IV. A FRAMEWORK FOR REFORM

One potential criticism of any proposed reform is that it will be overwhelmed by administrative difficulties. It is clear that setting separate standards wherever marginal benefits vary would be a political and administrative nightmare, despite its theoretical efficiency advantages. Uncertainties in the links among emissions, pollutant concentrations, and pollution damages would make the calculation of precise benefits highly uncertain and would be, no doubt, the source of endless litigation.

Introducing benefit-based flexibility, however, need not entail significant increases in the amount of information collected by EPA, nor in the efforts devoted to enforcement. Indeed, a great deal could be accomplished with the information and analyses already gathered by EPA, and in at least some cases benefit-based flexibility should reduce rather than exacerbate enforcement problems.

In this section, we describe a framework for incorporating benefit-based flexibility into environmental regulation. The key element in our proposal is a limited number of differential standards based on differences in the marginal benefits of control. Our goal is not to devise an "optimal" scheme, but rather to examine what could be accomplished with relatively modest changes in the existing system. We begin with a very basic approach, explained with the aid of a simple example. We then turn to the question of how many different classes should be established and how they should be defined. In Section IV, we examine some potential complications that may, at least in theory, call for somewhat more complicated strategies.

The Basic Plan

Consider how EPA might go about setting benefit-based standards for a category of sources emitting some hazardous substance (e.g., benzene from maleic anhydride plants or chromium in the water effluent from leather tanning plants). For simplicity, we assume that the damages are restricted to health effects, that exposure levels at any given site are proportional to emissions, and that risk is proportional to exposure. The basic system consists of four steps: (1) estimating the marginal costs of reducing emissions for several control options; (2) defining exposure classes and assigning individual sources; (3) estimating the marginal cost of exposure reduction for each combination of exposure class and control option; and (4) selecting the standard for each class.

Step 1. The first task is to conduct engineering studies, probably using a "model plant," to determine control options and to estimate their costs. EPA already prepares such estimates, though usually for only a very few alternatives. It would be desirable to have the analysis include a larger number of options, however, ranging from no control to a total ban. (As numerous critics have pointed out, even with its current uniform standards the agency should consider more control levels.) The marginal cost of controlling emissions could then be estimated for each control level. With a discrete number of options, the marginal cost for each option would be simply its additional cost (compared to the next most stringent control level) divided by its incremental reduction in emissions.

Step 2. The next step is to specify the exposure classes and to assign each source to a class. Exposure classes should be based upon differences in each source's exposure factor (population exposure per unit of emissions), which for an air pollutant might be measured in part-per-billion person years per kilogram of the substance controlled (ppb-person-years/kg). We discuss below in some detail alternative methods of defining the classes and of assigning sources. The key issues include whether the classes are standardized or unique for each regulation; how many classes are used; and whether assignments are based on source-specific data and modeling or on cruder criteria, such as location. Whatever the method of assignment, the end result of this step would be the grouping of sources into a limited number of classes, with the sources within each class having similar, though not identical, marginal benefits of controlling emissions.

Step 3. The results of the first two steps may be used to calculate the marginal cost-effectiveness of each control option for each exposure class. For example, if the marginal cost of 90 percent control is $1/kg, and the average exposure factor is 0.5 ppb-person-years/kg for a particular class, then the cost-effectiveness ratio for that combination is ($1/kg)/(.5 ppb-person-years/kg) = $2/ppb-person-year. The various options then maybe arrayed in increasing order of marginal cost per unit of exposure reduction.

Step 4. The final step is to decide what level of control will be required for each exposure class. The ranked list from Step 3 will give the cost-effective combinations, but the final choice will require a judgment as to the value of reducing exposure. This step is likely to be a difficult one, because of major uncertainties about the risks posed by the substance and disagreement about how much society should be willing to spend to reduce risk. These same difficulties also arise, however, in setting uniform standards.

Once the standard has been set for each class, enforcement would proceed in the same manner that it does now. Monitoring the compliance of individual sources should be no more complicated than with uniform standards. Indeed, to the extent that benefit-based flexibility led to exempting some low-damage sources from any controls, enforcement would be easier because fewer sources would need to be monitored.

An example. A simple example helps to clarify the process we propose. Suppose that there are 30 plants, each of which emits on average 1000 kg of the substance per year. Three control options have been identified: 50, 80, and 95 percent control. Table 2 presents the cost and emission-reduction estimates for the "model plant." It also shows the marginal cost of controlling emissions for each option.

Table 2. Control Options for Model Plant

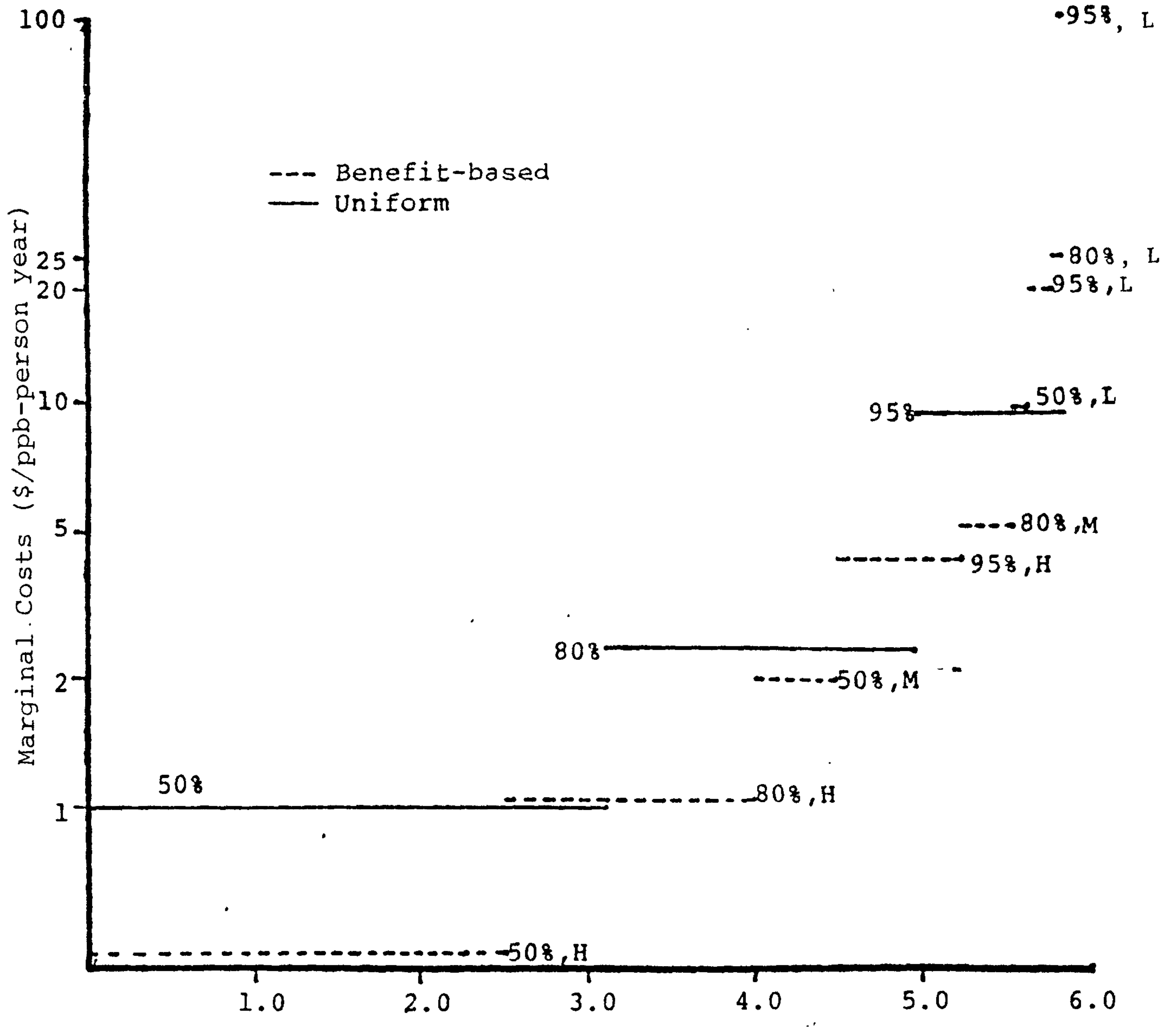
	Control Level (%)		
	50	80	95
Marginal Cost ($1000)	100	150	300
Marginal Emissions Reduction (1000 kg)	500	300	150
Marginal Cost-Effectiveness ($/kg)	0.2	0.5	2.0

The 30 plants are divided into three classes based on exposure factors (measured in our hypothetical case in "part-per-billion-person-years/kg"). For simplicity, the number of plants in each class is the same. The first column in Table 3 shows the average exposure factor for each class and for the plants as a whole. The other columns report the marginal costs per unit of exposure reduction. The most cost-effective option is 50 percent control of the "high-exposure" plants, with a ratio of $0.40/ppb-person-year, followed by 80 percent control of those same plants, with a marginal cost of $1/ppb-person-year. Note that while the marginal cost of reducing emissions is 10 times higher for 95 percent control than for 50 percent, it is substantially more cost-effective in terms of exposure to impose the tightest controls on the high-exposure plants than to require any controls on the plants in the low-exposure class. The final row of Table 3 shows the marginal cost-effectiveness ratios if controls are imposed uniformly on all plants.

The marginal costs of reducing exposure are plotted in Figure 4; the dashed lines represent the flexible strategy, the solid lines the uniform one. Note that the optimal reduction in total exposure depends on both the marginal benefit and on the strategy. If, for example, the marginal benefit is $1.50/ppb-person-year, the optimal uniform standard is 50 percent, which yields a total reduction in exposure of 3.1 million ppb-person-years. With the flexible strategy, the optimum is 80 percent control for the high-exposure plants, with no controls on the others, which yields a reduction of 4.0 million ppb-person-years. If the marginal benefit is $15, however, the optimal uniform

Table 3. Cost-Effectiveness Ratios for Exposure Classes

Exposure Class	Average Exposure Factor (ppb-years/kg)	Marginal Cost of Reducing Exposure ($/ppb-person-year)		
		50	80	95
High	0.5	0.40	1.00	4.00
Medium	0.1	2.00	5.00	20.00
Low	0.02	10.00	25.00	100.00
Average	0.21	0.97	2.40	9.68



Cumulative Exposure Reduction
(10^6 ppb-person-years)

Figure 4. Marginal Costs of Reducing Exposure with Benefit-Based and Uniform Standards

standard is 95 percent control, for a decrease in total exposure of 5.89 million ppb-person-years, while the optimal benefit-based flexible strategy is 95 percent for the high-exposure plants, 80 percent for the medium plants, and 50 percent for the plants in the low-exposure class, yielding an overall reduction of 5.65 million ppb-person-years.

Figure 5 plots the total costs of the two strategies as functions of the reduction in total exposure. As expected, at every point the cost is lower with the benefit-based strategy. The difference is particularly pronounced at intermediate levels of control; at low levels, costs are relatively small under both strategies, while at high levels most of the options must be exercised, leaving little room for cost savings under the benefit-based strategy.

Defining and Assigning Exposure Classes

The design of exposure classes must represent a compromise between the ease of administration that comes with standardization and simplicity, and the increased efficiency that comes with greater refinement and the tailoring of exposure classes to particular circumstances. The tradeoffs fall along a continuum. Uniform national standards are one extreme; there is but a single class (the nation) and it is the same for all regulations. At the other extreme lie source-specific standards, with exposure classes uniquely set for each source category. The uniform approach, as we have shown, sacrifices a great deal of efficiency. The opposite extreme, however, while fully efficient in theory, is clearly unworkable in practice. Fortunately, we

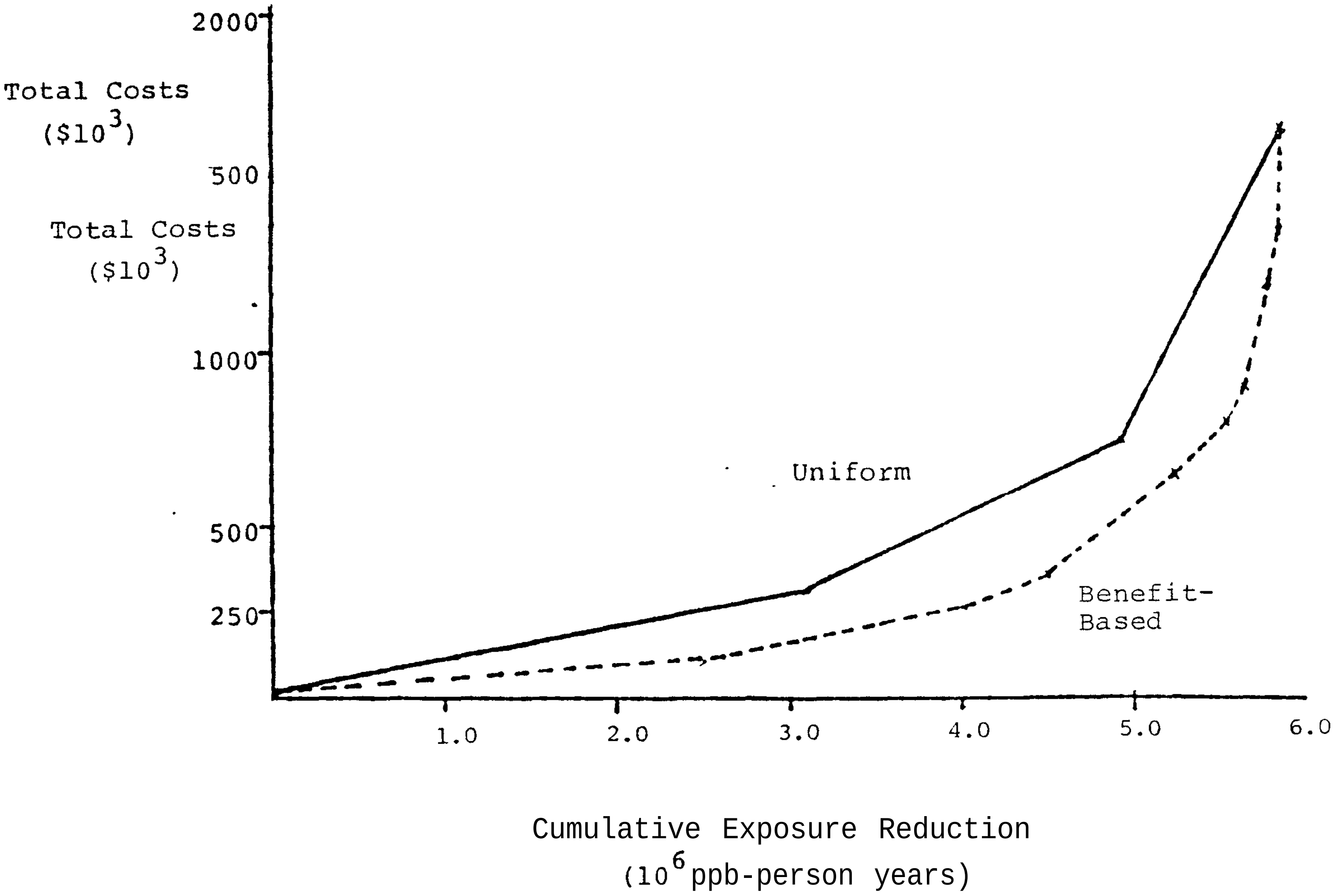


Figure 5. Total Costs of Reducing Exposure with Benefit-Based and Uniform Standards

believe that it is possible to achieve most of the potential efficiency gains with a relatively simple, easy-to-administer system.

