PN-AAT-963

East-West Environment and Policy Institute

Research Report No. 8

Economic Efficiency and Air Pollution Control

by Anthony C. Fisher

East-West Center
Honolulu, Hawaii

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PN-AAT-963

ISN-42526

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002712

Research Report No. 8 • September 1981
East-West Environment and Policy Institute

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Library of Congress Cataloging in Publication Data

Fisher, Anthony C.
Economic efficiency and air pollution control.

(Research report / East-West Environment and Policy Institute ; no. 8)
Bibliography: p.
1. Air quality management—United States—Costs.
2. Air quality management—United States—Finance.
3. Air quality management—Taxation—United States.
I. Title. II. Series: Research report [East-West Environment and Policy Institute (Honolulu, Hawaii)]; no. 8
HC110.A4F54 338.4'33637392'0973 81-17464
AACR2

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FOREWORD

The link between energy availability and economic growth has been the focus of much discussion during the last several dacades. More recently, the implications of energy conversion for environmental quality also have received increasing attention. The need to plan for a postpetroleum economy has provided additional impetus to such studies, since the commercial energy sources most likely to supply additional energy during the rest of this century are coal and uranium, both of which have more serious environmental problems. With these considerations in mind, the East-West Environment and Policy Institute has initiated a project on The Environmental Dimensions of Energy Policies. The major goal of the project is to provide policymakers with analyses that could be helpful in meeting the twin goals of energy supplies and a sustainable environment.

An area of high priority in the Asia-Pacific region, and within the project, has been the analysis of the links between air quality management and energy policies. A Workshop on that theme was held at the East-West Center in March 1980, with participation from nine countries in the region. A paper dealing with economic aspects of air pollution control was prepared by Anthony C. Fisher. Participants at the Workshop felt that the information in the paper would be useful to a wide audience. The Institute requested him to elaborate on his paper, which he kindly did. We feel that this product provides valuable insights dealing with issues of economic growth and air pollution, with the concepts and methods described applicable to other pollution types.

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Project Coordinator

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ABSTRACT

The question of how to deal with the problem of pollution—whether of the air or of the water—is controversial. Among the policy instruments available to control pollution are direct controls and economic incentives, such as taxes and subsidies. The policy instruments are evaluated and compared from an efficiency perspective, including a look at the Coase Theorem; the cost-effectiveness of a tax; a tax versus a subsidy; uniformity, spatial variation, and the administrative costs of a tax; and a tax versus marketable pollution permits. The advantages and disadvantages of a tax are compared to the other commonly suggested alternatives for controlling pollution—private bargaining, direct controls, a subsidy, and a permit auction system.

The methods available for determining benefits and costs in environmental decision making are examined also, with discussions of measuring impacts on vegetation and materials; evaluating impacts on human health; and the direct and indirect estimations of values.

In addition, a formal mathematical analysis is presented of the conditions required for economic efficiency in an economy in the presence of pollution. Although both the efficiency analysis and the description of benefit-cost evaluation methods refer to air pollution, the concepts and methods described are applicable to other pollution types. Extensive notes and references are included.

INTRODUCTION

Pollution—especially the air pollution associated with the mining, transport, and conversion of fossil fuel—generally is recognized as an important social problem. The question of how to deal with this problem is, however, a good deal more controversial. Should governments impose direct controls on the activities of polluters? Or should they rely on economic incentives, such as taxes and subsidies? This report looks at these and other policy instruments that have been proposed for dealing with pollution. For the most part, I shall be concerned with the *efficiency* properties of the alternatives. That is, can they achieve a balancing of the benefits and costs of pollution control? And what are the comparative costs of achieving a given degree of control?

Although a comparative analysis of pollution control instruments is a major focus of the report, the methods available for determining the benefits and costs of control also are discussed. And preceding both the comparative policy analysis and the discussion of benefits and costs is a somewhat more formal analysis of the conditions required for economic efficiency in an economy in the presence of pollution.

The plan of the report is as follows: First, a model is developed to determine the efficiency conditions (I also show how a tax on pollution can be used to bring them about); second, the advantages and disadvantages of a tax as compared to other commonly suggested alternatives for controlling pollution—private bargaining, direct controls, a subsidy, and a permit auction system—are explored; third, methods for determining benefits and costs are discussed; and finally, I consider the role of this sort of efficiency, or benefit-cost, analysis in environmental decision making.

Before proceeding with the formal model, it should be noted that it *is* rather formal, in terms of the mathematical methods used. The reader interested primarily in the strengths and weaknesses (from the economist's point of view) of the alternative control mechanisms, or in how benefits and costs of control can be estimated, can lightly skim the next section, and move quickly to the remaining sections where these topics are discussed. The more formal efficiency analysis is included only to provide a foundation for the later discussions.

Note also that, although both the example that motivates the efficiency analysis and the description of methods for evaluating benefits and costs refer to air pollution, concepts and methods will often be applicable to other types of pollution as well.

POLLUTION EXTERNALITIES AND ECONOMIC EFFICIENCY

The analysis will proceed in three steps. We shall derive, first, the conditions for an efficient allocation of resources in the presence of pollution externalities; second, the conditions for a market equilibrium; and third, the taxes required to make the two coincide. Following this, we consider a potential difficulty arising from the presence of a kind of nonconvexity. In the next section some further difficulties with the tax solution, or at least with achieving it in practice, are brought out and a number of alternatives examined.

The setting of the problem is as follows: The production of commodities by firms generates an air pollution externality—let us call it by the old-fashioned term "smoke"—that, in the aggregate, adversely affects each consumer. For convenience, we may think of the smoke generated by each firm as a factor of production for the firm, in the sense that it can be substituted for other (costly) inputs, such as labor and capital. For example, a given output can be produced by a process that involves the generation of 10 tons of smoke, or alternatively by one that, through the employment of a device that catches the smoke, generates just 5 tons. In either case, the smoke generated by the activities of all producers constitutes the externality, which then enters the utility functions of all consumers.

