factors which have not been taken into account, I believe the resulting numbers will be useful to those who are responsible for decisions which inevitably must weigh air pollution against money costs. Furthermore, such rough measures are precisely the inputs needed to design crude, aggregate policy instruments such as are often discussed by transport planners and researchers.

---

[1]Doctoral Candidate, Department of Economics, University of California, Berkeley. The work in this paper was supported by National Science Foundation grant No. GI-37181; the results and views expressed are those of the author and are not necessarily concurred in by the National Science Foundation. The author is grateful to Theodore Keeler for both initial inspiration and continuing interest in the research.

The approach is to select with some care the best existing evidence on air pollution costs, skimpy though it is, and from it derive measures specifically for transportation. Throughout, I attempt to err on the side of low costs, so that the resulting numbers are lower bounds rather than best estimates.

I. THEORETICAL ISSUES

General Approach. There are three possible ways to estimate the social cost of air pollution from a given source. First, one can attempt to measure the damage[1] for each of a number of mutually exclusive categories, and add them. In principle this can be done for each pollutant, but important categories may be omitted for lack of information.

Second, property values can be used as a measure of the capitalized value of expected damage from pollution over a given piece of property. The main drawback of this method is the difficulty of isolating the effects of pollution from those of all the other determinants of property values.

Third, rather than measuring the damage, one might simply accept society's judgement as to the optimal equilibrium as measured in some set of legal limits of ambient air standards. Under this approach, the social cost of pollution is the cost of achieving the desired standards. This approach is appealing if the democratic process is superior to economists' calculations for the resolution of a complex social question such as the optimal amount of air pollution; furthermore, within its scope lies a great deal of useful research into efficient methods of attaining given air standards. However, if we are to evaluate the validity of such standards, this approach would be circular.

---

[1]The terms "cost" and "damage" are used interchangeably.

Previous efforts to quantify pollution costs have used the first and second methods. Ridker (1967) reported the first econometric estimates of the relationship between property values and air pollution, and his results have been more or less verified by others in different cities.[1] Ridker also attempted to measure damage by specific categories, but was hampered by an almost complete lack of required data at the time. More recently, Barrett and Waddell (1973) of the U.S. Environmental Protection Agency used the category approach to derive a widely-quoted lower-bound estimate of $16.1 billion for U.S. air pollution damage. They drew on reasonably good information for health and materials damage, and added as a separate category the aggregate welfare loss indicated by Ridker's property value information. The California Air Resources Board (1974) has also used the category approach (with internationally more liberal estimates, but without property value declines) to estimate total U.S. damage at 30 to 74 billion dollars.

It is the present author's judgement that econometric problems make the property value studies simply too unreliable, at their present state of development, to be used as a primary source for a pollution cost estimate. They are likely to be biased by the high correlation between air pollution and important unmeasured variables such as noise, proximity to factories, residential density, and general appearance of the neighborhood. Furthermore, empirical studies have been restricted to residential property, thereby missing the large potential costs involved in the interaction between people and air pollution in industrial and commercial areas.

---

[1] For example, Anderson and Crocker (1971), Zerbe (1969), Peckham (1970). See Anderson and Crocker (1972) for a summary of results up to that time.

This paper therefore presents estimates by categories of damage. Effort is concentrated on two categories, health and materials, which appear quantitatively significant and for which reasonable damage estimates can be obtained. Property value declines are not added because of double counting.[1] The basic sources of information are those used in other studies, particularly Barrett and Waddell, but the interpretation of these sources differs in several important ways discussed below.

In subsequent sections, I first derive estimates of aggregate U.S. damage by all pollutants combined. Emissions inventories and measures of relative severity are then used to disaggregate by specific pollutants. Further disaggregation by geographical location is made possible by applying meteorological data. Finally, information on emission rates from various types of vehicles is presented in order to impute costs to these sources.

Linearity of Damage Functions. It is often presumed that the economic costs of air pollution are a non-linear function of pollution levels, with rising marginal costs.[2] If this were the case, it would greatly complicate our task, for not only would the interpretation and allocation of total damage estimates become more complex, but the final results would have to be disaggregated along one more dimension: namely, prevailing air quality.

---

[1] Barrett and Waddell (1973, pp. 26, 65) recognize that there is overlap between these measures, but argue that it is small because decline in property values is primarily due to aesthetic loss and soiling (which then do not measure elsewhere) and to physical damage to the home and surrounding plants (only a small part of which is included in their materials damage estimate). However, health costs are included elsewhere, and there is every reason to expect that people include their own evaluation of health costs in making residential choices; granted they may lack good information, but this might cause them to either over- or under-value these costs.

[2] For example, Wilson (1974) p. 46.

William Ahern (1974) has argued at length that the function for human health damage _is_ non-linear, and bases some rather strong policy conclusions on that fact. However, a careful reading shows that his results are due to non-linearity _assumptions_ built into the analysis of the effects on healthy individuals. Specifically, he assumes that damage per hour exposed is zero below a certain threshold. This is about as non-linear a response function as one could assume, and it is this assumed curvature which determines the curvature of the resulting total damage functions.[1]

Such a threshold function might seem justified by the Clean Air Act requirement that the Environmental Protection Agency establish some ambient standard, for each pollutant, to "protect the public health." However, one should not confuse a legal convenience with a scientific demonstration of a threshold for damage. In fact, the ambient standards were generally set at the lowest levels at which adverse health effects had been reported. But this merely reflects the state of the art in pollution research. Note for example the following statements by E.P.A. officials:

> At ozone concentrations as low as the existing primary standard (160 mg/m^3), animals exposed for only 3 hours exhibited increased susceptibility to experimentally induced bacterial infections, and a firm linear dose-response relationship was established for this effect....As more human and animal data...are acquired, we may be faced with a pollutant...for which no effects-threshold exists.[2]

---

[1]Shown in Ahern (1974), Figure 8-7, p. 199.

[2]Carl M. Shy, M.D., Acting Director, Human Studies Laboratory; in "Commentary on the Report 'Air Pollution and Public Health,'" in U.S. Senate Committee on Public Works, Subcommittee on Air and Water Pollution, _The Impact of Auto Emission Standards_ (October 1973), Chapter 2, Appendix B, pp. 56-57.

> I think, in general, what we found is that as we do more research on the health effects of the various pollutants with which we deal, we find health effects at ever lower levels of those pollutants in the ambient air.[1]

A perusal of the summaries in the Criteria documents[2] on which the E.P.A. standards were based, gives a general impression that health effects of gradually increasing severity occur over an extremely wide range of pollution concentrations. The naive observation that damage must rise to very high values, as toxic levels are approached, is balanced by the fact that those levels are many times greater than the level at which more moderate effects occur.

As for health effects on people with chronic and acute conditions, Ahern gives evidence obtained by expert opinion, namely estimates by a panel of three medical researchers of the number of "equivalent days of restricted activity" caused by various concentrations of carbon monoxide and oxidants. When these estimates are plotted, we see that for each pollutant, one panelist estimates increasing marginal damage, one decreasing, and one virtually zero throughout the range considered.[3] We must conclude that expert opinion is divided.