Determining Exposure Factors for Sources. A variety of methods could be used to estimate exposure factors for sources to determine the class in which they fall. For large plants, source-specific dispersion modeling and population data could be used. EPA already uses general dispersion modeling and plant-specific population data to estimate the benefits of many of its regulations. The use of local meteorological data and various plant-specific parameters (e.g., stack height) would make the assignments more accurate. It would also encourage damage-mitigation strategies other than emission control and relocation. For example, a plant located in a densely populated area is likely to cause less exposure if it has a "tall stack" that disperses emissions widely. Conversely, a similar plant located upwind of a large city might reduce exposures by using a short stack that led to less long-distance transport. Basing exposure class assignments on source-specific dispersion modeling would encourage consideration of such options.

For smaller, more numerous sources, cruder techniques could be used. Generalized modeling could be done for such categories, with individual assignments based on location. All automobiles in the Boston SMSA, for example, might be assigned the same exposure factor. More crudely yet, all automobiles in SMSAs with populations in excess of 2 million might be assigned the same exposure factor. The payoff to more accurate determination of exposure factors for individual sources depends in large part on how many separate exposure classes will be used for regulatory purposes; if the number of classes is small, with each class covering a wide range of exposure factors, fine accuracy will make little difference except in borderline cases.

Number of Exposure Classes. As the number of classes grows, the sources within each class become more homogeneous and the standards may be tuned more finely to specific circumstances. At the extreme, each source would be in a separate class and, at least in theory, full allocational efficiency could be achieved. Source-specific standards, however, obviously are impracticable; with every source treated individually, EPA would find it impossible to maintain the appearance of objectivity, and it might well face a large number of individual suits.

The central question is how much efficiency is gained as the number of classes increases As noted earlier, the empirical studies suggest that large gains may be reaped with only a few classes; most of the studies simply divided sources into two classes. That finding, however, may simply reflect the paucity of the data employed, in particular the very limited numbers of control options for which cost estimates were available. If the classes can be custom tailored for each regulation, there are no efficiency gains from increasing the number of classes beyond the number of control options.

We can gain some additional understanding of the importance of refining exposure categories with the aid of a simple numerical example. Suppose there are many sources (for ease of calculation, we shall assume an infinite number), each with the same cost function for controlling emissions:

$$C(r) = .5r^2 \quad , \tag{3}$$

where r is the level of emissions reduction ($0 \leq r \leq 1$). Sources differ, however, in their exposure factors, so the benefits of reducing emissions vary. For simplicity, let the marginal benefit of controlling exposure be 1, so the benefit per kilogram of reducing emissions by the factor r at a source with exposure factor E is rE.

Consider first a uniform emission-reduction standard, $\bar{r}$, for all plants. The net benefit from such a standard is:

$$N = \int_0^\infty (\bar{r}E - .5\bar{r}^2)f(E)dE \ , \tag{4}$$

where f(E) is the distribution of exposure factors. Again for simplicity, let f(E) be uniform over the interval $0 \leq E \leq 1.$ 1. The net benefit is then given by:

$$N = \int_0^1 (\bar{r}E - .5\bar{r}^2)dE \tag{5}$$

$$= .5\bar{r} - .5\bar{r}^2 \quad .$$

Net benefits are maximized when, $\bar{r}^* = .5$, at which point N = 0.125.

Now consider the opposite extreme, source-specific standards, each optimized to that source's exposure factor. The optimal standard for a plant with exposure factor E is simply x* = E. The average net benefit per source under perfect benefit-based flexibility is then:

$$N^* = \int_0^1 (E^2 - .5E^2)dE \qquad (6)$$

$$= 0.1667 ,$$

a 33 percent improvement over the uniform standard.

We also can consider intermediate cases. Table 4 reports the results for several different numbers of categories. Note that a simple two-class approach achieves 75 percent of the maximum gains possible under perfectly discriminating benefit-based flexibility. Each successive level of refinement improves efficiency, but by smaller and smaller increments. Relatively crude benefit-based differentials, including only a small number of classes, do almost as well as highly refined approaches, without imposing a significant administrative burden.

Several factors suggest that our example overstates the gains that would be reaped in most cases by increasing the number of classes. First, we assumed a continuous range of control levels, so that the optimal level of control was different for each exposure factor. In practice, however, the number of options considered is finite, and usually small, so that the same control requirement will be optimal for sources with a range of exposure factors. Second, we assumed an infinite number of sources, though in fact the number is always finite. Finally, we

Table 4. Gains from Refining Exposure Classes

Number of Classes	Net Benefits	Percentage of Maximum Improvement
1	0.1250	0.0
2	0.1563	75.1
3	0.1620	88.8
4	0.1641	93.8
5	0.1650	96.0
10	0.1663	99.0
	0.1667	100.0

assumed that the exposure factors were distributed uniformly. More typically, however, the distribution is denser in the middle, with "thin tails." That means that exposure classes can be defined more narrowly over those ranges where large numbers of sources are clustered, thus placing most sources in relatively homogeneous classes. These considerations reinforce our conclusion that in many cases most of the efficiency gains from benefit-based flexibility can be achieved with just a few classes.

Standardization of Exposure Classes. The optimal number of exposure classes depends in part on whether the classes are defined separately for each source category and substance, or are standardized for a range of source types and pollutants. The optimal exposure-class boundaries are functions of a variety of variable -- including the distribution of sources, the marginal costs of the control options, and the value ascribed to exposure reduction -- that will differ across regulations. If the boundaries can be optimized to take account of these factors in individual cases, only a few exposure classes are needed, as shown above. If a standardized set of boundaries is used, however, more classes may be desirable to permit finer tuning of regulations.

Standardization of exposure classes would offer two attractive features. Predetermined classes would simplify the analysis of individual regulations, eliminating the need to set boundaries on a case-by-case basis. Perhaps more importantly, they would appear more objective, a major potential asset for both political and legal reason. With exposure-class boundaries set separately for each regulation, EPA constantly would face arguments that boundaries should be redrawn slightly because the specific circumstances of an individual source made it imperative that it be subject to a looser (or tighter) standard. With uniform classes, such arguments would be much harder to sustain in the context of individual rulemakings.

For maximum ease of administration, it would be desirable to translate exposure class boundaries (expressed in units of exposure per unit of emissions) into geographic boundaries. The assignment of sources would then be virtually automatic. The difficulty is that the geographic boundaries for a given range of exposure factors will vary with the pollutant and the source type. For short-lived pollutants emitted at ground level, for example, exposure will depend almost entirely on the population density of the immediate surrounding area. For longer-lived pollutants emitted from tall stacks, however, areas far from the plant itself may be affected. Thus, it would probably make most sense to standardize exposure classes for certain classes of pollutants and types of sources, rather than to have a single, uniform system for all. For large sources, particularly those that can take actions that affect their own exposure factors, source-specific modeling might be desirable, though exposure classes still could be standardized in terms of exposure factors, if not geography.

Summary. In implementing benefit-based flexible standards, regulators have a wide range of options for defining classes and assigning individual sources to those classes. Although more research into these issues is needed, our tentative conclusion is that most of the advantages of benefit-based flexibility can be reaped with a relatively crude approach that relies on a small number of standardized classes for most pollutants and source types. More refined strategies may be appropriate in some cases, although the marginal costs of refinement are not negligible and the marginal gains appear to fall off rapidly.

V. EXTENSIONS AND POTENTIAL COMPLICATIONS

The basic procedure outlined above makes several implicit assumptions about the nature of the problem to be regulated: (1) the marginal benefit of reducing exposure is uniform across exposure classes; (2) the marginal costs and benefits of reducing emissions are independent; (3) human health effects are the only damage caused by the pollutant; and (4) at any given site, the marginal benefits of control do not vary over time. Although these assumptions are reasonable approximations for many important environmental problems, including toxic air pollutants, obviously they are inappropriate for many others. In this section, we examine how relaxing those assumptions affects the performance of the scheme we have proposed -- both in absolute terms and relative to uniform standards -- and suggest ways in which the basic plan could be modified to improve its performance.

Variation in Benefit of Exposure Reduction

Our framework assumes that areas with the same exposure factor will reap the same marginal benefit from reducing emissions. This assumption will be inaccurate if individuals in different zones differ in their average sensitivity to the pollutant, if they differ in their valuation of risk reduction, or if the relationship between emissions and risk is nonlinear.

Variation in Sensitivity. Physiological data suggest that individuals vary widely in their sensitivity to pollutants. To the extent that risk factors vary, the marginal value of reducing exposure also will vary. These individual differences, however, are unlikely to result in significant variations in the average sensitivities of different regions, and thus are not of direct concern to our plant. If certain areas were found to contain an unusually high proportion of particularly susceptible individuals, those areas could be placed in a higher class than their population densities would otherwise warrant, though we regard this event as unlikely.

Variation in Valuation. Individuals also vary in the valuation they place on risk reduction (Viscusi, 1978), primarily because of differences in tastes and incomes. Area-wide average valuations will exhibit much less variation, but significant differences may remain because of differences in average incomes. In theory, efficiency would require tighter standards for high-valuation areas which would tend to be those with higher average incomes. We believe, however, that most people would find it repugnant for the Federal government to set more stringent regulations to protect higher-income individuals. One alternative for taking account of local differences in valuation is to transfer authority for environmental regulation to lower levels of government, but that raises a wide range of difficult issues that are beyond the scope of this paper. Uniform standards, of course, also fail to account for variations in the marginal benefits of control due to variations in either valuation or sensitivity.

Nonlinearity. The major reason to expect differences in the marginal benefit of exposure reduction is nonlinearities in the links between emissions and health effects. Our basic plan assumes implicitly that in any given area, exposure is proportional to emissions and risk is proportional to exposure. These appear to be reasonable approximations for many health-threatening pollutants, but not for all. The link between emissions and exposure may be nonlinear because the chemical reactions that occur are a function of the volume of emissions. Ambient concentrations, and hence exposures, for example, may rise much more rapidly with emissions once the assimilative capacity of the air or watershed has been exceeded.[8] The dose-response function relating risk to exposure also may be nonlinear. If the marginal risk rises with exposure -- as most scientists believe with respect to acute effects, and many believe for carcinogens -- the marginal benefits of control will be higher in areas with higher ambient concentrations.[9]

These issues suggest that in some cases it may be desirable to adjust class assignments on the basis of ambient concentrations, placing areas with higher concentrations in classes that will be subject to tighter standards. (This was the basis for Luken et al.'s proposal that tighter water-effluent standards be imposed only where existing controls are insufficient to meet ambient standards.) Such adjustments are likely to reinforce the differences based solely on variations in exposure factors, as densely populated areas also tend to have larger numbers of emission sources, and thus higher ambient concentrations.[10]

To the extent that exposure factors and ambient concentrations are positively correlated, rising marginal damages from exposure makes uniform standards doubly inefficient, as the marginal benefits of controlling emissions are higher in densely populated areas both because there are more people exposed and because the benefit per individual of reducing exposure is larger due to higher ambient concentrations. Although it would be desirable under such circumstances to adjust exposure classes for differences in ambient levels, even a "naive" system that defines classes and sets standards on the assumption of constant marginal damages within each exposure class will be far more efficient than a uniform standard.

Correlation Between Marginal Cost and Benefits

Our basic approach, by using the same model plant for all exposure classes, implicitly assumes that the marginal costs and benefits of emission control are uncorrelated. That is, high-exposure sources have, on average, the same emission control costs as low-exposure sources. (We do not assume that costs are uniform, simply that there is no systematic relationship between control costs and exposure factors.[11]) We see little reason to doubt the accuracy of this assumption in the vast majority of cases. We are unaware, for example, of any evidence suggesting that emission-control costs are different in urban areas than in rural ones.

In a few cases, however, marginal costs and benefits may be correlated significantly. If the correlation is negative (i.e., high-exposure sources tend to have lower costs), the advantages of benefit-based differential standards are reinforced -- the optimum involves even more differentiation than our plan would indicate. If the correlation is positive, however, as noted in Section I, benefit-based differentials may be less efficient than uniform standards. In extreme cases, the optimal standards may be tighter in low-benefit (but also low-cost) areas. Kalt (1982) reports a possible example of such a case in his study of Federal regulations requiring reclamation of strip-mined land. He estimates the costs and benefits of those regulations for each of three regions: Appalachia, Midwest, and West. He finds that the benefit of reclamation per ton of coal is much higher in Appalachia than in the other regions, but the costs are even higher.[12] Thus, he argues that net benefits would be increased by relaxing controls in Appalachia, exactly the opposite recommendation that would flow from focusing solely on interregional variations in benefits.

The strip-mining example sounds a note of caution for our scheme, but we think a relatively unimportant one. That case is unusual in that many of the factors (such as topology and vegetation) that affect the benefits of control also affect its costs. Thus, little efficiency will be sacrificed by adopting a strategy that generally ignores the correlation between marginal costs and benefits, making exceptions only when there is evidence of a strong, systematic relationship.

Non-Health Damages

Our basic plan, by focusing on human exposure, is aimed at controlling health-threatening pollutants. Many environmental regulations, however, are designed primarily to provide non-health benefits -- such as improved visibility, greater recreational opportunities, protection of plant and nonhuman animal life, and preservation of wilderness areas Although exposure-based classes may make little sense in such cases, the basic principles of benefit-based flexibility apply; standards generally should be tighter where the marginal benefits of control are higher.