The externality is a pure public good—or "bad"; what one person "consumes" does not affect the amount available for consumption by others.[1] Though pollution is clearly a public good externality in this sense, equally clear is that it varies geographically; some areas are more polluted than others. We might say that the same aggregate emissions enter all utility functions, but the disutility suffered by any consumer depends also in part on his consumption of land, or in other words, on where he lives.[2]

Now let us state the problem formally. It is to maximize the utility of any one individual, subject to the restrictions that no one else is made worse off, and that the indicated outputs are feasible. The control variables are the consumption of each commodity by each individual and the production and input (including smoke) use by each firm. It is clearly not realistic to imagine a planner controlling directly the behavior of such a system down to the level of the consumption, by consumer j, of commodity i. We simply set up the problem in this form in order to determine (eventually) the value of a much less ambitious, and more realistic, control: a tax on pollution that makes a decentralized competitive equilibrium Pareto-optimal.

The problem, then, is:

maximize

$$u^1 (x_{11}, \ldots, x_{n1}, s) \qquad \ldots (1)$$

subject to

$$u^j (x_{1j}, \ldots, x_{nj}, s) \geq u^{j*} \qquad j = 2, \ldots, m \qquad \ldots (2)$$

$$f^k (y_{1k}, \ldots, y_{nk}, s_k) = 0, \qquad k = 1, \ldots, h \qquad \ldots (3)$$

and

$$\sum_{j=1}^{m} x_{ij} - \sum_{k=1}^{h} y_{ik} \leq r_i \qquad i = 1, \ldots, n \qquad \ldots (4)$$

where $u^j(\quad)$ is individual j's utility function; x_{ij} is the amount of good or resource i consumed by individual j; y_{ik} is the amount of good or resource i produced ($y_{ik} > 0$) or used ($y_{ik} < 0$) by firm k; r_i is the amount of resource i available, s_k is the smoke emitted by firm k; $s = \sum_k s_k$ is the smoke externality; and $f^k(\cdot)$ is firm k's production function.

What we have here is clearly a general equilibrium system, particularly if it is recognized that one of the goods or resources, x_{ij}, entering individual utility function can be leisure or labor. Although the analysis of externalities and optimal taxes has often proceeded in a partial equilibrium framework, the general equilibrium approach allows us to take account of important interdependencies. For example, as noted earlier, the impact of an externality will depend on the location decisions of individuals. These decisions and others that may influence the impact, such as whether trees are planted, or air conditioning is installed, and so on, are in principle part of the general equilibrium we are modeling. As we shall see later, the potential for adjustments like these, which would not be picked up in the ordinary analysis, may be important for policy. Note, however, that the model, as given in equations (1) – (4), does not explicitly reflect the interdependence implied by materials balance considerations.[3]

The model is also not dynamic. An alternative obviously would be to extend existing models of resource depletion to reflect environmental costs. But, in a sense, this is already implicit in those models, and making it explicit does not add much to our knowledge of the effects of pollution externalities or how to control them. In my judgment, the problems are essentially those of static misallocation. This is not to deny that pollution can accumulate—or be assimilated—over time, or that other dynamic processes might be relevant—for example, building a stock of control equipment. Interesting work has in fact been done that goes well beyond simply extending models of optimal depletion.[4] Where especially relevant, as for example to a choice among policy instruments, results will be indicated. But I continue to feel that the basic concepts—how do externalities arise; what are their optimal levels; how can a decentralized economy be controlled to bring these about?—can be elucidated without introducing the more complicated dynamics.

Now let us briefly indicate the salient features of each equation in our static general equilibrium model. The thing to note about the objective, consumer 1's utility function, is that it contains an argument, s, representing the externality. This same argument appears in the utility function of each consumer, as indicated in (2), the first constraint. This constraint says that the utility of each consumer other than the one whose utility is being maximized must be at least equal to some prespecified level (u^{j*} for consumer j). The second constraint, (3), is the set of production functions. The thing to note here is that s_k, the smoke emitted by firm k, appears in the firm's production function, where it is treated in effect as a factor of production. Finally, the third constraint, (4), is a general equilibrium condition. It says that no more of a commodity can be consumed, or a resource used, in the aggregate than is available to the economy.

The objective and constraints can be combined in the Lagrangian expression

$$L = u^1(\cdot) + \sum_{j=2}^{m} \lambda_j[-u^{j*} + u^j(\cdot)] + \sum_{k=1}^{h} \mu_k f^k(\cdot) \qquad \ldots (5)$$

$$+ \sum_{i=1}^{n} \omega_i \left(r_i - \sum_{j=1}^{m} x_{ij} + \sum_{k=1}^{h} y_{ik}\right).$$

Differentiating with respect to the x_{ij}, y_{ik}, and s_k, and assuming no corner solutions, we obtain the first order conditoins for a maximum

$$\lambda_j u_i^j - \omega_i = 0 \qquad \text{all } i, j \qquad \ldots (6)$$

$$-\mu_k f_i^k + \omega_i = 0 \qquad \text{all } i, k \qquad \ldots (7)$$

$$u_s^1 + \sum_{j=2}^{m} \lambda_j u_s^j - \mu_k f_{s_k}^k = 0 \qquad \text{all } k. \qquad \ldots (8)$$

The interesting result here is equation (8), which tells us that each firm should emit or employ smoke only to the point where the marginal benefit from doing so, the value of the marginal product of smoke, $\mu_k f_{s_k}^k$, is just equal to the marginal cost, literally the value of the weighted sum of marginal disutilities $u^1 + \sum_{j=2}^{m} \lambda_j u_s^j$. Since neither the disutilities nor the weights are observable, however, the result as stated may not be very useful. A little further analysis can yield one that is.