Lacking evidence to the contrary, then, it will be assumed in this paper that damage is a linear function of pollution concentration. There is one additional scrap of evidence, again relating to human health, which supports this procedure. Lave and Seskin, in their regressions of total

---

[1] William D. Ruckelshaus, E.P.A. Administrator, testimony before the Senate Subcommittee on Air and Water Pollution, as quoted in The Impact of Auto Emissions Standards, op. cit., p. 39.

[2] U.S.N.A.P.C.A. (1969a, b, 1970a, b, c); U.S.E.P.A. (1971).

[3] Interestingly, the magnitudes by all three were so low, that these damages were virtually negligible, upon aggregation, compared to damages to healthy individuals (who of course are more numerous). See Ahern (1974), Table 8-12, p. 198.

mortality to be described below, tried several specifications which were non-linear in the pollution variables.[1] In every case the hypothesis of linearity could not be rejected, though the power of these tests was not very great. If anything, these alternative specifications tended to suggest mildly decreasing marginal damage.[2]

Valuation of Sickness and Death. Benefit-cost analysis is based on the principle that the gain to society of any change is measured by the sum of the citizens' willingness to pay for that change. Here we are dealing with policies which change air pollution levels and thereby cause changes in the probabilities of various illnesses, and of death. As Mishan (1971) has pointed out, it is therefore the individual's own valuation of changes in probabilities that is relevant.

Unfortunately, empirical measures of this quantity are virtually nonexistent. One is therefore obliged to rely on a quite different, but presumably related, quantity consisting of direct medical expenditures plus foregone earnings. This is not entirely unreasonable, particularly if non-market productive activity is accounted for; it might be argued that a person would pay up to his total future lifetime earnings, less a minimal subsistence living allowance, to avoid certain death. Nevertheless, it is clearly a gross approximation, and is generally believed by those who use it to be an underestimate.[3]

Despite some debate among economists, it seems clear that the appropriate measure of earnings is gross, not net, of consumption. To

---

[1] Lave and Seskin (1973a), Chapter 3, pp. 12-19.

[2] Ibid., Table 3A, regressions no. 4, 6, & 7.

[3] Rice (1966), p. 112; Lave and Seskin (1970), p. 730.

use the latter would be to attach no value to the enjoyment a person receives from his consumption, clearly contradicting the tenets of benefit-cost analysis. As Fein puts it:

> It is true that man consumes partly in order to maintain himself, and in this sense some of his consumption may be considered as a gross investment to take care of depreciation; it is also true, however, that consumption is an end in itself and can be viewed as a final, rather than an intermediate, step in the creation of...products. The question involved concerns the purposes for which an economy exists... the individual enjoys life [enabled by his income], and it is for this purpose that the social economy exists.[1]

Allocation of Total Damage. In order to allocate a total estimate of air pollution costs to specific contributing pollutants, we must know where and in what quantities they are emitted, and the relative severities of each. The following model of nationwide pollution damage, for a given category, is adopted. All emissions within urban areas are dispersed uniformly, and cause damage in proportion to pollutant-specific "severity factors." Emissions outside urban areas cause no damage. Once costs are allocated to pollutants by this very simple model, we will examine the consequences of relating it to account for local differences.

The severity factors must be estimated from whatever skimpy evidence is available for the particular damage category in question. For materials, direct information on specific pollutant-material interactions is used. For human health, it is assumed that the severity of a pollutant is inversely proportional to its federal primary (health-related) ambient air quality standard. These standards were based on the exhaustive reviews of health effects reported in the Criteria documents referred to above.[2]

---

[1]Fein (1958), pp. 18-19, as quoted (and concurred in) by Rice (1966), p. 87.

[2]The standards are promulgated in the Federal Register 36, Part II, 8186 (April 30, 1971), P. 84.

There are a number of problems, however, in putting the health standards on a common basis. First, the averaging period to be used differs from one pollutant to another. What is worse, two of the standards refer to specific chemicals or classes of chemicals (nitrogen dioxide, reactive hydrocarbons) which are only components of the broader classes (nitrogen oxides, all hydrocarbons) for which auto emissions data and standards are given. Worse still, one of the pollutants (oxidants) is not emitted directly, but is rather the product of a complex photochemical reaction involving primarily nitrogen oxides and reactive hydrocarbons;[1] in fact, hydrocarbons are deemed by the E.P.A. to have no deleterious effects at all except through this reaction.

Clearly, any reduction of these standards to a single set of numbers, applicable to the five classes of pollutants identified in emissions data, must be a compromise and only an approximation to reality. Not surprisingly, various authors attempting to do this have come to different answers.[2] The studies by Caretto and Sawyer and by Walther tend to give large weight to automotive pollutants, and allocate something like 40% of total U.S. pollution damage to transportation. They have been persuasively criticized, however, for poor handling of the second and third problems mentioned in the previous paragraph, by Babcock and Nagda (1972), whose own calculations give far less weight to hydrocarbons and allocate only about 20% of total damage to transportation. Their criticism persuades

---

[1]The extreme complexity of this reaction is the subject of a great deal of research; see Calvert (1973). The simplest approach, used by Walther (1972) and by some of the State Implementation Plans for meeting the Clean Air Act, is to assume oxidant levels are proportional to hydrocarbon emissions; but Calvert (p. 61) claims that "significant ozone levels may continue to plague many urban areas even though a near total removal of the reactive hydrocarbons might be effected."

[2]See Babcock (1970); Caretto and Sawyer (1972); Walther (1972); Cumulative Regulatory Effects...(RECAT, 1972), pp. 26-28; Babcock and Nagda (1972, 73).

not only the present author but Walther as well (in a reply following their 1972 letter), so their method will be used here.

II. NATIONWIDE POLLUTION DAMAGE: HEALTH

Lave and Seskin have done by far the most thorough research on quantifying the relationship between air pollution and human health.[1] In their most widely known article (1970), they survey the health effects literature, mostly of an epidemiological nature, and rework some of the original data themselves, to estimate the increased morbidity and mortality for a variety of diseases. Using breakdowns by disease category of 1963 economic costs in Rice (1966), they arrive at an estimate that a reduction in urban pollution levels by one-half would have saved $2.08 billion. This figure has been widely quoted, and used as a starting point by several other studies.[2]

While this estimate is a careful one, it is limited to diseases caused primarily by sulfur dioxide and particulates. Yet since other pollutants could not be controlled for in the regressions, it is not clear to what extent their effects have been omitted. It thus seems unwarranted to <u>add</u> damage from other pollutants to the Lave and Seskin result, as is done in three of the studies just mentioned.

Fortunately another option, the one chosen by Lave and Seskin in their later writings, is available. They performed cross-section regressions of total mortality rates in 117 U.S. Statistical Metropolitan Areas, using as explanatory variables a variety of carefully-chosen socio-economic

---

[1]Lave and Seskin (1970, 1973a, b); Lave (1972).

[2]Barrett and Waddell (1973); California Air Resources Board (1974); Babcock and Nagda (1973); <u>Cumulative Regulatory Effects</u>...(RECAT, 1972).

characteristics, plus pollution levels (as measured by sulfur oxides and particulates). Here it is clear that the sulfur oxide and particulate measures are together serving as a proxy for all pollutants combined.