The central difficulty in applying this basic principle to non-health cases is that there is no single proxy for marginal damages that is as useful as the exposure factor is for health effects. The criteria for designating an area as "high-benefit," and thus qualifying for tight standards, vary widely, depending on the nature of the particular problem. National parks, for example, are likely to be low-benefit areas for health-threatening pollutants, but may be classified as high-benefit for pollutants affecting visibility.[13]

To a greater extent than with health-threatening pollutants, existing policy towards other environmental problems already incorporates some degree of benefit-based flexibility. In part this occurs because many of the decisions are made on a case-by-case basis, so that project-specific benefits (and costs) can be considered. Much of the controversy over off-shore oil drilling in New England, for example, has focused on the highly productive fishing grounds that might be harmed by oil leaks. Siting decisions for major facilities with potential adverse environmental impacts, such as nuclear power plants and oil refineries, routinely consider site-specific factors that affect marginal damages, and grant approval for construction contingent on particular actions to reduce damages.[14] Many such decisions are in the hands of state or local governments, thus providing further opportunities for tailoring requirements to local conditions, rather than imposing uniform national requirements.

Intertemporal Variation in Marginal Benefits

As noted in the introduction, marginal damages may vary across time as well as space. This temporal variation provides additional opportunities for enhancing efficiency through benefit-based flexibility. In some cases, the variation is predictable and daily. Aircraft noise, for example, is far more annoying at night than during the day.[15] Other variations are almost as a regular, but follow a much longer cycle. The damage caused by water effluents, for example, may be much higher in the summer, when flow rates in many rivers are low (Roberts, 1975). In still other cases, the temporal variations are less predictable, though they may have a strong seasonal component. Temperature inversions and low wind speeds, for example, lead to higher ambient concentrations of air pollutants (and thus higher exposure) for any given level of emissions. Such meteorological conditions occur irregularly, though almost always in the summer (Masters, 1974).

In theory, standards should vary over time in response to these changes in marginal benefits. Extending benefit-based flexibility to cover the dimension of time as well as space, however, is likely to be difficult and of limited usefulness in most cases. One problem is that pollution control techniques often are capital intensive, with high fixed costs. Thus, even if marginal benefits of control vary sharply over time, the optimal levels of control (and hence the optimal standards) are unlikely to vary much. If a control device has been optimized for 90 percent control, for example, minimal savings may be reaped by operating it at 50 percent control during low-damage periods, and it may well be impossible to achieve 95 percent control during high-damage periods. The other major problem with time-varying standards is that they may greatly complicate the enforcement process, requiring more frequent monitoring to ensure compliance.

Despite these problems, we believe that there are cases where it would be worthwhile to vary standards over time. Aircraft noise is a clear example; the apparent damage differential between night and day is large, and it is relatively simple, though not costless, to reschedule flights in response to day-night differential standards. Time-differentiated standards also may make sense for large stationary sources where a high fraction of control costs are for operating expenses. Some state regulatory agencies, for example, have negotiated "fuel-switching" agreements with certain sources, primarily power plants, whereby the source must use low-sulfur fuel when meteorological conditions lead to high ambient concentrations, but are allowed to use higher-sulfur fuels at other times (Birdsall, 1981).

VI. DISTRIBUTIONAL ISSUES

Our analysis thus far has focused on the efficiency of benefit-based flexibility, on its ability to increase the aggregate net benefits of environmental regulations. The gains, however, would not be universal; as with virtually any policy change, some individuals and some firms would be worse off than they are under the status quo. Three types of objections to benefit-based flexibility are particularly likely to be raised: (1) it would result in excessive levels of risk for residents of areas where emission standards were made less stringent; (2) it would impose an unfair burden on firms located in high-damage, tight-regulation areas; and (3) it would put those regions with tighter regulations at a competitive disadvantage in attracting and retaining industry.

Distribution of Risk

Switching from uniform to benefit-based standards would tend to decrease emissions and risks in densely populated areas, but raise them in lightly populated regions. Some observers may see this result as evidence of discrimination against residents of rural areas, placing a low value on protecting their health and imposing unfair levels of risk on them. Although perhaps superficially plausible, neither objection stands up to scrutiny; indeed, benefit-based flexibility is likely to be more equitable than uniform standards along both dimensions.

Consider first the implicit values placed on protecting the health of individuals residing in different locations. Suppose we have two plants that have the same costs for each control option, but the exposure factor for one plant is 100 times higher than that for the other. A uniform emission standard places the same marginal value on reducing emissions at the two plants, but it implies that protecting an individual living near the high-exposure plant is worth only 1 percent as much as protecting someone near the low-exposure plant. In contrast, benefit-based differential standards, which seek to equalize the marginal costs of exposure, places the same implicit value on protecting all individuals, regardless of where they live. Thus, if the criterion for equity is that health improvements for different individuals be valued equally, uniform emission-based strategies are distinctly inferior to those that incorporate benefit-based flexibility.

Another possible measure of equity is the extent to which risks are distributed reasonably equally and are not concentrated among a few individuals. This concern is reflected in certain provisions of the Clean Air Act, for example, which have been interpreted as requiring protection of particularly sensitive groups (White, 1981). This criterion is impossible to apply rigorously, however, as there always will be variations in exposure and individual susceptibility, and hence risk, regardless of the regulatory strategy chosen. Moreover, we see no reason to apply it on a source-by-source basis; what matters to individuals is the nature and magnitude of the risks they face, not whether the risks come, say, from benzene emitted by maleic anhydride plants or from particulates emitted by diesel-powered automobiles.

Whatever the merits of the criterion, in most cases benefit-based flexibility would decrease the variability in risk faced by different individuals. Under a uniform emission standard, emissions per unit capacity are constant, but ambient concentrations vary widely depending on the sizes and numbers of sources. As discussed in Section IV, the density of many types of emission sources tends to be correlated with population density, so that under uniform emission standards those who live in densely populated areas typically face higher concentrations and risks. Benefit-based flexibility, by imposing tighter controls on sources in highly populated areas, tends to counteract the impact of a larger number of sources, thus reducing, rather than increasing, the variability in ambient concentrations and individual risks.

Distribution of Control Burdens

In contrast to its effect on risk, benefit-based flexibility probably would lead to greater variance in the distribution of control costs across sources. Otherwise identical sources could face very different control costs depending on their locations. Firms forced to meet tighter requirements undoubtedly would question the fairness of this result, protesting that it placed them at an unfair competitive disadvantage, thus violating the principle of horizontal equity (see Harrison and Portney, 1982).

In those cases in which sources are owned by individuals for their personal use, the horizontal equity argument is less compelling, because the sources do not compete with one another. The argument also is less relevant for firms (e.g., service stations or dry cleaning plants) that compete only in geographically limited markets, because all of the firms that are direct competitors would face the same regulations. It is of little importance to the owner of a dry cleaning establishment in Boston, for example, if dry cleaners in rural Utah, or even Western Massachusetts, face more lenient regulations, and thus can charge lower prices.

The horizontal equity argument is potentially more compelling when geographically dispersed firms compete in the same markets, for then benefit-based flexibility may well affect relative competitive positions. It is important to realize, however, that no form of regulation affects all firms equally. The costs of complying with a uniform standard vary widely, with the result that some firms may sustain loses and a few may go out of business altogether, while others register increased profits as regulation-induced price increases outpace their higher costs (Leone and Jackson, 1978). Moreover, while most new-source standards are uniform, many current standards for existing sources vary widely across the country, depending on the control levels needed to achieve ambient standards. Thus, interfirm differences in the impact of regulation are not unique to benefit-based flexibility.

We also note that many other costs -- including wages, energy prices, and property rents -- vary widely. So far as we know, no one has suggested that government should seek to level these costs in different areas to promote greater equity among competitors. The prices imposed by regulation, either implicitly with standards or explicitly with incentive schemes, should reflect the opportunity costs of the environmental resources consumed, just as, for example, land prices reflect relative scarcity.

Regional Impacts

The most potent objections to benefit-based flexibility, politically if not logically, are likely to be based on concerns about its possible impact on the ability of certain regions to compete for new industry and jobs. The fear will be that with differential regulation, firms would not build new plants in areas with tight regulations, and in some cases would move existing plants to regions with more lenient standards.

Fears that benefit-based flexibility would lead to major regional dislocations are almost certainly overdrawn. As we argued earlier, given the large number of factors that affect location decisions, differences in environmental regulations are unlikely to be the controlling issue in most cases. Moreover, to the extent that location decisions are altered, those areas that lose polluting industries will benefit from lower risks and other environmental improvements. Conversely, regions that attract such industries will bear higher environmental costs than otherwise. The result is likely to be a net gain for both types of regions.[16]

VII. COMBINING BENEFIT- AND COST-BASED FLEXIBILITY

In this paper, we have focused on ways in which benefit-based flexibility could be incorporated into the existing regulatory system based on standards, devoting virtually no attention to economic incentives and other approaches designed to make regulations more sensitive to variations in marginal costs. Our focus does not reflect a belief that cost-based reforms are unimportant, or that they are incompatible with benefit-based flexibility. Full efficiency requires that the regulatory strategy be sensitive to variations in both the marginal costs and the marginal benefits of control. The arguments for cost-based flexibility, however, are well-known (if not widely accepted outside the economics profession), while very little attention has been given to important intersource differences in the marginal benefits of control and their implications for regulation We are also pessimistic about the political prospects for adopting full-fledged economic incentive schemes in the near future.

In this section, we examine how benefit- and cost-based flexibility could be combined. Our goal is not to evaluate the merits of cost-based reforms, but rather to show how they could be modified to incorporate the principles of benefit-based flexibility. We focus first on the two relatively "pure" incentive schemes favored by most economists (including ourselves) -- charges and marketable permits. We then turn to existing cost-based reforms, most of which can be thought of as versions of the marketable permit scheme with severe restrictions on trading.

Charges and Permits

The chief virtue of emission charges and marketable emission permits is that they allocate control efforts in accordance with marginal costs, thus minimizing the total cost of achieving any given level of overall emissions. As shown formally in Section I, however, efficiency requires that control efforts also account for intersource differences in marginal benefits. Incorporating benefit-based flexibility into either charges or permits is straightforward.[17]

Charges. Recall the efficiency condition from equation (2): $C_i'(r_i) = VE_i$. As noted earlier, a uniform emission charge cannot achieve this result. A system of emission charges that vary in proportion to exposure factors (E_i), however, can do so. Under such a scheme, a plant with an exposure factor 100 times that ofanother, for example, would face an emission charge rate 100 times higher.

Source-specific emission charges that are proportional to exposure factors are analytically equivalent to a uniform exposure charge. This is easily seen by rearranging equation (2):

$$C_i'(r_i)/E_i = V \quad \text{, for } i=1,...n. \qquad (7)$$

In this form, the optimality condition is that the marginal cost of reducing exposure (damage) should be the same at every source. If all sources pay the same charge per unit of exposure, each would control (through reductions in emissions or relocation) to the point where the marginal cost of reducing exposure equaled (or exceeded) the charge.

In most cases, of course, exposure levels associated with particular sources cannot be observed directly. An exposure charge could be administered, however, by combining monitoring of emissions with exposure factors estimated from dispersion models and population data, in the same ways that we discussed estimating those factors for differential standards in Section III. Alternatively source-specific emission charges could be based on the same estimated exposure factors. Although the two approaches are equivalent from the perspectives of both economic efficiency and administrative cost, the exposure charge may be more attractive politically, simply because it is uniform, thus highlighting the fact that the same value is being ascribed to protecting different individuals and that all firms are paying the same price for the risks they impose.

The major advantage of an exposure charge over benefit-based flexible standards is that it is responsive to differences in cost. In addition, however, it offers at least two other advantages. First,only a single charge would need to be set for each substance, as opposed to two or more standards for each of several source categories. This would ease administration and also appear more equitable, as all sources would face the same (exposure) charge rate rather than different standards. Second, the exposure charge would supply precisely the right incentive for firms to consider environmental damages in their siting decisions. Differential standards, as noted earlier, also provide an incentive to consider low-damage siting, but that incentive may not be at the appropriate level. Recall the example illustrated in Figure 3, in which a plant is choosing between a high-damage site (A) and a low-damage one (B). Under differential standards, moving to B reduces control costs by the area a+c because the standard there is more lenient. The net social gain, however, must include the change in residual damages, the area b-c. Under the standards, the firm does not consider the change in damages, but it does under the charge because it pays for them; moving from A, where the implicit emission charge rate is MBA, to, B, where it is MB_B, changes total charge payments (after the control level has been adjusted) by b-c.

Permits. Benefit-based flexibility also can be incorporated into marketable permit systems. If the markets for permits are relatively large, with significant intramarket variations in the marginal benefits of controlling emissions, benefit-based flexibility can be achieved with differential trading rates. Citing our earlier example, if plant A's emissions cause 100 times as much damage as those of plant B, emission permits between the two should trade at a 1:100 ratio; that is, if A increased its emissions by 1 unit, it would have to buy enough permits from B so that B reduced its emissions by 100 units. Alternatively, permits can be defined in units of exposure rather than emissions.

If many small markets are established, with trade prohibited across market boundaries and minimal variation in marginal damages across sources in any given market, then benefit-based flexibility merely requires that the number of permits in each market be based on the benefits as well as the costs of control in that region. This approach is likely to be particularly attractive for pollutants that exhibit sharply rising marginal damages as emissions increase, because it provides much firmer assurances than either charges or emission standards that particular ambient concentrations will be achieved.[18]

Incremental Cost-Based Reforms

Several EPA policies in recent years have begun to introduce cost-based flexibility into the regulatory system (del Calvo, 1981). The "offset" policy is designed to accomodate new sources in areas that have not attained ambient air standards. Such sources must employ "best available technology" controls, which still leave some remaining emissions. The firm must then find existing sources that will cut their emissions below the levels allowed by standards, so as to offset the new emissions. In essence, this creates a very limited system of marketable permits, in which existing sources implicitly are given permits based on the standards.