Let x_i be a good consumed by everyone. From (6), $\lambda_j = \omega_i / u_i^j$. The value of the marginal damage from pollution then becomes $\omega_i \sum_j u_s^j / u_i^j$. And as is well known, along an indifference curve between two goods, here pollution and x_i, the ratio of marginal utilities $u_s^j / u_i^j = -dx_{ij} / ds$, the marginal rate or substitution between the two. This leaves us with the value of damage equal to $\omega_i \sum_j (-dx_{ij} / ds)$, that is, the value of the x_i needed to offset an increment of pollution. If we further let x_i be the numeraire in this system, then the value of damage is just the amount of x_i needed, $\sum_j (-dx_{ij} / ds)$.[5] In any case, the value is at least observable in principle.

Now let us obtain the conditions that characterize a competitive equilibrium. By making the polluting firms subject to a tax, we then readily derive the optimal tax, that is, the tax required to make the competitive allocation Pareto-optimal. Almost as a by-product of this analysis we shall derive another result that sheds some light on an old controversy in the literature, concerning the compensation of victims. Many people have argued for compensation, which presumably could be paid out of the proceeds of the tax. Others have disagreed, on the grounds that it makes more sense to tax the "victim," since by his action—moving next to a smoky factory, for example—he increases the damage done by the smoke, and therefore the tax paid by the factory owner and, ultimately, the loss to owners of factor and to consumers of the factory's output. What we shall show is that the optimal compensation is either zero, or a lump sum that does not vary with the victim's actions and hence the damage he suffers.

Formally, the consumer's problem is to maximize his utility subject to a slightly unusual budget constraint. Expenditures are $\sum_{i=1}^{n'} p_i x_{ij}$, where p_i is the price of x_i, and $n' < n$. Income is $\sum_{i=n'}^{n} p_i x_{ij}$, where $x_{n'j}$ to x_{nj} are services *sold* by the consumer (there may be just one, labor). To this we add a term, t^j, as compensation for smoke damage suffered. The budget constraint then takes the form $\sum_{i=1}^{n'} p_i x_{ij} \leq \sum_{i=n'}^{n} p_i x_{ij} + t^j$ or, letting services sold be represented by negative x_{ij}'s,

$$\sum_{i=1}^{n} p_i x_{ij} \leq t^j. \qquad \ldots(9)$$

The Lagrangian expression for this problem is

$$L_j = u^j(\cdot) + a_j(t^j - \sum_i p_i x_{ij}). \qquad \ldots(10)$$

Differentiating with respect to the x_{ij}, and again ignoring corner solutions, we obtain

$$u_i^j + a_j(t_i^j - p_i) = 0. \qquad \ldots(11)$$

For the firm, the problem is to maximize profits subject to a production constraint. The only novel feature in this analysis is that the firm's profit function includes a term, $t_k s_k$, representing tax payments, at a per unit rate t_k, for the smoke it emits.

The Lagrangian expression then is

$$L_k = \sum_{i=1}^{n} p_i y_{ik} - t_k s_k - \beta_k f^k(\cdot). \qquad \ldots(12)$$

Differentiating with respect to the y_{ik} and s_k, and once again ignoring corner solutions, we obtain

$$p_i = \beta_k f_i^k = 0 \qquad \ldots(13)$$

and

$$-t_k - \beta_k f_{s_k}^k = 0. \qquad \ldots(14)$$

Comparing these conditions, and (11), to the corresponding ones for a Pareto-optimum, (6) – (8), it is clear that for them to coincide the following must hold:

$$p_i = \omega_i, \qquad \lambda_j = 1/\alpha_j, \quad \mu_k = \beta_k \qquad \ldots (15a)$$

$$t_i^j = 0 \qquad t_k = -u_s^1 - \sum_{j=2}^{m} \lambda_j u_s^j. \qquad \ldots (15b)$$

The interesting results are in (15b). Looking at the smoke tax, t_k, we see that it is *uniform,* that is, *the same for all firms* and *just equal to the value of the marginal damage from smoke at the Pareto-optimal smoke level.* From our earlier discussion of an observable expression for this value, the tax can also be written as

$$t_k = \sum_j dx_{ij} / ds. \qquad \ldots (15c)$$

Notice that the tax is not on output. It is sometimes suggested that the output of a good whose marginal social cost diverges from its marginal private cost, as would be true where smoke or other pollution is involved, ought to be reduced by means of a tax. Clearly, this is not correct. It is the smoke that is taxed optimally and reduced correspondingly, and if possibilities for substitution (away from smoke) in production are good, the effect on output may be negligible.[6]

The other result of interest here is that $t_i^j = 0$. This tells us that *compensation must not vary with changes in the victims' consumption levels.* Specifically, if they move next to a smoky factory, thereby suffering an increase in smoke damage, they should neither be compensated for this increase nor taxed to prevent it. *In other words, the compensation is not really compensation, in the sense of a compensating variation in income.* A lump-sum payment can of course be made, but this would not—indeed, must not—affect the allocation of resources.

Our first result, that a pollution tax ought to be set equal to the marginal damage from pollution, is generally well understood (apart from the confusion about whether the tax applies to the polluting product). Though most derivations are in a partial equilibrium setting and ours, along with a few others cited in note 3, is part of a general equilibrium, the intuition behind the result seems clear. This is probably less true for the no-compensation rule. Those who sympathize with pollution victims may be disturbed, and those who argue that the optimal compensation is in fact negative, that is, the victims ought to be taxed, may also feel let down.