The results of their regression on 1960 data were:[1]

$$\begin{aligned} MR = 19.607 &+ \underset{(2.53)}{0.041\,P} + \underset{(3.18)}{0.071\,S} \\ &+ \underset{(1.67)}{0.001\,PD} + \underset{(5.81)}{0.041\,NW} + \underset{(18.94)}{0.687\,OLD} \end{aligned}$$

where

MR = total mortality rate per 10,000 population: sample mean = 91.26

P = suspended particulates, annual arithmetic mean (mg/m^3): sample mean = 118.14

S = total sulfates, mimimum reading for a biweekly period (mg/m^3 x 10): sample mean 47.24

PD = population density (persons/$mi.^2$)

NW = % nonwhites in population (x 10)

OLD = % population of age 65 or greater (x 10)

The regression's R^2 was .827, and t-statistics are given in parentheses.

If we take these sample means to represent a typical metropolitan area, and calculate from the equation the effects of a fifty percent reduction in pollution levels, we find mortality is reduced by 4.5%.[2] An identical regression performed on 1969 data yields a corresponding figure of 7.0%.[3] It is not unreasonable, then, to follow Lave and Seskins conservative procedure of using the lower of these figures to apply to early

---

[1]Lave and Seskin (1973b), p. 287, regression no. 2-1. Also reported in (1973a), Table 3A and Lave (1972), p. 31.

[2]This is not to be confused with the figure quoted at the very end of their article in Science (1970), the derivation of which is unclear.

[3](1973a), Chapter 10, p. 11.

1960's pollution levels, and assuming that morbidity would be reduced by the same proportion as mortality.[1]

However, Lave and Seskin make one error in interpreting the result. Since the 4.5% figure is an elasticity derived from the mean pollution levels in the 117 SMSA's of their sample, and the relationship estimated is linear, it is erroneous to extrapolate the same percentage change back to pollution levels in rural areas. The appropriate conclusion is that a reduction by one-half of air pollution in those 117 metropolitan areas would reduce by at least 4.5% the incidence of disease and death in those same areas.

As for benefits from a percentage reduction of pollution levels elsewhere, we cannot say without knowing what those levels are, so some assumption must be made. It is assumed here that Lave and Seskin's sample mean was representative of pollution levels in all "urbanized areas" of the U.S., as defined by the Bureau of the Census; and that 55 percent of all US disease and death costs in 1963 occured in these areas, since that was the fraction of the US population living in them.[2] Pollution levels outside urbanized areas are assumed to be zero. A reduction by one-half of all pollution levels would decrease total US health costs by 2.5 percent.

Clearly, Lave and Seskin's regressions are subject to some of the same econometric problems as property value studies: in particular, correlation of air pollution with unmeasured socio-economic variables. However, they have taken great pains to devise ingenious tests which might show the relation to be spurious or seriously overestimated, all with

---

[1](1973a), Chapter 10, p. 12; Lave (1972), p. 33.

[2]"An urbanized area comprises at least one city of 50,000...plus contiguous, closely settled areas." Statistical Abstract of the United States, 1973, p. 17.

negative results.[1] One of these involved using the same independent variables to predict specific diseases expected to be related, and certain other phenomena (suicide, venereal disease, and crime) expected to be unrelated, to air pollution; and indeed the relationship with pollution was positive where expected and zero elsewhere. What is more, an analysis of the probable statistical biases shows that, on most counts, their numbers are likely to be under-estimates (1973b, pp. 285-286).

All in all, it is difficult after reading their work to believe the measured relation is not a real one. Let us therefore accept the 2.5% figure derived above as the best lower-bound estimate which still includes the effects of all pollutants. Using the linearity assumption to extrapolate to a zero-pollution level, we reach the following summary conclusion: The best lower-bound estimate of the 1963 health costs of urban air pollution in the U.S. is 5.0% of the total U.S. costs of disease and death in 1963. Applying Rice's figure discussed below, this amount is $4.21 billion.

The result of the very careful study by Rice (1966), discounting future earnings at 6%, is $84.19 billion, which is broken down in Table 1.

Table 1

Total Economic Costs of Disease and Death: U.S. 1963

Category	Cost ($ billion)
Direct expenditures	22.53
Lost earnings: morbidity	21.04
Lost earnings: mortality	40.62
TOTAL	84.19
TOTAL ESTIMATED DUE TO AIR POLLUTION	4.21

Source: Rice (1966), p. 110; and text.

[1] See all their post-1970 references, especially 1973a.

Since three-fourths of this estimate is lost earnings, it is worth noting some details in Rice's computation of this factor. The starting point was mean annual earnings of full-time workers, by sex and age category, from Census figures. These were then adjusted upward on fringe benefits, and downward by age- and sex-specific labor force participation rates for 1956 (a year of essentially full employment by U.S. standards). Probably the most important adjustment was for the services of unemployed housewives, who comprised 53% of the female population over fourteen years of age. These were valued conservatively at the mean earnings of a domestic servant, or \$2,670; this may be compared with mean earnings for all working females (\$4,027), and for working males (\$6,949).

It is also worth examining how these earnings figures become translated into present value of lost earnings due to death. The discount rate is fairly important: Use of a 4% instead of a 6% rate would have raised the aggregate total by about \$10 billion. Thus the 6% rate gives a more conservative figure; in case the reader feels an even higher discount rate would be warranted, however, note Rice's comment (p. 87) that these two rates (4% and 6%) correspond to real social discount rates of 6% and 8%, respectively, adjusted downward for the approximately 1.75% annual growth in real productivity which a worker may expect to occur based on historical trends.[1]

A look at the age distribution of present value of lifetime earnings at 6% confirms one's expectation that the early-to-middle age groups are weighted most heavily. For males, the distribution rises from

---

[1]A worker in 1963 expects his real income after t years to be $(1+p)^t E(t)$, where $p = .0175$ is the expected productivity growth rate, and $E(t)$ are the earnings calculated by Rice. Thus if the discount rate is r, the effective discount rate r^1 which should be applied to Rice's data is given by $E(t)/(1+r^1)^t = (1+p)^t E(t)/(1+r)^t$. Thus $1+r^1 = (1+r)/(1+p)$; or $r^1 = 6.14\%$ for $r = 8\%$.

approximately $31,000 at birth to $100,000 for ages 25-29, then falls gradually to $14,000 for ages 65-69. When weighted by number of deaths, however, the maximum comes for ages 55-59 (both sexes combined). The weighted mean (i.e. total discounted lost earnings divided by total deaths) is $22,405.

III. NATIONWIDE POLLUTION DAMAGE: MATERIALS

The estimation of total U.S. cost of deterioration of materials due to air pollution is less complex. The source used here is Salmon (1970) of the Midwest Research Institute. The report examines fifty-three types of materials which, in all probability, include nearly all the materials damage. For each material, the total in-place value (including labor cost for installation) is estimated, as well as the fraction exposed to air pollution. Information from a variety of sources is then used to estimate the increased rate of deterioration resulting from air pollution exposure. The report does not state the dates to which the numbers apply; the year 1965 is assumed here to allow for a five-year lag between basic data and publication.