The other major cost-based reform is the "bubble" policy. Many large plants have several sources, each of which emit the same pollutant and each of which is subject to a separate standard. The bubble policy sums those individual standards to arrive at a plant-wide limit on emissions of that pollutant. The plant is then free to relax controls on high-cost sources, so long as it makes compensating reductions in emissions from others. The policy may be thought of as allowing plants to set up internal markets in emission permits. Many firms originally complained, however, that the cumbersome requirements for using the bubble policy greatly limited its usefulness. The EPA has relaxed many of the procedural requirements, although the bubble is still limited to trades within a single plant (American Enterprise Institute, 1981, pp. 23-24). A possible extension of the bubble process would allow nearby plants owned by the same firm to form a single bubble. A more ambitious extension would allow bubbling across plants owned by different firms. This latter change, if adopted, would would come very close to the marketable permit schemes advocated by many economists.

EPA is also considering allowing automobile manufacturers to average the emission rates of the cars they sell (46 Fed. Reg. 43734). Under current law, every car must meet the same limits. Emission averaging would allow manufacturers to sell some cars with higher emissions, so long as they sold enough cars with lower emissions that their sales-weighted average met the standard. The plan is essentially the same as the one currently used for gasoline milage standards. It is also very similar to the bubble policy used for stationary sources.

The key characteristic of each of these plans is that it defines the starting point as existing emission standards and then allows limited trading across sources and, in some instances, across firms. Benefit-based flexibility would be easy to incorporate, as benefit-based differential standards would merely define a new starting point from which trades could proceed. Under both the offset policy and the bubble policy, trades are restricted to sources that are close enough that they almost certainly would fall into the same exposure class under benefit-based flexibility.

Emissions averaging for automobiles poses a mild problem, as current proposals would allow manufacturers to average over all of their sales. Thus, for example, the sale of a low-emission vehicle in a rural area could be averaged with the sale of a high-emission vehicle in a densely populated city, which would not increase emissions but would increase exposure. Trades of that sort clearly would violate the principles of benefit-based flexibility.

At lease two approaches solve the problem. Both start with differential standards for different areas based on exposure classes. The first approach would be to restrict averaging to cars sold in areas with the same exposure class. The second approach would allow firms to average across different exposure classes, but give greater weight to vehicles sold in high-exposure areas, in essentially the same manner that we proposed above for cross-class trades in emission permits. This latter approach would give manufacturers greater flexibility to reduce costs, while still providing incentives for tighter controls in high-benefit areas.

VIII. CONCLUSIONS

Cost-based reform -- such as offsets, bubbles, and more ambitious incentive schemes -- are attractive because they allow significant cost savings without the need to reduce environmental-quality goals. They achieve their savings by exploiting wide variations in the marginal costs of controlling emissions under uniform standards. Benefit-based reforms can achieve similar results, by exploiting wide variations in the marginal benefits of control.

Benefit-based flexibility is not a substitute for cost-based reform, but rather an important complement. In allocating control efforts, efficiency demands that we be sensitive to both marginal costs and marginal benefits. Just as it makes little sense to impose a uniform standard on plants with very different costs, so too is there little to recommend uniform treatment for plants with very different benefits. Cost- and benefit-based flexibility are both designed to direct control expenditures where they yield the largest improvements.

The two major objections to benefit-based flexibility are that it would be difficult to administer and that it would be unfair. As we have stressed throughout this paper, however, relatively crude and simple approaches to benefit-based flexibility can yield large gains, and may ease enforcement. More sophisticated and finely tuned systems may do even better in some circumstances, but are not essential.

The fairness objection is more difficult to counter, if only because definitions of fairness are numerous and elusive. Any policy change, no matter how beneficial overall, causes some individuals and firms to suffer losses, which they may well regard as "unfair." Benefit-based flexibility would create two obvious types of losers: plants in densely populated areas would face higher costs (or at least a loss in competitive position as firms in other areas lowered their costs), and residents of lightly populated areas would see some increases in pollution. Judged against a status quo of uniform standards, these losses may seem unfair. Judged against more general criteria, however, we believe that benefit-based flexibility is at least as equitable as the current system.

It is myth to think that the current system of uniform <u>emission</u> standards produces uniform results. Different firms in an industry and different industries face different compliance costs -- the result that provides the motivation for cost-based flexibility -- and expenditures to protect different households would vary enormously even if costs were identical. As we argued in Section V, uniform standards implicitly place a much lower value on protecting the residents of more densely populated areas. Moreover, they lead to very unequal levels of risk, because uniform emission standards do not lead to uniform ambient concentrations. This variation in risk is exacerbated if marginal damages rise with concentrations. Benefit-based flexibility would reduce variations in the implicit valuations of risk reduction and, in most cases, would also reduce variations in the absolute levels of risk.

Benefit-based flexibility would have adverse effects on the competitive positions of some firms located in densely populated areas and, consequently, on the ability of such areas to attract and retain heavily polluting industries. These objections have little force for sources owned by individuals for personal use (primarily automobiles) or for firms that compete in local markets. For firms competing in national markets, differential standards would be added to a long list of factors to be considered in choosing plant locations. It makes no more sense to structure environmental laws to make firms indifferent about sites with very different environmental consequences than it would to adopt a national policy of uniform water rates so that water-intensive industries would be indifferent between locating along rivers or in the desert.

It is also a myth that the application of benefit-based flexibility to environmental regulation would be a unique deviation from traditional values of uniformity. The safety regulations for jumbo jets, for example, are more stringent than those for small charter aircraft that carry only a few passengers. Traffic lights are placed at heavily traveled intersections. Fire stations are more plentiful in cities than in rural areas. Zoning ordinances limit where plants may be located. The Interstate Commerce Commission requires that trucks carrying hazardous cargoes seek to avoid tunnels, places where crowds have assembled, and other areas where the consequences of an accident would be unusually severe (Breyer, 1982, p. 104). All of these are examples of the basic principle of benefit-based flexibility. Even in the environmental area, we find examples of the principle, including the use of separate state plans to set standards for most existing sources of air pollution and the authority given California to set more stringent interim auto emission standards.

The most compelling argument for adopting benefit-based flexibility in environmental regulation is not equity or precedent, but the large gains in net benefits that it offers. The empirical studies reviewed in Section II suggest that the potential gains are likely to be on a par with those from cost-based reforms, and thus justify vigorous efforts to overcome political objections.

We have no illusions that benefit-based flexibility will be adopted on a large scale in the near future. The slow pace at which cost-based reforms have been adopted is instructive in this regard. The best strategy is probably one of limited experimentation, introducing benefit-based flexibility into selected areas of regulation. Potential candidates include the new source performance standards for air emissions, the Best Available Technology (BAT) requirements for toxic water pollutants, and the Section 112 standards for hazardous air pollutants. Such experiments could help refine administrative details, and also help convince skeptics of the importance of incorporating benefit-based flexibility in environmental regulation as it continues to mature in the 1980s.

NOTES

1. Under the Clean Air Act, most uniform source standards apply only to new sources; existing sources are covered by state implementation plans, which are subject to national uniform ambient standards and other requirements. In some cases, however, including hazardous air pollutants covered by Section 112, existing sources are also subject to uniform national standards. Under the Clean Water Act, new and existing sources are subject to uniform national standards.

2. In 1980, the Council on Enviromental Quality estimated that total federal expenditures on air and water pollution control over the decade from 1979 to 1988 would be $588.8 billion, in 1979 dollars (Council on Environmental Quality, 1980, p. 397). Using information on the implicit price deflator for 1980-82 (Council of Economic Advisors, 1982, p. 236, for 1980 and 1981 and assuming a rate of 3.9% for 1982), these expenditures total $727.6 billion in 1982 dollars.

3. The economic literature criticizing standards and promoting incentives is extensive. For brief, nontechnical presentations, see Ruff (1970, 1981). For more extended, but also nontechnical discussions, see Kneese and Schultze (1975) and Anderson et al. (1977). Baumol and Oates (1975) provide a book-length, more technical treatment that cites much of the relevant literature.

4. For a more detailed theoretical treatment, see Nichols (1981, ch. 6).

5. If the marginal cost function is linear (i.e., total costs are a quadratic function of the reduction in emissions), it can be shown that the savings due to benefit-based flexibility will be proportional to the variance in marginal benefits and inversely proportional to the slope of the marginal cost curve.

6. As Nichols (1981, ch. 6) shows, the incentive to move with differential standards will be exactly right only if the marginal cost of control is unit elastic with respect to the level of emissions (not emissions controlled).

7. A third group of empirical studies evaluates the advantages of accounting for the location of emitters within a single air quality region or river basin. The regulatory problem is to devise a set of controls on individual emitters that results in achieving an ambient quality target at some critical point (typically the location estimated to have the highest concentration). The baseline option is to require

an equal proportional reduction from all emitters regardless of compliance costs or location. These studies then calculate the cost savings from an "emissions least cost" alternative that accounts for cost differences among emitters, and from an "ambient least cost" alternative that accounts for differences in in both costs and the proximity of emitters to the critical point. These studies typically find substantial cost savings from accounting for location. For examples in air and water pollution, see Atkinson and Lewis (1974) and Kneese and Bower (1968), respectively.

8. The relationship between emissions and concentrations depends on the specific substance. For example, the formation of photochemical oxidants ("smog") depends upon the emissions of hydrocarbons and nitrogen oxides as well as the presence of sunlight. The effect of decreasing nitrogen oxide emissions on oxidant concentrations will thus depend upon the concentration of hydrocarbons and the amount of sunlight. Indeed, there is some evidence that decreasing nitrogen oxide emissions may actually increase oxidant concentrations in some cases. See Masters (1974, Chapters 8-10) for an general overview of air pollution and Masters (1974, Chapters 4-7) for a general overview of water pollution.

9. For a useful discussion of the concept of thresholds in relation to carcinogenic and noncarcinogenic effects, see the National Academy of Sciences (1977, ch. II). Even if the true dose-response model exhibits thresholds, however, the expected dose-response function (which is the one that decision makers should use) may be close to linear at low doses if there is substantial uncertainty about the level of the threshold (See Nichols, 1981, ch. 4, for an example).

10. To test for the relationship between population density and air pollution, we computed simple correlations for a sample of 76 cities (Berry et al., 1974, pp. 258-263). The results are:

Pollution variable	Correlation with density
Air quality index	0.22
Sulfur dioxide	0.50
Particulates	0.01

Similar results are reported by Mills, Feenberg, and Zisler (1978), who estimated the independent effect of population density on particulate and sulfur oxides concentrations in a sample of 38 urban areas after accounting for other factors likely to affect air quality. They found positive regression coefficients for both variables, with the coefficient for sulfur dioxide both larger and more statistically significant than that for particulates (p. 180-182).

11. See Nichols (1981, ch. 6) for a detailed examination of the impact of a correlation between marginal costs and damages on the performances of alternative regulatory strategies.

12. Kalt's benefit estimates for the West are questionable because they are dominated by "existence value" -- benefits to people who will never use or even see the land in question, but who gain pleasure from knowing that it has been restored to its natural state. If that component of his estimate is deleted, the stripmining reclamation law yields negative net benefits in all regions of the country.

13. The 1977 Amendments to the Clean Air Act established a ranking scheme for areas with air quality above the national ambient standards. National parks, national monuments, and scenic wilderness areas were designated Class I areas, with the strictest limits on sulfur and particulate emissions. Stringent controls can be justified because -- unlike health effects -- the marginal damages from emissions tend to be greatest at low concentrations and because individual valuations of visibility are much higher in scenic areas (see Repetto, 1983).

14. For federally-financed projects, the site-specific effects must be documented in an Environmental Impact Statement to meet the requirements of the National Environmental Policy Act. Many states have similar requirements for projects that require state approval. See Council on Environmental Quality (1980, pp. 370-386, 426-430).

15. The most widely used method of calculating overall exposure to aircraft noise, the Noise Exposure Forecast, assumes that one nighttime flight (10 p.m. to 7 a.m.) is equal in annoyance to about 12 daytime flights. See Harrison (1983, p. 46).

16. This assumes that the current regulations are correct "on average." Some observers (e.g., Pashigan, 1982) argue that environmental legislation was designed to meet the needs of older, densely populated areas with severe pollution problems, and that uniformity was imposed to protect such regions from the exodous of industry. To the extent that benefit-based flexibility would not lead to any changes in current standards for densely populated areas, only a relaxation in other areas, densely populated areas may "lose," as they do not gain tighter standards (though environmental quality may improve because some large emitters leave the area) and other areas become relatively more attractive to industry.

17. See Nichols (1981) for a much more extended and detailed treatment of the issues involved in modifying incentive schemes to take account of variations in the marginal benefits of controlling emissions.

18. Montgomery (1972) argues that the best way to meet ambient targets is to use "pollution licenses," which establish permits in air quality rather than emissions. Under that scheme, many sources would have to purchase permits in several different markets, as their emissions would affect air quality in several areas. Tietenberg (1974a,b) proposes a similar approach, which he calls an "air-rights market."

REFERENCES

American Enterprise Institute. *Major Regulatory Initiatives During 1980*. Washington, DC: American Enterprise Institute, 1981.

Anderson, F. R. et al. *Environmental Improvement Through Economic Incentives*. Baltimore: Johns Hopkins University Press, 1977.

Atkinson, S. E. and D. H. Lewis. "A Cost-Effectiveness Analysis of Alternative Air Quality Control Strategies," *Journal of Environmental Economics and Management*, 1 (1974), 237-50.

Baumol, W. J. and W. E. Oates. *The Theory of Environmental Policy -- Externalities, Public Outlays, and the Quality of Life*. Englewood Cliffs, NJ: Prentice-Hall, Inc., 1975.

Berry, B. et al. *Land Use, Urban Form and Environmentaal Quality*. Chicago, IL: University of Chicago, Department of Geography, 1974.

Birdsall, T. H. "Should Environmental Standards be Relaxed During Oil Supply Disruptions?" Cambridge, MA: Harvard University, 1982. Energy and Environmental Policy Center Discussion Paper E-82-01.

Breyer, S. *Regulation and Its Reform*. Cambridge, MA: Harvard University Press, 1982.

Council of Economic Advisers. *Economic Report of the President* Washington, D.C.: U.S. Government Printing Office, February 1982.

del Calvo, Jorge A. "Markets in Air: Problems and Prospects of Controlled Trading." *Harvard Environmental Law Review* 5 (1981): 377-430.

Haigh, J. "EPA and the Regulation of Coke Oven Emissions: A Cost-Benefit Analysis," unpublished paper, Harvard University, April, 1982.

Harrison, D. *Who Pays for Clean Air*. Cambridge: Ballinger, 1975.

Harrison, D. "The Regulation of Aircraft Noise," in Schelling et al. (1983).