Let us try to indicate why the result makes economic sense.[7] Consider an external *economy* that, like pollution, is also a public good in the sense that what one individual consumes does not reduce the amount available for others. Examples (assuming no congestion) might be a bridge crossing, or a scenic view—or, if one is fortunate enough to live in the San Francisco Bay area, the Golden Gate, which is both. If the external economy is not a gift of nature but must be produced, the same reasoning that established the optimality of a tax on a diseconomy suggests a subsidy to the producer.[8] What about a charge to the consumers, perhaps to cover the subsidy? Again assuming no congestion, the optimal charge is clearly zero. The reason is that any positive charge will lead to a reduction in consumption, when its marginal social cost is zero.

The case of the external diseconomy is exactly analagous. The producers should indeed be taxed, but the consumers should not be compensated, or at least not in proportion to their consumption. By inhaling smoke, consumer j does not provide a benefit to consumer j'—unless, of course, j' is a malevolent individual and derives satisfaction from j's ill fortune. But ignoring the possibility of a *consumption* externality of this type, no compensation is required. Moreover, just as a charge on consumption of the public good would lead to too little being consumed, compensation for damages from the public bad would tend to lead to too much being "consumed." If the potential victim were fully compensated for the damage he suffers by living next to the smoky factory, he would have no incentive to adjust his consumption behavior to reduce the damage, as for example by moving or by not locating there in the first place. Note, finally, that negative compensation—a tax—is equally unjustified. The victim absorbs the full social cost of his decision to live near the factory and needs no additional incentive to look elsewhere.

One important qualification to this discussion is that the public good or bad externality be excludable, in the sense that an individual can be excluded from consumption. Some public goods—national defense comes to mind—are nonexcludable, and this has sometimes been taken as a defining characteristic, along with nonrivalry in consumption (what one consumes does not reduce the amount available for others). I have carefully specified only that pollution exhibits nonrivalry. If it were completely nonexcludable as well, compensation could be justified. Suppose an individual has no real option of living away from a polluted area, and there are no other actions he can take to reduce substantially or eliminate the impact of the pollution. Then, compensation, which may be desirable for reasons of equity, would not impair allocative efficiency. The same reasoning of course applies to the external economy. If it were in fact completely nonexcludable, a charge would not lead to less being consumed; only the distribution of income would be affected.

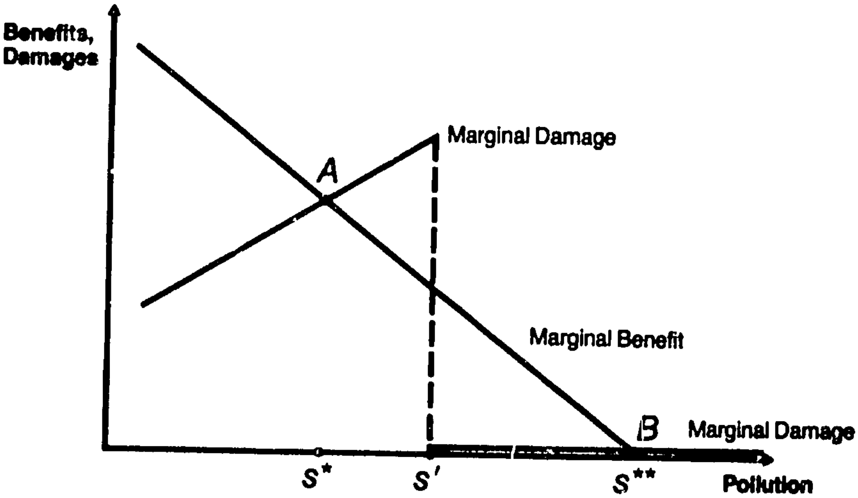

Figure 1a. Externality, nonconvexity, and multiple equilibria.

A Qualification: Externality and Nonconvexity

In the introduction I noted a different qualification to the optimal tax solution, related to the presence of a nonconvexity. The basic difficulty is that externalities can be associated with nonconvexities in affected preference or production sets, and these nonconvexities can lead to multiple tax equilibria. This sounds rather formiable, but I think the point can be made fairly simply with the aid of a diagram and some examples.[9]

Consider the case of individuals faced with increasing marginal damage from pollution. As our general equilibrium analysis suggests, they need not accept this indefinitely. They may instead take action to protect themselves by installing some sort of filtering system, for example; or by ceasing to use the contaminated medium where this is possible, as in the case of a polluted swimming place; or by moving away.[10] As a result, the marginal damage falls, perhaps to zero. The situation is represented in Figure 1a, where a well-behaved marginal produce, or benefit, of pollution curve is also shown.

The nonconvexity is introduced by the defensive action taken at the point where pollution reaches the concentration denoted by s' in the diagram. At this point the marginal damage curve drops sharply, to zero. As a result, two equilibria exist: at point A, and again at point B, where the marginal benefit curve reaches zero and again intersects the marginal damage curve, this time at a much higher concentration. Note that it is not necessary that marginal

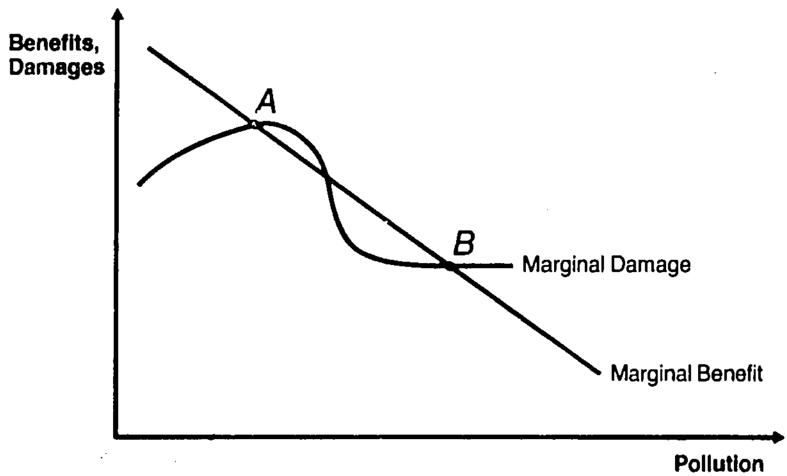

Figure 1b. Externality, nonconvexity, and multiple equilibria (many parties).

damage drop to zero; it need only fall far enough to intersect the marginal benefit curve a second time. Further, the drop need not be sharp. Suppose many individuals are affected, as in our model, and more important, as in the typical pollution case. Probably they would not all react to the increasing damage at precisely the same point, but as increasing numbers did so over some range of concentrations the sum of marginal damages would begin to fall. A situation like this, with the potential for a second equilibrium, is represented in Figure 1b. Note finally that, especially in this case, multiple equilibria cannot be ruled out.