I have added a separate estimate by Salvin[1] for textile dyes, the one important category omitted from the MRI study. For some categories more elaborate studies have been done subsequent to the MRI study, but given the level of accuracy attainable in this paper, it seems preferable to retain the consistency behind the MRI results for all materials.

The results are given in the first two columns of Table 2. Of the $4.00 billion total, nearly half is accounted for by paint and by zinc (in the form of galvanized steel and alloys). Wherever possible,

---

[1] Victor S. Salvin, University of North Carolina, Greensboro, as reported in Barrett & Waddell (1973), pp. 23-25.

TABLE 2

Materials Damage: U.S., 1965

Material	1965 Economic Damage ($ billion)	Allocation to pollutants where possible		
		NO_x	OX	SO_x
Paint	1.20	.40	.40	.40
Zinc	.78			.78
Cement & Concrete	.32			
Nickel	.26			.26
Dyes	.20	.12	.08	
Rubber (natural & Syn.)	.20		.20	
Cotton	.15			.15
Tin	.14			.14
Aluminum	.11			.11
Copper	.11			.11
Wool	.10			
Steel	.06			.06
Nylon	.04			
Cellulose ester	.03			
Building brick	.02			
Urea & Melamine	.02			
Paper	.02			.02
Leather	.02			.02
All other	.22			
TOTAL	4.00			
Total Allocable	3.25	.52	.68	2.05
% Allocation		16%	21%	63%

Key: NO_x = Nitrogen Oxides
OX = Oxidants
SO_x = Sulfur Oxides

Source: MRI (1970), Table XIII, p. 53; and text.

the qualitative ratings of resistance to corrosion given in the MRI report[1] have been translated (by the present author's judgement) into allocations of damage to specific pollutants.

The MRI figures are based on corrosion to materials actually in place, and do not include the cost of more expensive substitutes for corrosion-prone materials: for example, using aluminum in place of galvanized steel. Separate studies for two materials, which include such substitutions, result in damage estimates much higher than the corresponding categories in the MRI study: five times as high in the case of galvanized steel,[2] and twice as high in the case of rubber products.[3] This suggests that the materials damaged may be badly underestimated, but by how much it is impossible to say.

The MRI report also estimates damage from soiling, but this has not been included here. The estimate is based on "what it would cost if all [buildings and materials] were kept clean" (p. 58), which is clearly an overstatement of the economic cost; and indeed, their $100 billion estimate is so large that it would overwhelm all other categories of air pollution considered in this paper. Soiling costs have proved extremely difficult to measure, and the safest course is to omit them entirely, though recognizing their potential significance.

IV. NATIONWIDE POLLUTION DAMAGE: OMITTED CATEGORIES

Only for health and materials damage, discussed in the previous two sections, is the data deemed sufficiently reliable to include a

[1] Table IV and V, pp. 25-27. Since the ratings for particulates include soiling damage, which we reject (see below), no allocation to particulates is made, although they do cause some chemical corrosion.

[2] Haynie (unpublished).

[3] Battelle-Columbus Laboratories (1970).

quantative estimate of air pollution costs. Thus the estimates in this paper are specifically restricted to those two categories, and any use of the figures derived below should take into account not only their basic conservatism with respect to methodology, but also whatever subjective appraisals of other damage categories seem relevant to the problem at hand. In this section some guidelines as to likely orders of magnitude of other categories of damage are discussed.

Air pollution damage to agriculture is really an exception to the above paragraph: it is omitted not because the data is poor, but because estimates are so small in comparison to health and materials, that the effort to review them carefully is simply not warranted. Two estimates may be cited here, both relying on prior work: \$0.12 billion by Barrett and Waddell (1973), and \$0.09 billion by California Air Resources Board (1974).

Given the location of most air pollution, one might suspect that damage to urban ornamental vegetation is considerably greater. Yet the only known study estimated an even smaller value than for agriculture,[1] although it cannot be considered definitive since it is essentially measuring aesthetic valuation.

Considerable effort has gone into measuring the damage caused by soiling due to particulate pollution, but with uniformly poor results. Ridker (1967) had moderate success at establishing a relationship between household cleaning costs and a pollution index in the cases of an isolated severe episode (a boiler failure in Syracuse, N.Y.); but none for the

---

[1]By the Stanford Research Institute, reported in California Air Resources Board (1974), p. 56.

everyday pollution of interest to us. Other studies, reported in Barrett and Waddell (1973), have either failed to find a reliable relationship, or else have found very small damage for a restricted set of possible recipients.

Finally, there are a few categories for which there seems no possibility of a quantitative estimate: aesthetic loss, damage to wildlife, and possible long-term ecological damage. In principle, property values might give us a clue to people's willingness-to-pay for aesthetic amenities, but only if studies included some measure of loss of view; and this would probably show so little variation within a single metropolitan area that its effect could not be reliably estimated. As for damage to wildlife, any estimate based on commercial valuation must surely be small, which leaves only such concerns as extinction of endangered species or general decline in wildlife populations. This kind of concern is similar to that for potential long-term ecological effects, which, being at best poorly understood and at worst not even suspected, are poor candidates for quantification. It is this kind of concern, particularly, which must be weighed without the benefit of dollar estimates in the formation of policy.

V. ALLOCATION TO POLLUTANTS

The major issues involved in allocating total U.S. damage to specific urban pollutant emissions were discussed in the first section. Details and results are presented in the following paragraph and in Table 3.

For materials damage, the entire $4 billion is allocated according to the same percentages as the $3.25 billion for which known chemical properties enabled the allocation shown in Table 2.

TABLE 3

Allocation of U.S. Damage to Pollutants

	Pollutant					
	CO	HC	NO_x	SO_x	Part.	Total
HEALTH:						
Tolerance factor, 24-hour exposure[a] (mg/m^3)	7800	788	330	373	260	
Severity factor[b] (CO=1)	1	10	24	21	30	
U.S. urban emissions, 1963[c] (million tons)	88	21	11	17	17	
Cost allocation[d] (percent)	6.1	14.7	18.5	25.0	35.7	100.0
U.S. cost, 1963[d] ($ billion)	0.26	0.62	0.78	1.05	1.50	4.21
Cost per urban emission, 1974[e] ($/ton)	6.22	62	149	131	186	
MATERIALS:						
Severity factor, implicit[g] (HC=10)	0	10	49.5	77.0	0	
U.S. urban emissions, 1965[c] (million tons)	92	22	12	18	17	
Cost allocation[f] (percent)	0	10	27	63	0	100
U.S. cost, 1965[f] ($ billion)	0	0.40	1.08	2.52	0	4.00
Cost per urban emission, 1974[e] ($/ton)	0	34	168	262	0	
TOTAL:						
Severity factor, implicit[g] (CO=1)	1	15.5	51.1	63.1	30.0	
Cost per urban emission, 1974 ($/ton)	6.22	96	317	392	186	
Cost per urban emission, 1974 (10^{-4} ¢/gram)[h]	6.85	106	350	432	206	
U.S. Cost, 1970[i] ($ billion)	0.43	1.57	3.38	6.25	2.24	13.86
U.S. Cost, 1974[i] ($ billion)	0.57	2.10	5.25	9.96	2.85	20.74

Key: CO = Carbon Monoxide
HC = Hydrocarbons
NO_x = Nitrogen Oxides
SO_x = Sulfur Oxides
Part. = Particulate matter

Source: see footnotes

Footnotes to Table 3:

[a]Babcock and Nagda (1973), p. 174, Table 1, except the secondary ambient standards are replaced by the primary standards where they differ, namely for SO_x and particulates. See text for further details.