Harrison, David Jr. and Paul R. Portney. "Who Loses from Reform of Environmental Regulation." in Wesley A. Magat, editor. *Reform of Environmental Regulation*. Cambridge, Mass.: Ballinger Publishing Company, 1982, pp. 147-179.

Kalt, Joseph P., "Energy Supply, Environmental Protection, and Income Redistribution Under Federal Regulation of Coal Strip Mining," Harvard Institute of Economic Research Discussion Paper Number 880, Cambridge, Mass.: Harvard University, January 1982.

Kneese, A. V. and B. T. Bower. *Managing Water Quality: Economics, Technology, Institutions*. Baltimore: Johns Hopkins Press, 1968.

Kneese, A. V. and C. L. Schultze. *Pollution, Prices and Public Policy*. Washington, D.C.: Brookings Institution, 1975.

Lathrop, J. and J. Linnerooth "The United States Case Study," in H. Kunreuther and J. Linnerooth, *Risk Analysis and Decision Processes*, preliminary draft. Laxenburg, Austria: IIASA, March, 1982.

Leone, R. A. and J. E. Jackson. "The Political Economy of Federal Regulatory Activity," working paper, Graduate School of Business, Harvard University, March 1978.

Luken, Ralph A., Daniel J. Basta, and Edward H. Peckham. *The National Residuals Discharge Inventory*. Washington, D.C.: National Research Council, 1976.

Masters, Gilbert. *Introduction to Environmental Science and Technology*. New York: John Wiley and Sons, 1974.

Miller, C. *Exposure Assessment Modelling: A State-of-the-Art Review*. Report prepared for U.S. EPA; EPA-600/3-78-065, 1978.

Mills, Edwin S., Daniel Feenberg, and Randall Zisler. "Environmental Problems in the Mature Metropolis," in *The Mature Metropolis*, Charles L. Leven, editor. Lexington, MA: D.C. Heath and Company, 1978.

Montgomery, W. C. "Markets in Licenses and Efficient Pollution Control Programs," *Journal of Economic Theory*, 5 (1972), 395-418.

National Academy of Sciences. *Drinking Water and Health*. Report of the Safe Drinking Water Committee, Washington, D.C., 1977.

Nichols, A. L. *Choosing Regulatory Targets and Instruments, with Applications to Benzene*. Ph.D. dissertation, Harvard University, 1981.

Nichols, A. L. "Regulation of Airborne Benzene," in Schelling et al. (1983).

Pashigan, B. P. "Environmental Regulation: Whose Self Interests are being Served," University of Chicago, working paper, August, 1982.

Perl, L. J. and F. C. Dunbar. "Cost-Effectiveness and Cost-Benefit Analysis of Air Quality Regulations," American Economic Review, 72 (May, 1982), 208-213.

Rabin, R. L. "Ozone Depletion Revisited," Regulation (March-April 1981), 32-38.

Repetto, R. "Air Quality Under the Clean Air Act," in Schelling et al. (1983).

Roberts, M. J. "Environmental Protection: The Complexities of Real Policy Choice," in I. K. Fox and N. A. Swainson (eds), Water Quality Management: The Design of Institutions. Vancouver: University of British Columbia Press, 1975.

Ruff, L. "The Economic Common Sense of Pollution," The Public Interest (Spring 1970), 69-85.

Ruff, L. "Federal Environmental Regulation," in L. Weiss and M. Klass (eds.), Case Studies in Regulation. Boston: Little, Brown, 1981.

Schelling, T. C. et al. Incentives for Environmental Protection. Cambridge, MA: MIT Press, 1983.

Tietenberg, T. H. "Derived Decision Rules for Pollution Control in a General Equilibrium Space Economy," Journal of Environmental Economics and Management, 10 (1974a), 3-16.

Tietenberg, T. H. "The Design of Property Rights for Air Pollution Control," Public Policy, (Summer, 1974b), 275-92.

Viscusi, W. K. "Labor Market Valuations of Life and Limb: Empirical Evidence and Policy Implications," Public Policy, 26 (Fall 1978), 359-86.

White, Lawrence J. Reforming Regulation: Processes and Problems. Englewood Cliffs, N. J.: Prentice-Hall, 1981.

Wilson, Richard, Steven D. Colome, John D. Spengler, and David Gordon Wilson. Health Effects of Fossil Fuel Burning: Assessment and Mitigation. Cambridge, Mass.: Ballinger Publishing Company, 1980.

Yellin, J. and P. L. Joskow. "Siting Nuclear Power Plants," unpublished paper, Massachusetts Institute of Technology, April, 1979.

PART 8

THE POLICY IMPLICATIONS OF NONCONVEX ENVIRONMENTAL DAMAGE FUNCTIONS

Robert Repetto

I. INTRODUCTION

THE IMPORTANCE OF PARTIAL INFORMATION ABOUT ENVIRONMENTAL BENEFITS

It is almost inevitable that the costs of protecting the environment should be easier to quantify than the benefits of doing so. The costs stem from the diversion of resources from the production of final goods and services. These resources typically have market values that can be added up. The benefits rarely have market values, because there is typically a "publicness" about environmental damages that precludes the enforcement of property rights in them or the operation of markets. The very sources of "market failure" that create the need for government regulation of environmental pollution make it difficult to establish the benefits of such regulation.

As a result, the costs of environmental protection have been much more inlfuential in the formation of regulations than have the benefits. The criterion of efficiency, that regulations should be set to maximize benefits net of costs, has not been perused very far, despite the emphasis afforded it by policy analysts. This is largely, though not entirely, due to the problem of measuring benefits. As a result, environmental regulations are widely perceived to be inefficient either in the sense that abatement costs are imposed that are not justified by the resulting benefits or in the sense that another pattern or level of abatement would create much larger net benefits.[1]

In recent years, pressure for regulatory reform, in the field of environmental protection and others, has centered on the achievement of greater efficiency through mechanisms to measure costs against benefits. Whenever possible, this has been promoted through greater reliance on market mechanisms. When markets are impossible, regulators are admonished to compare benefits and costs in policy analyses and to frame regulations accordingly.[2]

The problem in implementing such instructions is that, given the conceptual and empirical problems in estimating benefits, conscientious analysts are usually forced to report such wide ranges as confidence intervals or their equivalents around their estimates of benefits that little guidance can be gleaned from them. These wide ranges arise from the compounding of uncertainties. Benefit estimates are constructed through analysis of a sequence of effects: typically, (a) the reduction in emissions consequent to a change in control measures; (b) the reduction in ambient levels consequent to a change in emissions; (c) the response of receptors (human or non-human) to a change in ambient levels; (d) the economic valuation of that response. Since serious uncertainties exist at each step, the final estimates inevitably have very wide confidence limits.

Benefit estimates and benefit-cost comparisons cannot be used, therefore, to fine-tune regulatory decision making. For a commercial firm, a prospective rate of return over costs of 1.2 might justify an investment decision, but a benefit:cost estimate of 1.2 should rarely, if ever, be sufficient to justify a regulatory decision. The margins of error are too wide. Moreover, it is unlikely that refinements in measurement, improvements in data, or scientific research will appreciably narrow those margins in the intermediate term, to the extent that the social profitability of regulatory decisions can be analyzed through benefit:cost analysis with the same scale of precision that market investment decisions can be analyzed.

What then can be the role of benefit analysis in regulatory policy making? The temptation is to relegate it to the research agenda until such time, in the vague future, as the methodology and data are strong enough to bear the weight of policy analysis. However, the proper role is to use benefit analysis to prevent gross misallocation of resources, using such partial and imprecise estimates as can be generated at present. Gross misallocations may be detectable even with imprecise benefit estimates.

Using benefit analysis in this way usually involves strategically employing partial information about pollution damages. For example, damages may be roughly proportional to the number of people or other receptors exposed. Since exposure data are usually obtainable through enumeration, a geographical mapping up to a factor of rough proportionality can be obtained. For another example, damages may be zero over a certain range, because of thresholds of tolerance. In the case of environmental effects, information about the stability and instability properties of the ecosystem can provide useful information about damages and potential benefits from regulation. Knowledge that some contaminants persist as stocks may imply that additional emissions give rise to a long-term flow of damages, not simply a current adverse impact. This information supplies useful insights into the potential magnitude of damages.

All these examples illustrate partial information about damages that can be useful in preventing gross errors in regulatory policy making, even in the absence of complete and accurate benefits estimates. Another important example is knowledge of the behavior of incremental damages with increasing levels of pollution: in short, the convexity or nonconvexity of the damage function.

II. THE GENERAL PROBLEM OF NONCONVEX ENVIRONMENTAL, DAMAGES

The usual assumption is that environmental damage functions, which represent the physical or economic losses from pollution as functions of the concentration of the pollutant, rise with increasing concentrations at a non-decreasing rate. That is, they are convex from below. A common rationalization for this assumption is that the receptor, whether it be the human organism or the natural environment, has a certain tolerance or buffering capacity which permits it to cope with low concentrations without appreciable harm, but suffers progressive functional impairment at higher dosages. Figure 1a illustrates convex damage functions.

Since abatement costs are typically convex over a wide range, the costs rising at a non-decreasing rate as emissions are progressively reduced, the efficiency goal of maximizing benefits from environmental protection net of costs implies a search for the best level of pollution, an interior optimum at which the marginal costs of abatement equal the marginal benefits. The search for this level implies the precise use of benefit:cost analysis: regulatory policy should be adjusted so long as incremental benefits exceed incremental costs, or vice versa, as illustrated in Figure 2.