I suggested earlier that the nonuniqueness resulting from general equilibrium adjustments may be important, for policy. To see why, consider the imposition of a tax set, as in equation (15), equal to marginal damage at the optimal point. Suppose the *ex ante* pollution concentration is at a point where marginal damage is still rising. On the somewhat simpler Figure 1a, this would mean at some $s < s'$. Then a tax t^*, set as indicated, will clearly lead to the A equilibrium, where $s=s^*$. If the *ex ante* $s>s^*$, the tax is greater than the marginal benefit and pollution accordingly is reduced. If the *ex ante* $s<s^*$, the tax is less than the marginal benefit and pollution is increased. The equilibrium is at $s = s^*$.

Now suppose the *ex ante* concentration is at $s > s'$. Here marginal damage has fallen to zero, and a tax that reflects this must lead to the B equilibrium, where $s = s^{**}$. For *ex ante* s between s' and s^{**} the optimal tax is just zero, and thus remains below the marginal benefit until $s = s^{**}$.

The problem this poses is that a pollution tax, or indeed any policy instrument based (appropriately) on marginal efficiency conditions, may produce an outcome that depends on pollution levels and related adjustments in force at the time it is imposed. Since damages generally have not been internalized (though this is changing), adjustments will have been made that result in low observed marginal damages. In other words, by consulting marginal conditions in the neighborhood of the *ex ante* point, which is probably all we can do, we are likely to end up at the high pollution *B* equilibrium rather than the low pollution *A* equilibrium. This may be globally optimal, but the point is we don't know. A benefit-cost analysis of the move from *A* to *B*, or vice versa, would be required to determine whether the likely local maximum at *B* is also a global maximum. The question is whether (on Figure 1a) the area under the marginal benefit curve from s^* to s^{**} exceeds the area under the marginal damage curve from s^* to s^{**}, or, as in this case where marginal damage falls to zero at s', from s^* to s'. The answer looks easy on paper, but an actual empirical analysis of the move back from *B* to *A* could be very difficult, because one would have to determine what adjustments had already been made or would be made if pollution loads were cut back.

POLLUTION CONTROL POLICIES: A COMPARATIVE ANALYSIS

We have just seen that a tax on pollution can lead to an optimal degree of control, though the potential for adjustments by victims can make attainment of a global optimum difficult. In fact, the other methods we shall discuss—direct control, subsidy, pollution rights market—face the same difficulty, so this is not necessarily an argument against a tax. Indeed, there are several advantages to a tax, as compared to those methods. In this section we shall be concerned primarily with the comparative strengths and weaknesses of the several alternatives. First, however, we consider a rather novel challenge to all. It has been raised by Coase (1960) specifically against a tax as the traditional remedy advocated by Pigou, but in fact it applies to all of the other forms of collective action as well.

The Coase Theorem: A Challenge to Pollution Policy

Coase's Theorem can be stated simply: with a clear definition of property rights, resources will be put to their highest valued (Pareto-optimal) use without any need for government intervention.[11] What does this have to do with pollution? Consider the case of a factory dumping wastes in a stream used also as a source of irrigation water by a farm. Suppose the farmer has no

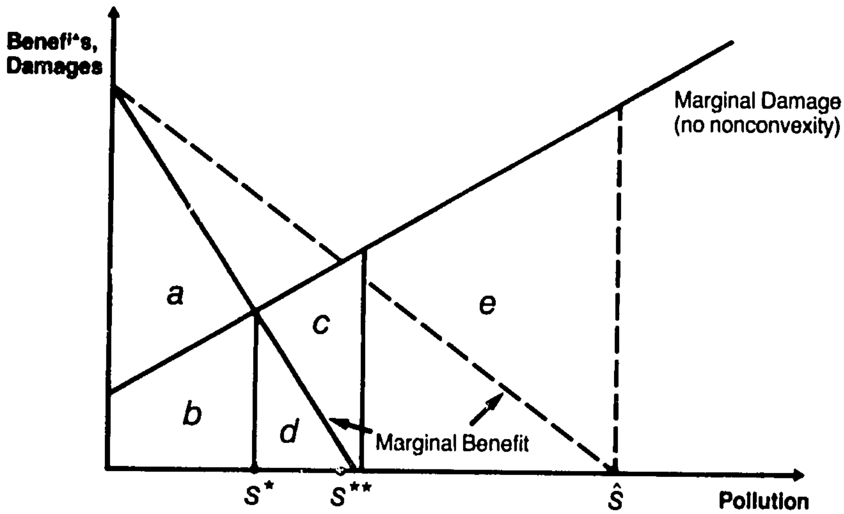

Figure 2. The Coase Theorem.

protected right to the water, and there is no law against dumping. The farmer presumably would be willing to pay the factory for each gallon of wastewater not discharged, as long as the payment is not greater than the marginal damage. The factory, for its part, would require a payment not less than the marginal benefit of dumping. The equilibrium payment then results in an amount of dumping that equates the marginal benefits to the marginal damage.

Now suppose the farmer enjoys a right to clean water from the stream. The factory would be willing to pay to discharge each gallon of wastewater as long as the payment does not exceed the saving. And the farmer would require a payment at least equal to the damage done by the discharge. Again, equilibrium comes where the marginal benefit from dumping equals the marginal damage.