[b]7800 divided by tolerance factor.

[c]Total emissions are interpolated linearly between 1960 and 1968; then multiplied by 0.65, the approximate percentage estimated to occur in "urban areas" in 1969 for all pollutants except particulates, for which no estimate is given. U.S.E.P.A. (1973b), p. 4, Table 1; and pp. 13-20.

[d]Total damage from Table 1 is allocated to each pollutant in proportion to its severity factor times its U.S. urban emissions.

[e]Cost allocation divided by U.S. urban emissions, then inflated to 1974 by current-dollar gross national product per capita.

[f]From Table 2, assuming oxidant damage is allocable equally to NO_x and HC emissions.

[g]Relative severity factors are simply proportional to damage per ton of urban emissions.

[h]One ton = 907,185 grams.

[i]Damage per ton is updated to 1970 or 1974 by current-dollar gross national product per capita, then multiplied by estimated emissions from U.S.E.P.A. (1973b), p. 4, applying as before the factor of 0.65 to get urban emissions. 1970 emissions are given directly; for 1974, we extrapolated the 1968 to 1970 trend.

For health damage, the method of Babcock and Nagda (1973) is modified so as to be based on the federal primary (health) rather than secondary (welfare) ambient air standards. As explained in their 1972 letter, Babcock and Nagda deal with the problems discussed above by assuming that 50% of all nitrogen oxides are nitrogen dioxide (for which the ambient standard is set); and (apparently) by assuming that 98% of all sulfur oxides are sulfur dioxide. They deal with photochemical oxidant formation by assuming a simple model of the chemical reaction,[1] then allocating the standard for the resulting oxidant molecule equally to its two molecular precursors, on the assumption that the reaction is 20-25% complete under average conditions of solar radiation.

Given these estimates of costs per unit of emissions in various years, how can they be extrapolated to a common recent date, say, 1970 or 1974? There are two factors changing: the price level, and the real value of materials and persons exposed to emissions. To inflate by an aggregate measure of the total economy, such as gross national product, would be an overestimate because much of this growth has taken place in the size of urban areas, whereas the cost of a given unit of urban emissions is more closely related to the density of economic activity. Let us assume that geographical size of each urban area grows in proportion to total U.S. population; per-unit emission costs then grow (in current dollar terms) in proportion to current-dollar gross national product per capita. The detailed results for 1974 are shown in Table 3.

It may be useful, for purposes of comparison with other studies, to give our results in terms of total U.S. damage for a given recent year, taking into account urban emissions in that year. This is done for 1970

---

[1] One molecule NO_x (molecular weight 46) plus one molecule HC (weight 16) form one molecule oxidant (weight 48). See (1972), p. 728.

and 1974 in the last two lines of Table 3, with the resulting total damage estimates of $13.9 and $20.7 billion, respectively.

VI. GEOGRAPHICAL DISAGGREGATION

To this point, we have followed the simple model of uniformly dispersed emissions within all urban areas. This seemed sufficient for allocating total damage to pollutants, given the approximations involved anyway. However, variations in local conditions are so significant that some account of them needs to be taken.

First, there is local variation in the amount of atmospheric dispersion, which of course is what determines the ambient air concentrations resulting from a given amount of emissions. To deal with this kind of variation between urban areas, I have elaborated an innovation by Cook and Helms (1973). For a given urban area, the nationwide average cost per unit of urban emissions is multiplied by a "meteorological correction factor," which is proportional to that urban area's average frequency of days of high meteorological potential for air pollution. Such days are known as "episode-days" and are defined by criteria on wind speed, vertical mixing height, and precipitation set by the National Air Pollution Potential Forecasting Program.[1] Note that they have nothing to do with actual pollution emissions.

Using data collected by the National Weather Service,[2] and population fiuures from Statistical Abstract of the U.S.,[3] we find that the population-weighted average frequency of episode-days, for the 33 largest

---

[1]An episode is two or more consequetive days with no significant precipitation, mixing heights less than 1,500 meters, and wind speeds less than four meters per second. Holzworth (1972), p. 21.

[2]Holzworth (1972), Figure 58, p. 83.

[3](1972), pp. 19-20.

TABLE 4

METEOROLOGICAL CORRECTION FACTOR
SELECTED CITIES

Metropolitan Area	Criterion: Official (1500 m 4 m/sec)	Criterion: Less Stringent (2000 m 6 m/sec)
Northeast:		
Boston	.27	.28
New York	.12	.31
Washington D.C.	.53	.62
Mid-West:		
Chicago	.35	.56
St. Louis	.41	.59
Minneapolis	.04	.38
Denver	.56	.54
South:		
Atlanta	.53	.80
New Orleans	.95	1.4
Dallas	.00	.42
West:		
Seattle	.90	1.3
Portland	2.3	1.5
San Francisco	2.8	2.3
Los Angeles	3.4	3.2
San Diego	7.7	3.4
Pop-Weighted Average:		
Continental U.S.	1.00	1.00
California	3.8	2.9
Other	.36	.55

Source: Holzworth (1972), Figures 58, 62; and text.

U.S. metropolitan areas, is 14.7 per year. The ratio of the frequency for individual cities to this average is given for selected cities in the middle column of Table 4.

It is obvious from Table 4 why California has led the rest of the nation in air pollution control efforts. Removing its six from the list of 33 metropolitan areas, the weighted average for the rest of the U.S. drops to 5.3 episode-days per year, while California's average is 55.4: a factor of ten difference!

This meteorological correction factor is not wholly satisfactory, because it implicitly relies on a model which conflicts with our basic linearity assumption of pollution damage: that only when conditions reach a particular degree of severity is there pollution damage. This criticism would be less forceful if that degree of severity were more like a typical moderately-bad day. For this reason the correction factors are recalculated, in the last column of Table 4, using a less stringent definition of "episode-day."[1] For the most part, the two sets of factors are similar enough to warrant reasonable confidence in using them. By the less stringent definition, which is adopted henceforth, California differs from the rest of the U.S. by a factor of 6 instead of 10.

To summarize, it is assumed that California cities have unit emissions costs of 2.9 times the values shown in Table 3, and that the rest of the U.S. has costs of 0.55 times those values.

Another kind of local variation to be considered is in the density of economic activity. It is clear that, other things equal, a ton of

---

[1]Mixing heights less than 2,000 meters, winds less than 6 meters per second. Data from Holzworth (1972), Figure 62, p. 87. By this definition the U.S. average was 70.8 episode-days per year.

carbon monoxide emitted in Manhatten is more damaging than one in the suburbs of Syracuse, New York.