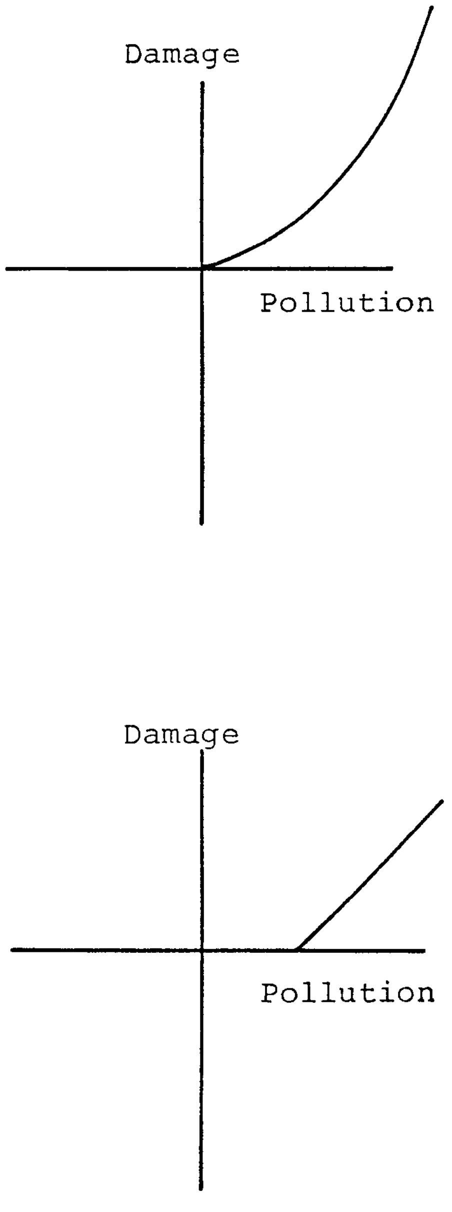


Figure 1A. Convex Damage Functions

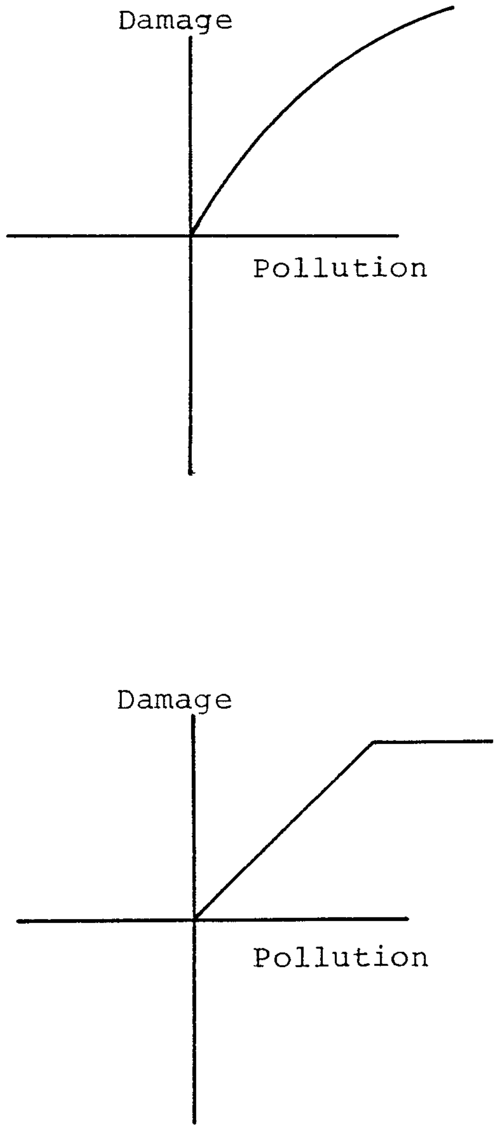


Figure 1B. Nonconvex Damage Functions

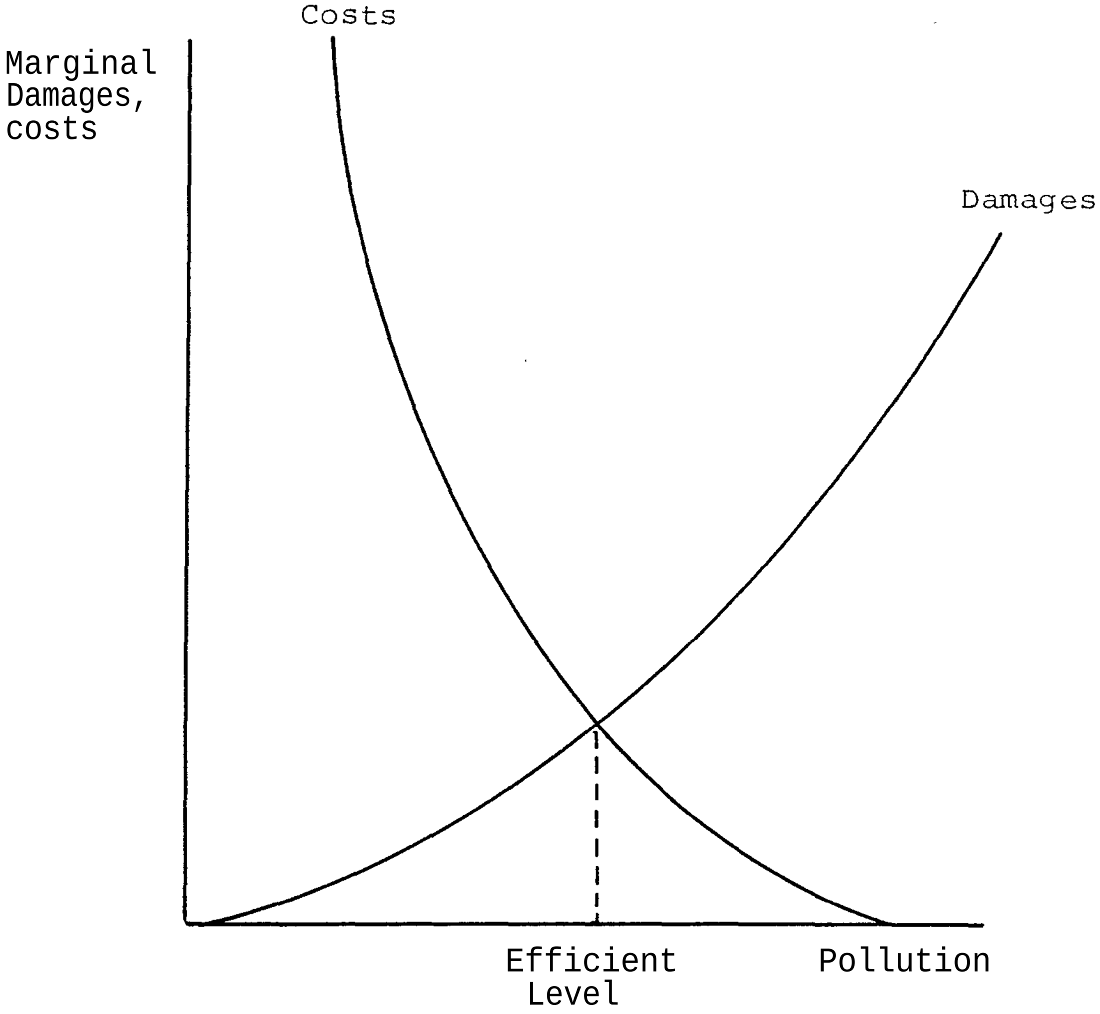


Figure 2. Marginal Damage and Cost Curves

However, not all damage are convex. Some are as illustrated in Figure 1b. This has long been recognized, mostly in the form of qualifications and exceptions to the conventional representation of the problem.[3] It has drawn the attention of economists for two reasons. First, it raises the possibility that regulatory policy perhaps should not be concerned with finding the best balance between pollution damage and abatement costs, the interior optimum of Figure 2, but should pursue an all-or-nothing policy which either bans emissions totally in certain areas or else leaves them uncontrolled. Second, it raises the possibility that decentralized incentives to private decision makers conveyed through prices and markets might not lead to an efficient allocation of resources. This impugns the economist's prized recommendation of Pigovian pollution taxes as an instrument of regulatory policy, and also, more fundamentally, the efficacy of price signals in the allocation of resources in the economy at large.[4]

The broader problem, the nonconvex damage functions may lead to nonconvexities either in marginal rates of transformation between commodities in production, or marginal rates of substitution in consumption, is illustrated in Figure 3 for the production side. Electricity and laundry are final items of production, but electricity generation results in smoke as a by-product, which interferes with the cleansing of laundry. Figure 3 shows that the nature of the by-product relationship by which electricity production leads to smoke pollution is critical to the shape of the transformation between electricity and laundry, the final goods. In the lower left quadrant, the "transfer"

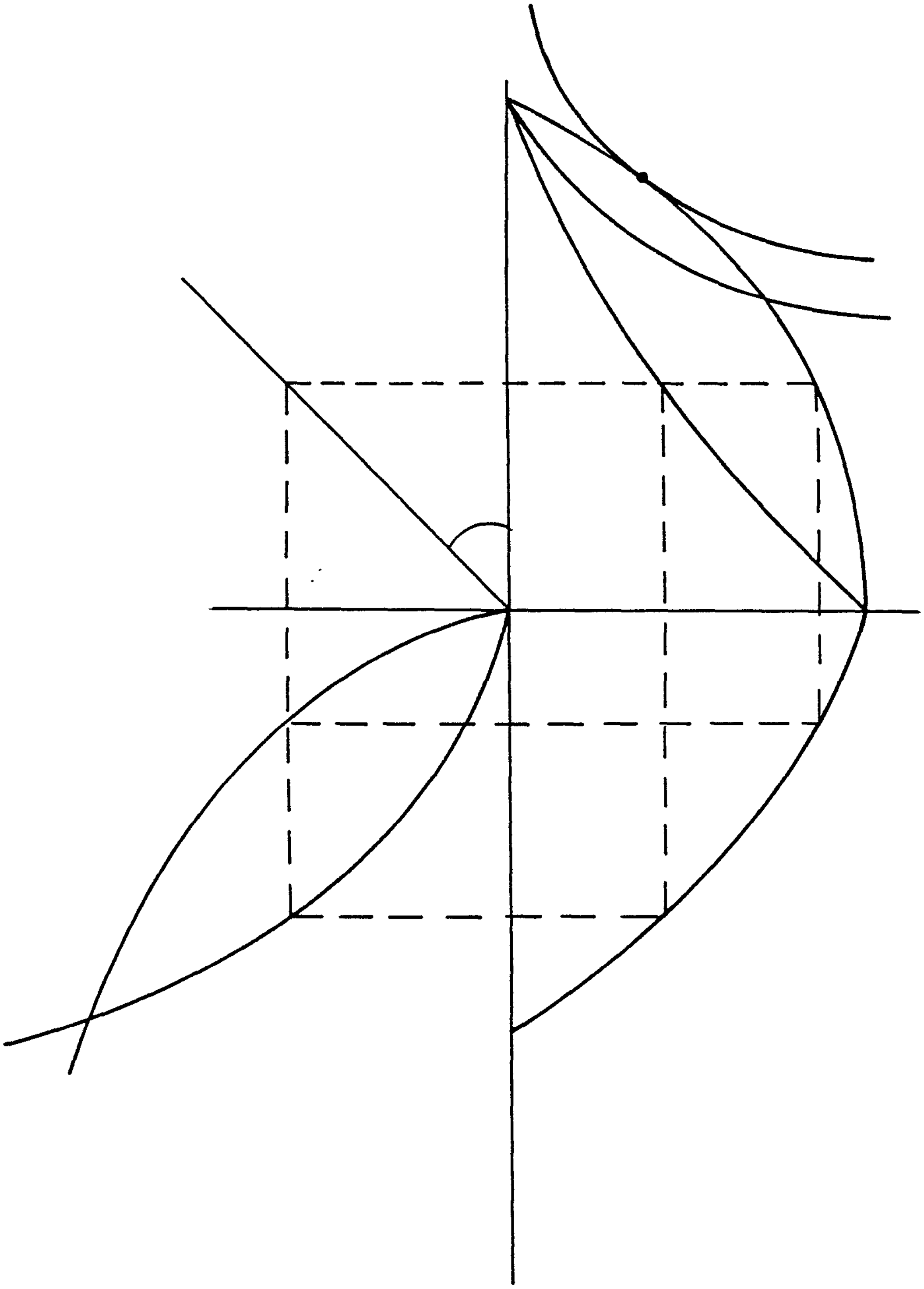


Figure 3. Nonconvexities in Marginal Rates of Transformation between Commodities in Production, or Marginal Rates of Substitution in Consumption

function relating smoke to electricity, OB, is nonconvex (in electricity). Even though the effects of smoke on laundry is convex, the production possibilities transformation between electricity and laundry, the final goods, MNP, is not. On the other hand, the convex transfer function OA leads to a conventionally convex production transformation, MOP, between electricity and laundry.

The significance of this illustration is that it reveals the possibility that nonconvex damage functions may lead to all-or-nothing choices in the entire economy. In the absence of nonconvexities, given consumer preferences, resources would be allocated to produce a commodity bundle like Q, through the action of market prices. With nonconvexities, given the consumer preferences shown, it is optimal to be well-lighted and heated but unclean, and there are no sustainable prices that would allow laundries to remain in business. It is clear that the key element in the situation is the nonconvex damage function between electricity and smoke.

The narrower problem, of all-or-nothing regulatory choices, is illustrated in Figure 4. Nonconvex damage functions imply that marginal damages decline over a range as pollution concentrations increase. This means that, if marginal damages exceed marginal abatement costs at very low concentration levels, it may be efficient to allow no emissions at all, as illustrated in Figures 4a and 4b. On the other hand, if marginal costs exceed marginal damages at very low concentrations, it may be efficient not to control emissions at all, as illustrated in

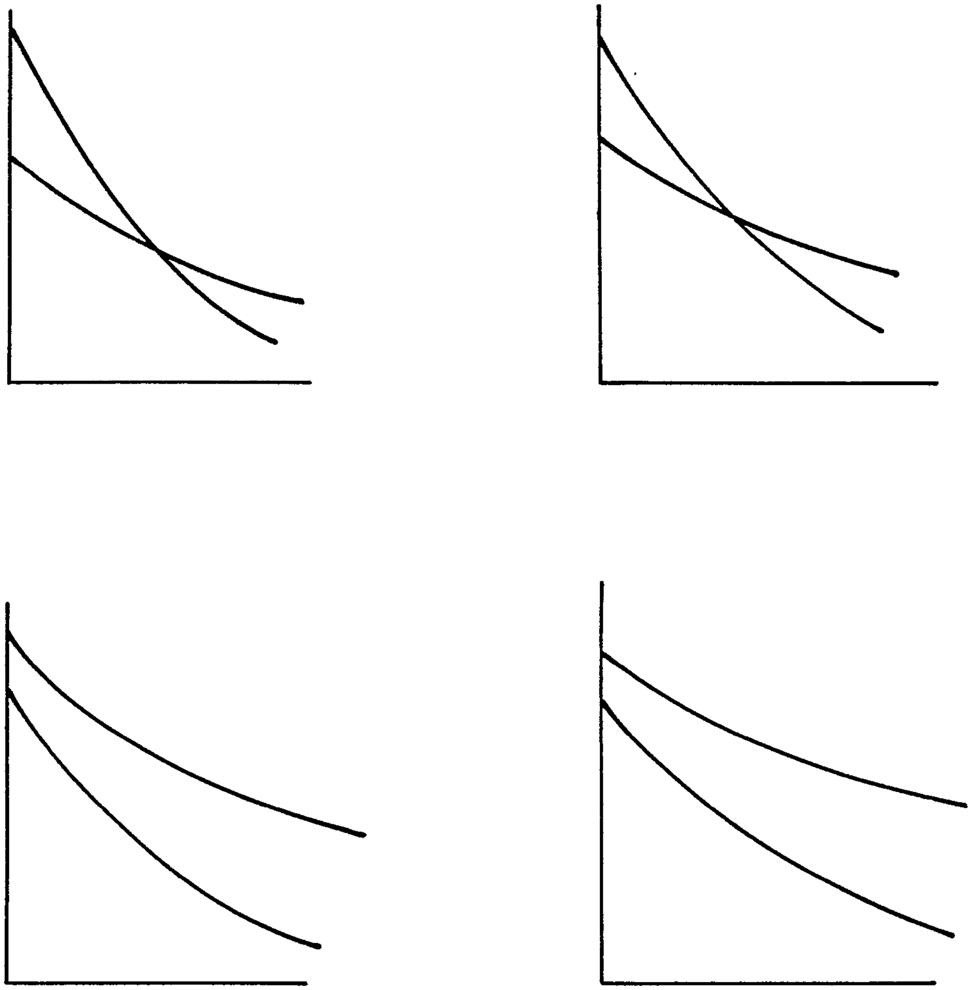


Figure 4. All-or-Nothing Regulatory Choices:
Marginal Costs (C) and Marginal Damages (D)

Figure 4d. The level of control at which marginal costs equal marginal benefits may either minimize net benefits, as in Figure 4a, or maximize them, as in 4c.

Of what value to policy analysts is the information that damage functions for specific environmental problems are convex? It directs the analysis to extreme values and reduces the range of search for efficient policy options. If analysis suggests that incremental benefits at low pollution levels exceed costs of protection, and are less than costs at high levels, then an all-or-nothing choice is indicated. This may be simpler, and more appropriate, than a search for the right degree of control.

To what extent are nonconvex damage functions empirically important? It appears that there are numerous and important examples, some arising out of physical characteristics of pollution processes, others out of behavioral responses of pollution victims. The impairment of visibility by fine particulates is an important case of a technical nonconvexity.[5] Figure 5 shows the decline of visibility with increasing ambient concentrations. Incremental damages sharply diminish. The psychological losses due to the congestion of natural environments is a widespread instance of a behavioral nonconvexity. Figure 6 shows the loss of user satisfaction as a function of congestion, measured in terms of the number of encounters with other parties in wilderness recreation trips. It shows that the incremental losses as congestion increases diminish sharply: it is the first one or two intrusions into solitude and privacy that do the most damage.