This is shown in a slightly different way in Figure 2, an illustration of the theorem due to Turvey (1963). If the farmer is not entitled to clean water, he would be willing to pay, in total, an amount up to $c+d$ to secure a reduction in discharge to s^*, whereas the factory would cut back to this level for payment of anything over d. If the farmer does have rights, the factory would be willing to pay up to $a+b$ for the privilege of discharging s^*, and the farmer would accept the damage for a payment of anything over b.

We have established the following: that the allocation of resources will be the same regardless of the assignment of property rights; that the allocation will maximize the value of production: and that no intervention by government is required to achieve this result. In short, we have established the Coase Theorem. There are, however, a number of objections that can be raised to the assumptions needed to obtain this result and which, in my view, rob the theorem of any practical applicability to pollution problems. A question arises even as to whether the theorem is correct on its own terms.

In our example the only affected party was the farmer. But stream pollution ordinarily will affect many parties—other producers, like the farmer, and perhaps more important, consumers. Recreational opportunities will be diminished, there may be public health impacts, and so on. Thousands or even millions of people could be affected. Coase explicity assumes no transaction costs, which is realistic in the two-party setting of his examples—a rancher whose wandering cattle trample a farmer's crops, a confectioner whose machinery disturbs a doctor in an adjacent office, and so on. But in the typical many-party pollution case, the transaction costs will be prohibitive. All of the affected parties would have to be assembled and asked what they would be willing to pay or would require in compensation, depending on the assignment of property rights. Suppose the damage, in the aggregate, exceeded the benefit to the polluters from a projected increase in pollution. If the damaged parties did not have the right to clean water, the costs of getting together and negotiating a payment could be so high that it would not be done. The stream water would not go to its highest valued use, nor would this use be independent of the assignment of property rights.

Even if the barrier of transaction costs could be overcome somehow, another confronts a bargaining solution. Where many parties are involved, there will be an incentive for each to engage in strategic misrepresentation of preferences. Suppose, again, that damages exceed benefits and that the victims have no rights. Each will have an incentive to understate willingness to contribute to a bribe to the polluter, on the assumption that one portion will not appreciably affect the total. Yet, if enough people behave in this fashion, the total will indeed fall below the amount required to compensate the polluter, and once again stream water is allocated inefficiently. In other words, where the externality is a public good, as pollution normally is, the conditions required for the theorem to hold are simply not met.[12]

Questions have also been raised as to the validity of the theorem in a two-party setting. Let us return to our original example of factory and farm. Even here there seems to be scope for strategic behavior that would upset the Coasian equilibrium. The factory can claim that its marginal benefit curve, in Figure 2, is really farther to the right, say through point $\hat{s}$. Then the bribe it can extract from the farmer is increased, by an amount equal to e on the figure, and a new equilibrium, at s^{**}, is established. If the potential gains from this sort of behavior were large enough, one can imagine that real resources would be used (wastefully, from a social point of view) for the purpose of establishing a credible threat. The factory might, for example, at least begin to build a larger-than-needed effluent outfall in order to frighten the farmer into offering a larger bribe.[13]

Another problem for Coase is the presence of income effects, which can drive a wedge between the amount an individual is willing to pay for, say, clean water, and the amount he would require in compensation for loss of this good. In our example, and in Coase's, the two parties are producers, so this difficulty is not likely to arise. The loss to the farmer is measured unambiguously by the loss of output or the cost of obtaining clean water, whichever is less. But where the damaged party is a consumer—and this, we have argued, is the more typical case—willingness to pay may differ from required compensation because the former is constrained by the consumer's income. The result is that the assignment of property rights *will* affect resource use.[14]

In summary, then, it appears that the Coase Theorem fails as a challenge to pollution control policy involving some form of public intervention. It does offer an insight into the virtues of the market in dealing with certain kinds of externalities, but generally not those associated with pollution or other environmental disruption.

The Cost-Effectiveness of a Tax

Another kind of challenge to a pollution tax comes not from a somewhat narrow school of academic economists, as in the case of the Coase Theorem, but instead from noneconomists. The contention is that the information required to implement a tax—the marginal damage at the optimal point to all pollution receivers—is just not available. One implication is that neither a tax, nor the economic theory on which it rests, is very relevant to practical attempts to deal with pollution. Many economists accept, at least provisionally, the first part of this criticism, to the effect that we do not know enough about damage functions to design a tax to achieve full Pareto-optimality.[15] But these same economists have shown how a tax can be used to achieve the more modest, but still important, objective of cost-effective control.[16] That is, for any desired level of control, a tax will achieve it at least cost. We can view the problem as one of choosing, through the political process, a desired level of standard of environmental quality—much as we choose amounts of other public goods, such as national defense—and then seeking a method to achieve it at least cost. In what follows, we show that a tax will do this, and further that direct controls on emissions, a method favored by many noneconomists, probably will not. There are, however, some circumstances in which controls may be superior to a tax, or can usefully supplement it, as well shall indicate.

Our approach in proving the Cost-Minimization Theorem is similar to the one adopted in the preceding section. We first derive necessary conditions for achieving a preselected level of pollution at minimum cost and then show that the same conditions are satisfied by the decentralized decisions of polluting firms subject to an appropriate tax.