One proxy for density of economic activity is population density. A better one is "net residential density," defined as the population per unit area of residential land; this measure more accurately captures the very high densities of central business districts, where population density might be quite low. According to Meyer, Kain, and Wohl,[1] net residential densities vary from over 200 (thousand people per square mile) in downtown Chicago to around 25 in its suburbs; for Detroit the range is from 60 to 10, and for Pittsburgh from 55 to 10. Calculations for the San Francisco Bay Area[2] give values of 36 for the city of San Francisco, 16 for Oakland, and 8-12 for suburban cities.

It seems reasonable from these figures to expect pollution costs per unit emission to vary by a factor of at least four between most central cities and their suburbs. Of course, where the cities are small this is somewhat modified by dispersion of the pollutants. The problem is sufficiently complex to discourage further refinements here.

There are many other sources of variation in cost per urban emission. Two which may be mentioned are time-of-day and seasonal variation. These are of real importance in detailed planning for a metropolitan area, but cannot be taken account of here.

VII. COSTS OF TRANSPORTATION VEHICLE EMISSIONS

Using the costs per unit of urban emissions calculated thus far, it is straightforward to compute the costs of any source of known emissions located in an urban area.

---

[1] (1965), p. 207.

[2] Calculated from Bay Area Transportation Study Commission, Bay Area Transportation Report (1969), pp. 43-44.

Emissions from transportation vehicles is a field of study with its own extensive literature, and one fraught with complexities. To name a few: Measured emissions vary with type of road, average speed, type of car, air temperature, engine temperature, driver habits, and maasurement device. Estimates of emissions in the future are even more hazardous. Nevertheless, the Environmental Protection Agency has reviewed current information and issued a set of factors which it considers the best available estimates, and which are to be used by the states in estimating the effects of alternative strategies for meeting ambient air regulations. In general, the factors rely on field tests, and thus reflect actual emissions as opposed to legal standards. Details and sources are given in Table 5.

Application of the damage estimates from Table 3 shows that, in both uncontrolled and post-1977 autos, about half the damage comes from hydrocarbons. As is well known, nitrogen oxides and carbon monoxide are the other major contributors, except that for post-1977 autos fully one-fourth of the costs are due to sulfur oxides and particulates. The dramatic exhaust reductions mandated for these models shift the burden to lesser-controlled emissions (including evaporative hydrocarbons). In the 1974 models, on the other hand, nitrogen oxides are a major component, reflecting the fact that NO_x standards are phased in later than CO and HC standards; in fact, NO_x emissions were <u>increased</u> by the measures used to reduce CO and HC emissions in the models of the late 1960's and early 1970's.

So far, I have ignored air pollution costs from the manufacture or generation of inputs to transportation services, particularly electric power (an input to rail transit services) and oil refining (an input

TABLE 5

AIR POLLUTION EMISSIONS AND COSTS

Vehicle Type	Emissions[a] (gm/mile) CO	HC^c	HC^d	NO_x	SO_x	Part.[e]	1974 Cost[b] (¢/mile) U.S.	Calif.	U.S. Excl. Calif.
AUTO[f]									
Pre-1961 model (in year 1974)	95	8.9	6.6	3.3	.13	.54	.36	1.05	.20
1969 model (in year 1974)	68	5.8	2.5	5.1	.13	.54	.33	.96	.18
(in year 1974, with NO_x device)[g]	68	5.8	2.5	3.0	.13	.54	.26	.74	.14
1974 model (new)	37	3.2	1.76	3.1	.13	.25	.20	.57	.11
(5 years old)	47	4.7	1.76	4.1	.13	.25	.25	.74	.14
1974 composite[h]	60	5.6	2.4	3.9	.13	.47	.28	.81	.15
Post-1977 model (new)[i]	2.8	.27	1.76	.24	.13	.25	.04	.12	.02
(5 years old)	4.2	.54	1.76	.73	.13	.25	.06	.18	.03
1995 composite[j]	3.9	.48	1.76	.66	.13	.25	.06	.17	.03
MOTORCYCLE[k] (4-stroke engine)									
Pre-1973 model	33	2.9	.96	.24	.022	.046	.07	.21	.04
DIESEL BUS[l] (50 seats)									
Pre-1973 model	21.3	4.0	0	21.5	2.8	1.3	.96	2.78	.53
DIESEL LOCOMOTIVE[m]									
With one car (80 seats)	41	93	0	219	35	16	10.52	30.51	5.79
With four cars (320 seats)	96	216	0	508	83	36	24.46	70.94	13.45
	(Lbs per landing-plus-takeoff cycle)						($ per landing-plus-takeoff cycle)		
AIRCRAFT[n]									
Boeing 747 (250 seats)	187	49	0	126	7.3	5.2	24.82	71.98	13.65
Boeing 727 (150 seats)	51	15	0	31	3.0	1.2	6.49	18.83	3.57

Footnotes to Table 5:

[a]Emissions assume low altitudes and urban arterial driving at average speed of 19.6 miles per hour.

[b]Costs per unit of emissions in U.S. urban areas in 1974 are from Table 3. These are multiplied by 2.9 on California, and by 0.55 for other U.S. (see text). For other years, inflate or deflate by current-dollar gross national product per capita.

[c]Exhaust emissions.

[d]Crankcase and evaporative emissions. In addition, evaporation at filling stations during transfer to autos may amount to 11.67 pounds per thousand gallons, or 0.35 gm/mile for an auto which gets 15 miles per gallon. Loading of underground fuel tanks may contribute up to the same amount, depending on type of loading. See U.S.E.P.A. (1973a), pp. 4.4-6 and 4.4-7. Filling station evaporation is omitted here due to changing control requirements.

[e]Includes particulates from tire wear. Reduction for post-1972 vehicles assumes use of non-leaded fuel.

[f]Emission factors are derived from measurements on in-use autos, using the 1975 federal test procedure (Federal Register, July 2, 1971), a combined cold- and hot-start, constant volume sample procedure. Deterioration with age beyond calendar year 1972 is estimated separately for each pollutant and model year. Source: U.S.E.P.A. (1975), Appendix D, Tables 1-3, 1-13, 1-19, 1-21, 1-24, 1-25.

[g]California required retrofitting of its 1966-70 model year vehicles with devices to reduce NO_x emissions by 42% or more. This began in 1974 but was repealed in early 1975; probably less than half the affected vehicles had the devices installed before repeal.

[h]Composite the same tables as in footnote f, plus Table 1-22, giving exhaust emissions. These are calculated from the age distribution of U.S. autos in 1974.

[i]Assuming enforcement of the last reductions called for in the 1970 Clean Air Act Amendments, originally scheduled for 1975 models and subsequently postponed to 1978 models. U.S.E.P.A. (1975, Appendix D, Table 1-25) estimates that available technology would also reduce crankcase and evaporative emissions to 0.5 gm/mile, but this reduction is not in present regulations so is excluded here.

[j]Composite exhaust emissions are calculated assuming a steady-state population of post-1977 model cars, with age distribution and estimated deterioration from U.S.E.P.A. (1975), Appendix D, Tables 1-20 and 1-22.