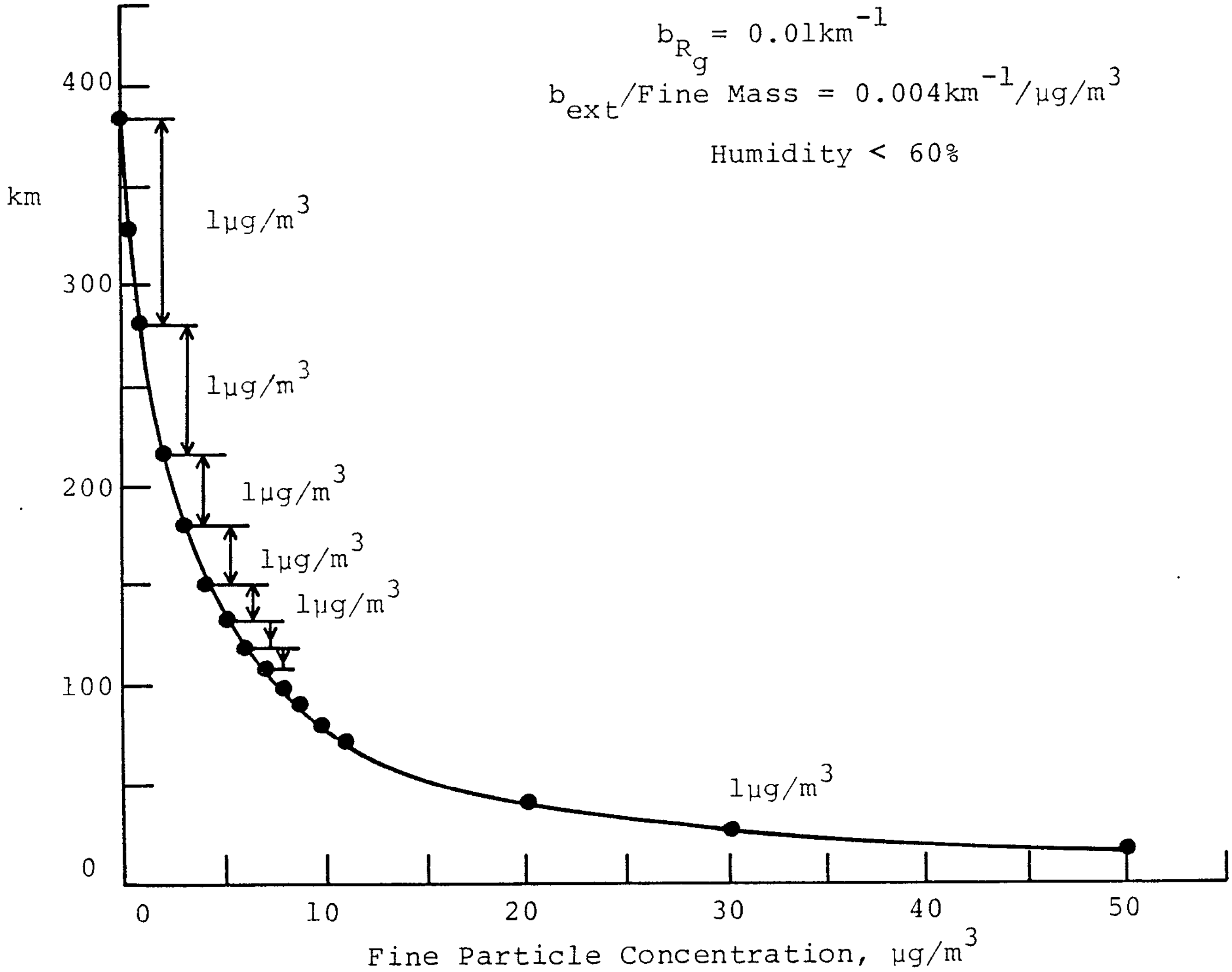


Figure 5. Decline of Visibility with Increasing Ambient Concentrations

Source: R. Repetto, "The Economics of Visibility Protection," <u>Natural Resources Journal</u>, XXI(2), April 1981.

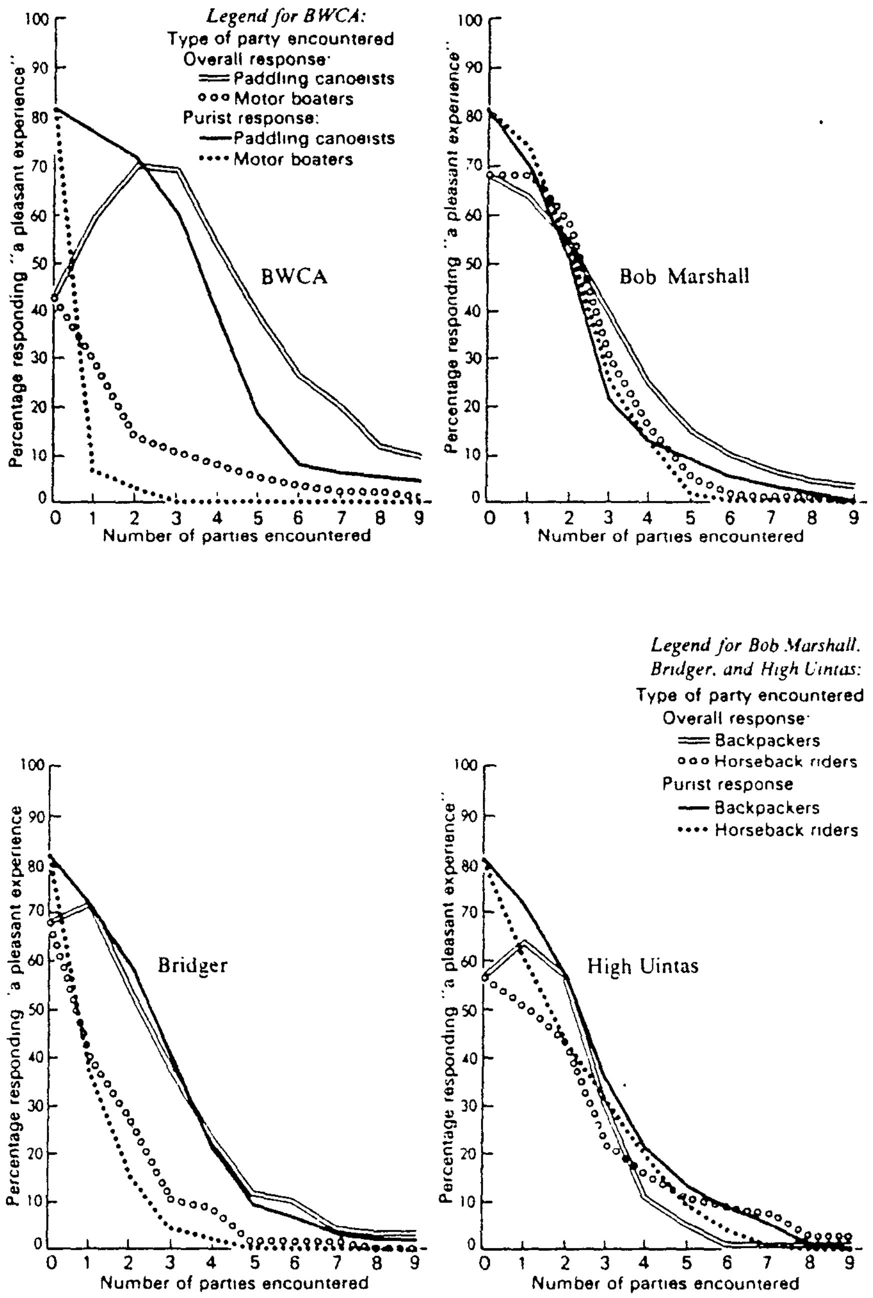


Figure 6. Satisfaction Curves for BWCA, Bob Marshall, Bridger, and High Uintas

Source: George H. Stankey, "A Strategy for the Definition and Management of Wilderness Quality," in John V. Krutilla, ed., <u>Natural Environments: Studies in Theoretical and Applied Analysis</u>, The Johns Hopkins University Press, Baltimore, Md., 1972, p. 108.

A phenomenon known to economists as averting behavior creates another important class of nonconvexities. As pollution damages increase with rising concentrations, the victims are induced to take averting actions, which may be to relocate away from the pollutant, to install cleaners or barriers, or one of many other possible averting strategies.[6] The result, in the extreme case of relocation, is an absolute upper bound to damage, which implies zero marginal damages beyond some level of concentration, and a non-convexity similar to that portrayed in Figure 1b.

The remainder of this paper explores in detail another important technical nonconvexity that arises in the generation of photochemical oxidants -- atmospheric smog. Since the oxidant problem is the most widespread air pollution problem in the United States, the existence of nonconvexities considerably extends the significance of the phenomenon.

III. NONCONVEXITIES IN THE FORMATION OF PHOTOCHEMICAL OXIDANTS

The problem of photochemical oxidants, smog, is one of the most widespread and persistent of air pollution problems. Smog over the Los Angeles basin, the urbanized Eastern seaboard and other metropolitan areas, provided one of the earliest stimuli for the regulation of automotive emissions. Ozone, the indicator species for photochemical oxidants, is the substance for which national ambient air quality standards are most widely violated.[7] As the monitoring network has been expanded, it has been found that large rural and suburban areas downwind of urban concentrations frequently experience ozone levels in excess of primary standards. In the most severely affected regions, like New York and Los Angeles, it has been that even the most drastic abatement of precursor emissions would probably not suffice to eliminate the problem.[8]

The buildup of ozone and photochemical oxidants results from complex reactions involving reactive hydrocarbons and nitrogen oxides, in the presence of solar energy. The chemistry involved is complicated, and in the process is rendered very much more complex by the storage aloft in inversion layers of ozone and precursors, and the transport of reactive materials downwind over the course of several diurnal cycles.[9] There is considerable uncertainty about process of oxidant formation and transport over several days and at regional scale. Detailed modeling and simulation of the process is very demanding of data and computational time. Much of the present body of knowledge about oxidant formation rests on smog chamber results.

The basic atmospheric chemistry underlying the role of nitrogen oxides builds on two interacting processes: the photolysis of NO_2 and the oxidization of nitric oxide. The key reactions are:

$$NO_2 + hv \longrightarrow NO + O$$

$$O_2 + O \longrightarrow O_3$$

$$NO + O_3 \longrightarrow NO_2 + O_2$$

which tends toward a photostationary state in which the ozone concentration is related to the ratio of NO_2 and NO:

$$(O_3) = k\ (NO_2)/(NO)$$

Since NO is the principal oxide of nitrogen emitted by combustion sources, ozone concentrations tend to be reduced near strong sources and increased by oxidized products at greater distance.

The role of hydrocarbons in the process is very complex, but is thought to lie basically in the provision of alternative pathways for the oxidization of NO to NO_2, preserving higher concentrations of ozone. Low NO concentrations can limit the speed of the process, so that an increase in NO_x concentrations can raise ozone levels, while at higher NO_x levels, peak ozone concentrations are found to be roughly proportionate to the HC/NO_x precursor concentration ratio.[11]

This implies a strong technical nonconvexity in the relationship of NO_x precursor emissions to peak ozone concentrations. Not only do the marginal effects on peak ozone levels decrease as NO_x emissions increase, they become negative: at higher concentrations, ozone formation is inhibited, at least for short irradiation and atmospheric residence times. Smog chamber experiments give results similar to those presented in Figure 7 below. For given HC concentrations, higher initial NO_x levels both retard the attainment of peak ozone concentrations and, beyond a critical level dependent on the HC/NO_x ratio, reduce the level of peak ozone concentration.

A similar nonconvexity exists in the relationship of peak ozone concentration to the level of the HC precursor. Although there is no stage at which increasing HC concentrations actually reduce ozone concentrations through any scavenging process, at high HC concentrations the impact of reductions in HC emissions may be considerably less than the impact when HC inputs are low and the HC/NO_x ratio lower. This is illustrated in Figure 8.

Translating these findings based on smog chamber experiments and chemical analysis into predictions about the impacts of control strategies on ambient air quality is extremely difficult. Models of photochemical pollution processes involving a variety of precursors, variable atmospheric conditions, transport, and multi-day episodes, are both inacccurate and expensive. For purposes of this paper, which are to explore and illustrate the implications of nonconvexities, rather than to formulate actual regulatory strategies, it is sufficient to employ a relatively simple model of the precursor-oxidant relationship. This model,

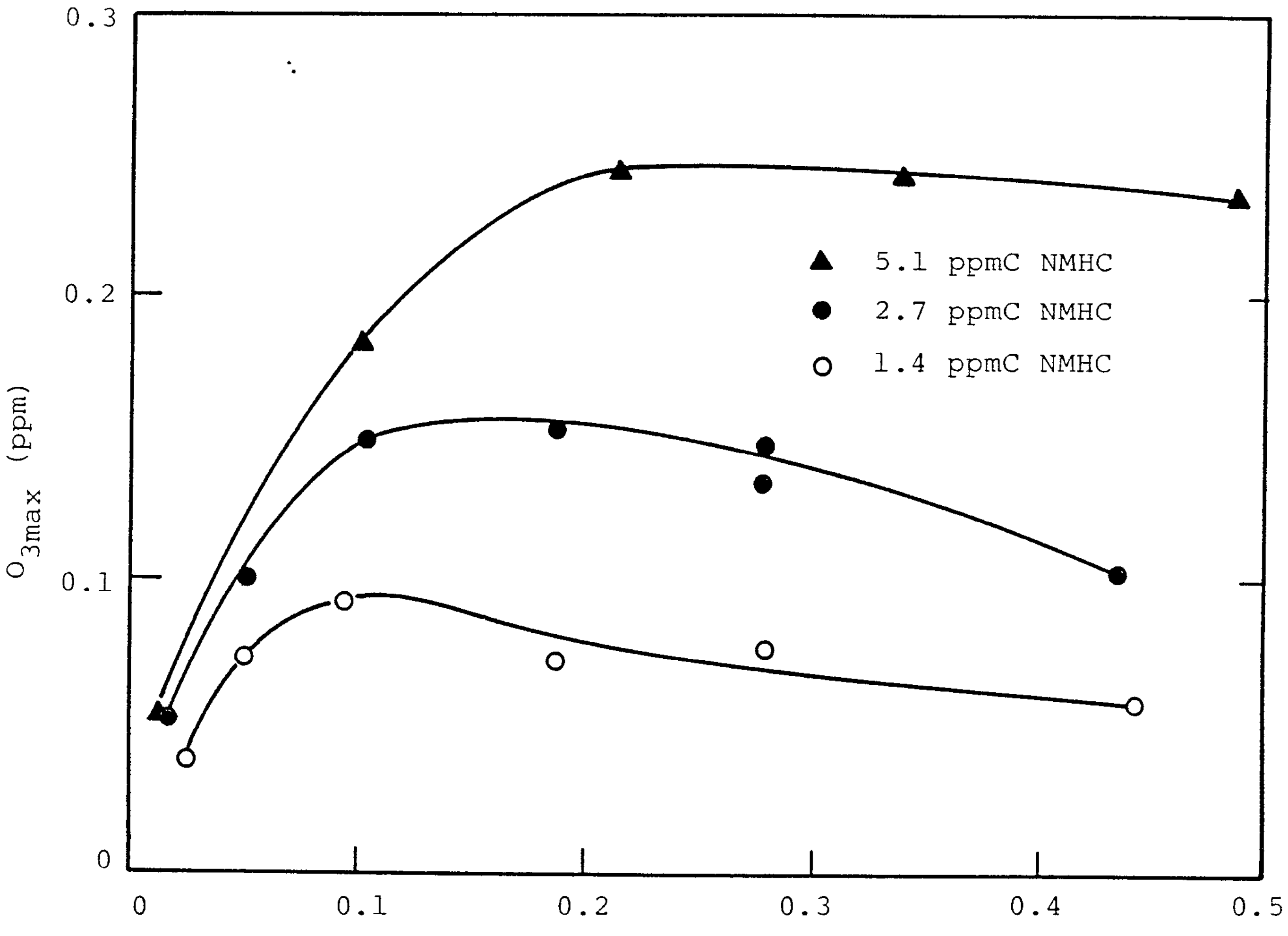


Figure 7. Effect of $(NO_x)_o$ on (O_3) maximum

Source: William Glasson, "Effect of Hydrocarbon and NO_x on Photochemical Smog under Simulated Transport Conditions," *Journal of the Air Pollution Control Association*, XXXI (11), November 1981, p. 1170.