Formally, the planner's problem is to minimize the sum of expenditures on two kinds of inputs—those used to produce conventional goods and services and those used to control pollution—subject to restrictions on production, on the relation between production and pollution, and on pollution. Previously, we considered pollution as just another factor of production. This, of course, implied some expenditure on control, since less pollution meant more of other, costly inputs. Here, however, the expenditure is made explicit in order to obtain an expression for the indicated pollution tax in terms of the cost of pollution control. While this has some advantages in interpretation, and in comparing the costs of a tax with those of other methods, such as direct controls, it sacrifices some detail in modeling the role of pollution within the firm, as we shall see.[17]

In symbols, the problem is:

minimize

$$\sum_i \sum_k p_i r_{ik} + \sum_k p_v v_k \qquad \ldots(16)$$

subject to

$$f^k(r_{1k}, \ldots, r_{nk}) = y_k^*, \qquad k = 1, \ldots, m \qquad \ldots(17)$$

$$g^k(y_k^*, v_k) = s_k, \qquad k = 1, \ldots, m \qquad \ldots(18)$$

and

$$\sum_k s_k \leq s^*, \qquad \ldots(19)$$

where r_{ik} is the amount of input i; and v_k is the amount of control input v employed by firm k; p_v is the price of v; y_k^* is the output of firm k; $g^k(\cdot)$ is a function that relates smoke emissions to levels of output and control for each firm; s^* is the environmental quality standard; and other symbols are as before.

At least a couple of features of this model deserve further explanation. As indicated in (18), smoke emissions are determined by two things: the level of output and the input (v) devoted to abatement or control. This formulation is not as rigid as it may seem, since the control input can be understood rather broadly as a method or technique for reducing emissions in conjunction with physical factors like labor and capital. Just one such input is specified for simplicity without loss of generality.

A vector of outputs, the y_k^*, is specified because otherwise the problem is trivial. By having the firms produce nothing, or very little, the planner obiviously could minimize costs and satisfy the smoke constraint. What we are interested in are the conditions for minimizing costs associated with *any* given output, just as in the ordinary theory of the firm. The output actually selected will presumably depend on demand and on the planner's, or the firm's, objective. We assume only that it is desired to produce the chosen output at least cost and seek the conditions that will assure this. As before, we do not suppose that a planner really can determine input use at the firm level. We simply pose the problem in order to show how a much less ambitious approach, the setting of a (uniform) tax, can achieve the same results.

Proceeding with the solution, the Lagrangian expression can be written—first substituting $g^k(\cdot)$ directly for s_k — as

$$L = \sum_i \sum_k p_i r_{ik} + \sum_k p_v v_k + \sum_k \lambda_k [y_k^* - f^k(\cdot)] + \lambda(\sum_k g^k(\cdot) - s^*). \qquad \ldots (20)$$

Differentiating with respect to the r_{ik} and v_k, and assuming no corner solutions, we obtain the necessary conditions for a minimum

$$p_i - \lambda_k f_i^k = 0 \quad \text{all } i, k, \qquad \ldots (21)$$

$$p_v + \lambda g_v^k = 0 \quad \text{all } k. \qquad \ldots (22)$$

Now suppose the decisions on input levels will be made by the individual firms. The problem facing each is to minimize the sum of expenditures on inputs *and* a pollution tax, subject to the same restrictions on production and the relation between production and pollution. Note that our results will apply to imperfectly competitive firms as well, since we may assume they are interested in keeping costs down, however much they choose to produce.[18]

The firm's problem, then, is:
minimize

$$\sum_i p_i r_{ik} + p_v v_k + t_k s_k \qquad \ldots (23)$$

subject to (17) and (18). The Lagrangian expression—again substituting $g^k(\cdot)$ for s_k – is:

$$L^k = \sum_i p_i r_{ik} + p_v v_k + t_k g^k(\cdot) + \alpha_k [y_k^* - f^k(\cdot)] \qquad \ldots (24)$$

where t_k is the pollution tax. Differentiating with respect to the r_{ik} and v_k we obtain

$$p_i - \alpha_k f_i^k = 0 \quad \text{all } i \qquad \ldots (25)$$

and

$$p_v + t_k g_v^k = 0. \qquad \ldots (26)$$

Comparing these conditions to (21) and (22), it is clear they are the same, provided the tax t_k is set equal to λ, the shadow price of the pollution constraint, for all k.[19] λ clearly depends on the standard, s^*. For full efficiency, s^* would be set where the marginal damage for pollution just equals the marginal benefit, but this brings us back to the preceding section's approach, which we have since suggested is impaired by lack of information about damages.

Still, we have shown a great deal. Let us take stock. We have shown that a *uniform* tax on polluters ($t_k = \lambda$, all k) will achieve a preselected standard for environmental quality at minimum cost, provided the tax is set appropriately. Importantly, the result emerges from the decentralized decisions of the polluting firms. The central authority need know nothing about the control options facing each firm in setting the tax, and it need do nothing beyond setting the tax. On the other hand, to set the tax appropriately the authority must solve for λ, the change in the minimum expenditure on production and control associated with a small change in the pollution constraint. This is a kind of aggregate marginal cost of control and in practice might be estimated from knowledge of the costs of an "average" polluter.[20] Even where this is not feasible, however, a uniform tax has the desirable property of minimizing the cost of achieving *some* quality standard, and doing so in decentralized fashion.

To see this, consider the expression for the tax implicit in (26). Rewriting this to make the tax explicit, we have:

$$t = p_c \, / \, g_c^k \qquad \ldots (27)$$

The right hand side (rhs) is the price of the control input divided by its marginal product, or the marginal cost of control (the minus sign corrects for the negative g_c^k). Now, suppose the tax required to achieve a given quality standard, call it q^*, where q^* represents units of pollution *abated* and is related inversely to s^*, is not known. Instead, a tax is set that will in fact result in a different quality, q^{**}. The marginal cost of control will still be equated across sources of pollution, because each will push control to the point where the marginal cost equals the common tax. This is shown for two sources with different control costs in Figure 3. A tax t^* will achieve the desired quality level q^* at least cost, but a tax t^{**} will achieve q^{**} at least cost.