[k]U.S.E.P.A. (1973a), p. 3.1.7-2.

[l]U.S.E.P.A. (1975), Table 3.1.5-1.

[m]Assumes a two-stroke non-turbocharged engine, 100 ton locomotive and 80 ton cars, fuel consumption .00764 gallons per gross-ton-mile (from 1970 Southern Pacific records). This is dirtier than some engines presently in use, and new engines have additional pollution control devices. Emissions factors from U.S.E.P.A. (1973a), pp. 3.2.2-1 and 3.2.2-2.

[n]U.S.E.P.A. (1973a), pp. 3.2.1-2 and 3.2.1-4.

to all other transport modes). One can show by the following rough calculations that such costs may safely be neglected in comparison with the costs of direct vehicular emissions of the kind in Table 5.

Air pollution data are taken from a recent U.S. Council on Environmental Quality report on energy systems.[1] Costs are calculated using the unit damage estimates in Table 3 above, and assuming the worst possible case in terms of location and pollution control: All power stations and refineries supplying an urban transportation system are assumed to be located within the urban area and to operate without any of the newer pollution controls which are gradually being phased in. Electric power is assumed to be produced by a composite of hydroelectric, coal, oil, gas, and nuclear power plants reflecting the nationwide distribution.[2] The oil refinery is that assumed by the CEQ report; its total pollution is allocated to gasoline and other outputs in proportion to their respective volumes.

The results are shown in Table 6. While the magnitudes are not insignificant compared to the prices of electricity and gasoline, they are nevertheless a small fraction of the environmental cost of the direct air emissions of an uncontrolled auto. For example, the pollution cost of electricity generated for use by the BART rapid rail system[3] is only .04 cents per seat-mile; the pollution cost of refining gasoline for use by autos[4] is .05 cents per auto-mile. Given that these costs are calculated under worst conditions, it seems safe to neglect them here.

---

[1](1973), pp. 40-58.

[2]In the case of oil-burning power plants, pollution from the refining process is included.

[3]Using 4.5 KWH per car-mile, 80 seats per car.

[4]Assuming 15 miles per gallon fuel economy.

TABLE 6

AIR POLLUTION DAMAGE FROM ELECTRIC POWER GENERATION AND OIL REFINING

	Emissions (tons)					COST
	CO	HC	NO_x	SO_x	Part.	
Electric power plant[a]	10933	2047	14990	59353	85378	$44.2 mill.
Cost per kwh						0.74¢
Oil Refinery[b]	71290	11027	1108	3290	419	$3.2 mill.
Cost per gal gasoline						0.72¢

Footnotes:

[a]1000 megawatt pl nt running at load factor 0.75 for one year, thus producing 5.99 billion kwh after transmission losses of 8.8%

[b]That portion of pollution attributable to production of 450 million gallons of output.

Source: U.S. Council on Environmental Quality (1973), Figure 7 and Tables A-2 (ftn. 16), A-5 (ftns. 14, 16), and A-8 (ftn 16).

VIII. CONCLUSIONS AND SAMPLE POLICY ANALYSES

It has been emphasized that the value of the estimates made here is that they reveal minimum orders of magnitudes. Are the large numbers large or small compared to other components of transportation costs? Should environmental considerations be given much weight in transportation planning?

One's general impression is that the numbers are small but not negligible. For 1974 and earlier model autos in California, the approximate range is 0.5 to 1.0 cent per vehicle-mile, a cost which is of borderline significance in comparison with privately perceived costs, but which is of the same order of magnitude as current user charges in the form of gasoline taxes. Charging pollution taxes of this magnitude would probably not greatly affect the total amount of automobile travel, which suggests that the known evidence on health and materials damage does not justify extreme measures to reduce overall use of automobiles. On the other hand, one-half cent per vehicle-mile is not at all insignificant in comparison with pollution control costs, suggesting that some controls probably can be so justified; we shall examine this more closely below. It should also be noted that the expected variation within an urban area can make our estimates negligible in some areas and highly significant in others; it is certainly plausible to argue, for example, that damage of at least several cents per mile is caused in high-density central business districts by automobiles in slow-moving congested traffic.

We now turn to some examples of the kind of analysis which can profitably be carried out with these pollution damage estimates.

Example: Clean Air Act automotive pollution controls.

Are the nationwide auto controls mandated in the 1970 Clean Air Act Amendments worth the cost? To answer this question, let us find the approximate increase in the purchase price of a new car which could be jsutified, from our estimates, by a 90% reduction in pollution emissions from their levels in a five-year-old 1969 model car.

The reduction is worth 0.30 cents per mile, according to Table 5. Assuming that a car is driven 10,000 miles per year, the savings is $30. per year over the car's lifetime. With a lifetime of ten years, this savings, discounted at six percent back to the time of purchase, is worth $221. This is of the same order of magnitude as the expected cost of the control equipment, leaving the answer ambiguous. It is probably safe to say that we cannot justify placing controls on all cars in order to reduce emissions from those driving in urban areas; it is also safe to say that such controls are clearly justified in California cities.

There are of course many other considerations. The catalytic converters used to achieve these reductions have been accused of imposing a consumption penalty, breaking down easily, and emitting dangerous metallic particulates and sulfuric acid mist. On the other hand, cheaper means of reaching the pollution standards may be just in the offing. There is no pretense that the conclusions offered here are final.

Example: California's retrofit requirement for nitrogen oxides.

For years a battle raged in the California legislature over a requirement that all 1966-1970 model cars be fitted retroactively with a device to reduce nitrogen oxide emissions by no less than 42 percent, at a cost of no more than $35. per car. The law was passed and remained in force long enough to require some of these cars to purchase the devices, but was then repealed. Should it have remained in effect?

We can apply the same approach as in the last example. A typical car to which the law applied would, according to Table 5, have its pollution damage reduced by 0.22 cents per mile of urban driving. A car driven in an urban area, with a remaining life of five years at 10,000 miles per year, would then save $22. per year, justifying (at a 6% discount rate) an initial expenditure of $93. Thus the program appears clearly worth the cost.

It must be cautioned that this result relies heavily on the procedure used to allocate damage from oxidants to their photochemical precursors, including nitrogen oxides. Other considerations affecting the decision are problems of enforcement, and alleged damage to engines operating at high speeds.

Example: Support of public transit.

Can massive public subsidies to urban bus systems be justified on the basis of air pollution?

Suppose that a typical urban area, with automotive pollution costs in 1974 of 0.28 cents per mile, can induce some commuters to switch to full 50-passenger buses (with pollution damage of .02 cents per passenger-mile) by offering a new subsidized service. On air pollution grounds alone, one can justify a subsidy of 0.26 cents per passenger-mile, or 13 cents per bus-mile. This must be regarded as rather small: Typical variable costs of express peak-hour bus service are on the order of $1.00 per bus-mile. Thus it is difficult, from the evidence gathered in this paper, to justify bus subsidies as a major approach to reducing air pollution.