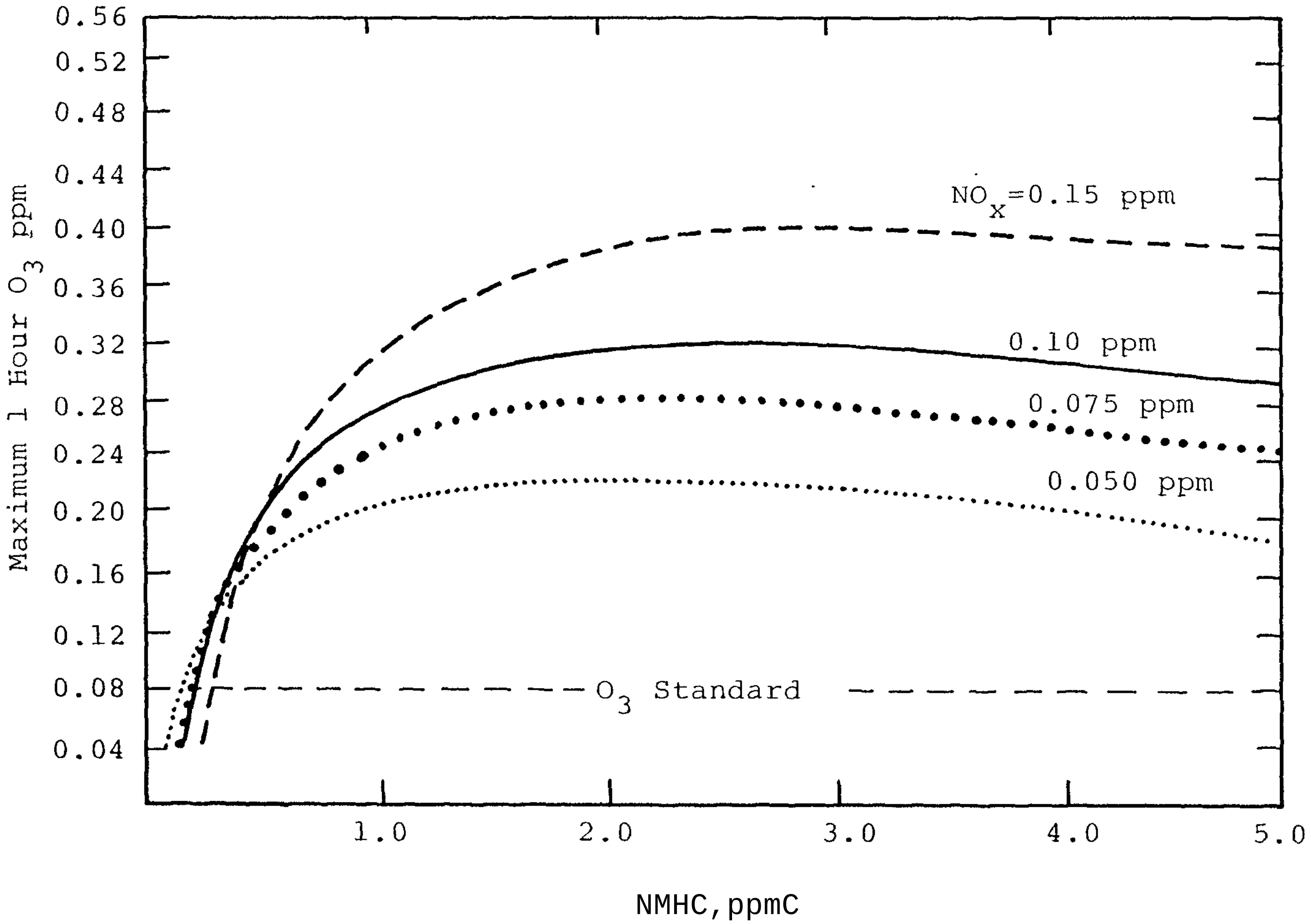


Figure 8. Relationship of Peak Ozone Concentration to Level of HC Precursor

Source: M.C. Dodge, "Combined Use of Modeling Techniques and Smog Chamber Data to Derive Ozone-Precursos Relationships," International Conference on Photochemical Oxidant Pollution and its Control, US E.P.A., Raleigh, N.C., September 1976 called EKMA ("empirical kinetic modeling approach") has been proposed and popularized by EPA officials and is widely used for planning purposes.[12] It adjusts "isopleths" -- loci concentrations of precursor NO_x and reactive hydrocarbon concentrations that yield equal maximum ozone concentrations -- derived from smog chamber experiments to conditions existing in specific urban airsheds. EKMA predicts the percentage reductions in early morning precursor concentrations required to achieve given percentage reductions in late afternoon maximum ozone concentrations.

The typical pattern of these isopleths is reproduced in Figure 9. Transects parallel to either the NO_x or HC axis demonstrate the pattern of decreasing marginal effects shown in Figures 8 and 9. The isopleths show absolutely declining impacts to increasing NO_x concentrations for low $HC:NO_x$ ratios, indicating an inverse association between maximum ozone concentrations and NO_x emissions under some conditions. These isopleths can be particularized to specific meteorological conditions and ambient ozone and precursor concentrations, without substantial change in these essential characteristics. Moreover, isopleths derived through other modeling techniques, both statistical and large-scale urban airshed simulations, also predict declining marginal effects of both precursors, and absolute declines in the effect of NO, on ozone concentrations over some range.[13]

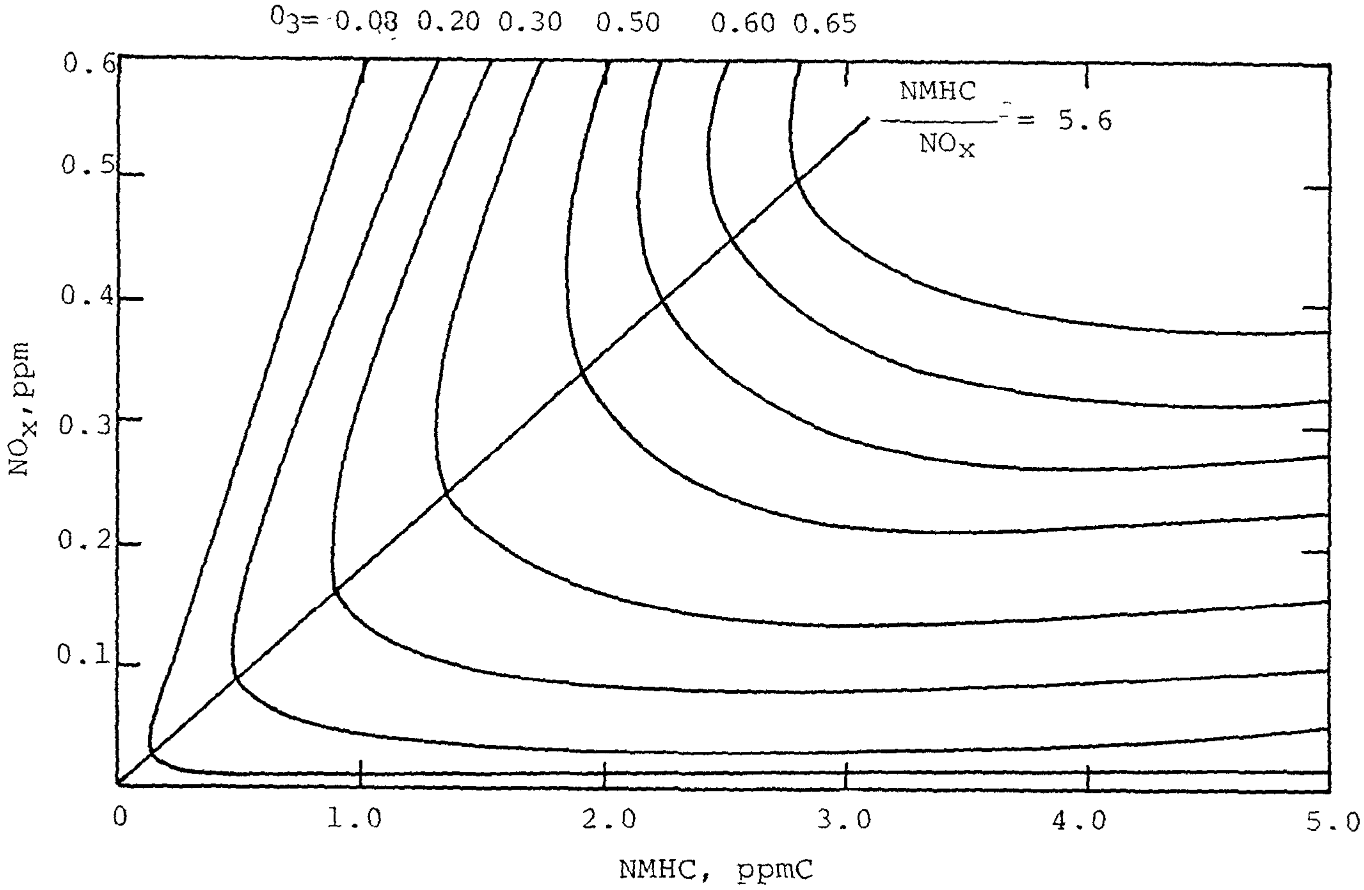


Figure 9. Ozone Isopleths Corresponding to Maximum One-Hour O_3 Concentrations

Source: M.C. Dodge, "conbined Use of Modelling Techniques and Smog Chamber Data to Derive Ozone-Precursor Relationships", International Conference on Photochemical Oxidant Pollution and its Control, U.S. EPA, Raleigh, N.C., 1976

IV. IMPLICATIONS OF NONCONVEXITIES FOR OXIDANT CONTROL STRATEGY

The interaction of nitrogen oxides and hydrocarbons in the photooxidation process has rarely been fully considered in the formulation of emissions control strategies. Air quality regions in violation of the national ambient zone standard have usually relied on hydrocarbon abatement strategies for reduction of ozone concentration, without explicit recognition of the effects of changing NO_x emission. Early efforts at formulating abatement requirements relied on an approximate "linear rollback" assumption: i.e., that the reduction in ozone concentrations would be proportional to the reduction in hydrocarbon emissions, independently of changes in NO_x emissions. The early use of EKMA in formulating control strategies determined the rollback, if any, in NO_x emissions by the requirements of attaining the national ambient standard for nitrogen dioxide, and, given that level of project NO_x emissions, determined the hydrocarbon abatement needed to meet the ozone standard.[14] This approach, while taking explicit account of the chemical interaction of the precursors, ignored the economic interaction. No attempt was made to find the least-cost pattern of emissions reduction which would result in attainment of the ozone (and NO_2) standards. In regions, like Los Angeles, with severe oxidant problems, there have been efforts at sophisticated photochemical modeling of precursor interactions to investigate the feasibility of attaining standard throughout the airshed. As these simulations demonstrated that only Draconian measures to abate hydrocarbons would yield the required ozone reductions, more emphasis has been given to the implications of NO_x controls. Some researchers have concluded that NO_x emissions should be allowed to increase in such metropolitan areas, as a cost-effective means of reducing ozone levels in the urban center.[15] These conclusions are based on the finding that maximum ozone levels for one-day irradiations would be lower with greater NO_x availability, given the HC:NO ratio prevalent in the downtown area. This suggestion that NO_x control requirements should be relaxed has stimulated a vigorous debate about the downwind effects on suburban and rural ozone levels, in regions where transported ozone and precursors are significant and where $HC:NO_x$ ratios can be much higher.[16]

Full consideration of the range of possibilities for oxidant control requires that both the effects of various patterns of HC and NO_x abatement and the costs be considered. The objective, to find a least-cost control strategy, demands that the greatest reduction in peak oxidant levels be found for any given expenditure on emissions abatement. For such a least-cost or cost-effective strategy, the reduction in ozone concentration per dollar spent on hydrocarbon abatement and the reduction in ozone per dollar spent on NO_x abatement should be the same. Equivalently, in a fashion familiar to economists, the "isopleths" of Figure 9 must be confronted with "isocost" contours, to ensure that the marginal rate of substitution between HC and NO_x that keeps peak ozone concentrations unchanged is the same as the marginal rate of substitution between HC and NO_x abatement that keeps total control costs unchanged. In terms of marginal conditions for the achievement of a least-cost control strategy:

$$\left.\frac{d(NO_x)}{d(HC)}\right/ dO_3=0 \qquad = \qquad -\frac{dC/d(NO_x)}{dC/d(HC)}$$

The left-hand term defines the slope of the ozone isopleth. That on the right is the ratio of the incremental abatement cost for NO_x to the incremental abatement cost for HC, the slope of the isocost contour. Attainment of such a least-cost abatement pattern is a necessary condition for any optimal strategy for ozone control, because any degree of ozone control that is desirable must be accomplished at least cost if the overall strategy is to be efficient.

A number of investigations of optimal oxidant control strategy undertaken in the past attempted to reach conclusions without adequate information on relative abatement costs, mainly on the basis of relative impacts of NO_x and ozone abatement.[17] This is not adequate, because, as shown below, the peculiarities of abatement costs play important parts in shaping the least-cost strategies. More complete analyses, employing both cost and impact estimates, have been carried out for the important problem area of Los Angeles, concluding that due to the sharply increasing marginal costs and limited effectiveness of NO_x controls, an efficient strategy would emphasize HC abatement.[18]

This is in sharp contrast to a general investigation of the oxidant problem in the Northeast, which concluded quite the opposite, that a control strategy should emphasize NO_x abatement.[19]