Conclusion. This paper has attempted to bring together the best current information relating certain specific components of air pollution damage to money costs. This information has been applied to the problem of air pollution from transportation vehicles, primarily automobiles. Insofar as the openly conservative estimates are of the correct order of magnitude, they suggest that for typical urban areas, pollution costs do not justify large reductions in automobile travel, but do justify significant expenditures on air pollution control. Due to wide variation in urban density and driving conditions within an urban area, either of these conclusions may be reversed for local subregions.

It must be emphasized that these conclusions apply only to those categories of environmental damage considered, namely health and materials damage from air pollution. The transportation planner will have to use his own judgement as to the weight to be given to toxic lead accumulation, asbestos particles from brakes, aesthetics of smog, oil spills or water pollution from gasoline production, visual aesthetics of transportation facilities, possible long-term ecological damage, and many other factors. It is hoped that the evidence presented in this paper, by moving a few of the planner's considerations into a more objective realm, can facilitate this process.

BIBLIOGRAPHY

Ahern, William R., Jr. (1974), "Health Effects of Automotive Air Pollution," in Henry D. Jacoby, John D. Steinbruner, et.al., Clearing the Air: Federal Policy on Automotive Emissions Control (Cambridge, Mass. Ballinger).

Anderson, Robert J., Jr., and Crocker, Thomas D. (1972), "Air Pollution and Property Values: A Reply," Review of Economics and Statistics 54, pp. 470-473.

__________, __________ (1971), "Air Pollution and Residential Property Values," Urban Studies 8, pp. 171-180.

Babcock, Lyndon R., Jr. (1970), "A Combined Pollution Index for Measurement of Total Air Pollution," Journal of the Air Pollution Control Association 20, pp. 653-659.

Babcock, Lyndon R., Jr., and Nagda, Niren L. (1972), "Response: Rating of Pollutants by Effect," Journal of the Air Pollution Control Association 22, pp. 727-728.

__________, __________ (1973), "Cost Effectiveness of Emission Control," Journal of the Air Pollution Control Association 23, pp. 173-179.

Barrett, Larry B., and Waddell, Thomas E. (1973), "Cost of Air Pollution Damage: A Status Report," U.S. Environmental Protection Agency Publication No. AP-85.

Battelle-Columbus Laboratories (1970), "A Survey and Economic Assessment of the Effects of Air Pollution on Elastometers," prepared for U.S. National Air Pollution Control Administration.

California Air Resources Board (1974), "Air Pollution Damage: A Survey of Recent Studies," preliminary draft.

Calvert, Jack G. (1973), "Interactions of Air Pollutants," in National Academy of Sciences, Proceedings of the Conference on Health Effects of Air Pollutants, prepared for the U.S. Senate Committee on Public Works (Washington: U.S.G.P.O.).

Caretto, L.S., and Sawyer, R.F. (1972), "The Assignment of Responsibility for Air Pollution," presented at the Annual Meeting of the Society of Automotive Engineers, Detroit, Michigan.

Cook, Scot . . and Helms, L. Jay (1973), "Automotive Emissions Taxes: Some Theoretical, Empirical, and Methodological Results," prepared for the Twelfth Annual Meeting, Western Regional Science Association, Monterey, California.

Cumulative Regulatory Effects on the Cost of Automotive Transportation - RECAT (1972), Final Report of the Ad Hoc Committee, prepared for the U.S. Office of Science and Technology (Washington: U.S.G.P.O.).

Fein, Rashi (1958), Economics of Mental Illness (New York: Basic Books).

Haynie, F.H. (unpublished), "Estimation of Cost of Air Pollution as the Result of Corrosion of Galvanized Steel," U.S. Environmental Protection Agency, Research Triange Park, N.C.

Holzworth, George C. (1972), "Mixing Heights, Wind Speeds, and Potential for Urban Air Pollution Throughout the Contiguous United States," U.S. Environmental Protection Agency Publication No. AP-101.

Lave, Lester B. (1972), "The Economic Costs of Air Pollution," in Frank C. Emerson, ed., The Economics of Environmental Problems, Michigan Business Papers No. 58 (Ann Arbor: University of Michigan Graduate School of Business Administration).

Lave, Lester B., and Seskin, Eugene P. (1970), "Air Pollution and Human Health," Science 169 (August 21), pp. 723-733.

________, ________ (1973a), "Air Pollution and Human Health,: Part I of Seskin, The Social Costs of Air Pollution, Ph.D. Dissertation, Carnegie-Mellon University.

________, ________ (1973b), "An Analysis of the Association Between U.S. Mortality and Air Pollution," Journal of the American Statistical Association 68, pp. 284-290.

Meyer, John R.; Kain, John F.; and Wohl, Martin (1965), The Urban Transportation Problem (Cambridge, Mass.: Harvard University Press).

Midwest Research Institute (1970), "Systems Analysis of the Effects of Air Pollution on Materials," prepared for U.S. National Air Pollution Control Administration, revised edition.

Mishan, E.J. (1971), "Evaluation of Life and Limb: A Theoretical Approach," Journal of Political Economy 79, pp. 687-705.

Peckham, Brian W. (1970), "Air Pollution and Residential Property Values in Philadelphia," U.S. National Air Pollution Control Administration.

Rice, Dorothy P. (1966), "Estimating the Cost of Illness," U.S. Public Health Service, Health Economics Series No. 6 (P.H.S. Publication No. 947-6).

Ridker, Ronald G. (1967), Economic Costs of Air Pollution: Studies in Measurement (Praeger).

U.S. Council on Environmental Quality (1973), "Energy and the Environment: Electric Power."

U.S. Environmental Protection Agency (1971), "Air Quality Criteria for Nitrogen Oxides," Publication No. AP-84.

U.S. Environmental Protection Agency (1973a), "Compilation of Air Pollutant Emission Factors," 2nd edition, as revised by Supplements No. 1 and 2, dated 1973.

U.S. Environmental Protection Agency (1973b), "Nationwide Air Pollutant Emission Trends 1940-1970," Publication No. AP-115.

U.S. Environmental Protection Agency (1975), "Supplement No. 5 to Compilation of Air Pollutant Emission Factors," draft version.

U.S. National Air Pollution Control Administration (1969a), "Air Quality Criteria for Photochemical Oxidants," Publication No. AP-63.

U.S. National Air Pollution Control Administration (1969b), "Air Quality Criteria for Sulfur Oxides," Publication No. AP-50.

U.S. National Air Pollution Control Administration (1970a), "Air Quality Criteria for Carbon Monoxide," Publication No. AP-62.

U.S. National Air Pollution Control Administration (1970b), "Air Quality Criteria for Hydrocarbons," Publication No. AP-64.

U.S. National Air Pollution Control Administration (1970c), "Air Quality Criteria for Particulate Matter," Publication No. AP-49.

Walther, Eric G. (1972), "A Rating of the Major Air Pollutants and Their Sources by Effect," Journal of the Air Pollution Control Association 22, pp. 352-355.

Wilson, D.G. (1974), "Estimates of Pollution from U.S. Nonfreight Highway Transportation," International Journal of Environmental Studies 6, pp. 35-50.

Zerbe, R.O., Jr. (1969), "The Economics of Air Pollution: A Cost-Benefit Appr ch," Ontario Department of Public Health, Toronto.