The advantage of a tax over direct controls on emissions is easily demonstrated in this format as well. Suppose the two sources in Figure 3 are producing the same amount of pollution before the tax or other control. Now it is desired to achieve a reduction to q^*. One obvious way to do this is to impose a uniform control on each source: a reduction of $q^*/2$. The difficulty is that, in general, this will result in violation of the cost-minimizing equimarginal outcome assured by the tax. As long as marginal costs differ,

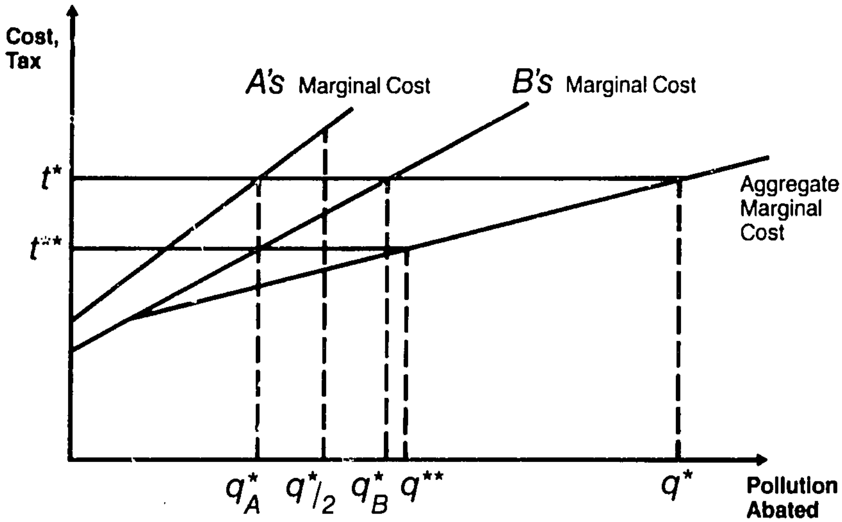

Figure 3. The minimum cost tax.

the cost of achieving q^* can be reduced by shifting a unit of abatement from the high-cost source to the low cost. Of course, a uniform reduction, which could also be stated in percentage terms for sources of different sizes, may have some appeal on grounds of equity. But it will almost certainly not be cost-effective.

Alternatively, the control could be tailored to the individual source to achieve the standard at least cost, as under the tax. In Figure 3, this would involve setting a standard of q_A^* for source A, and q_B^* for B. The difficulty here is that the central authority would have to know the control cost functions for *all* of the individual sources. Where there are only two, the difficulty may not be serious—though even in this case, the incentive to misrepresent would be very strong. And where there are a great many sources, it is just not realistic to imagine that the central authority could be informed about the types and costs of options available to each for controlling pollution.

Another advantage that has been claimed for a tax as opposed to direct controls is that the tax provides a continuing incentive to the polluter to cut back on emissions. No matter how low they already are, cutting back further will reduce tax payments. This may be especially important in a dynamic setting, where polluters are encouraged to seek new, low-cost ways of cutting back.[21]

A disadvantage of a tax is that extensive monitoring of emissions is required. Thus far we have tended to ignore the administrative costs of the policy alternatives. Yet it is clear, as noneconomists especially have argued in their attack on the feasibility of a tax, that the real resource costs of monitoring could be substantial.

A first response to this criticism is that it appears to apply to direct controls, and, for that matter, to other alternatives such as a subsidy or a permit system as well. Certainly this is true for controls on emissions, whether uniform or individually tailored. Monitoring costs may, however, be considerably lower for another form of control: a requirement that the polluter use a particular type of control technology. This is, in fact, a very popular approach in the management of both air and water quality in the United States. My feeling is that there is no reason to believe mandated technology will be cost-effective any more than other controls. Horror stories of almost perverse inefficiency in specific instances are common knowledge among students of environmental economics.[22] But technology controls do offer the advantage of reduced monitoring costs, and the trade-off may occasionally favor their use. I remain somewhat skeptical because the monitoring costs may not in fact be reduced all that much. As the history of mandated control devices on automobiles suggests, continuing inspection may be required to ensure that the devices are functioning properly, indeed that they are in place and functioning at all. Prospects are perhaps better in other areas, but it is hard to imagine a technology that does not require some monitoring. A fair conclusion here might be that the question of which approach to pollution control accomplishes a desired degree of control at least cost, including monitoring cost, is an empirical one. Cases in which mandating a technology will represent the least-cost alternative conceivably do exist.

There are a couple of other situations in which direct controls may improve on a tax or other policy instrument for protecting the environment. One is where the desired emission level is zero, as for example with highly toxic substances. In this situation a simple ban on use may be indicated.[23]

A second situation favoring controls is one of rapid or temporary variation in desired emission levels, for example, as a consequence of changing weather patterns. Taxes, subsidies, and the number of pollution permits sold can, of course, all be varied to meet changing emission targets. But this might be impractical over the short periods involved. Changing prices can be costly, which is presumably one reason why peak or time-of-day prices are not more widely employed. An in-place tax system, on air pollution for instance, could be supplemented usefully by direct controls on emissions in unusual circumstances, such as an atmospheric inversion that inhibits the dispersal of pollution.[24]

Tax Versus Subsidy

With the exception of the cases just discussed, a tax appears generally superior to direct controls. But a tax is not the only fiscal instrument that can be used to reduce pollution. Some economists have suggested that a subsidy, or payment to reduce pollution, will work just as well. In its strongest form, the suggestion is that resource allocation, including the emission of pollutants, does not depend on the assignment of environmental property rights. Whether the polluter is in fact paid for the emissions he controls, or taxed for those he does not, the outcome will be the same. Only the distribution of income is affected.

This may sound familiar and indeed has been called a Coasian position—though Coase considered mainly two-party situations and advocated direct negotiation between the parties as opposed to government intervention in the form of either a tax or a subsidy. Still, if we accept the proposition that *some* form of intervention is necessary in the typical large-numbers pollution case, the question of whether tax and subsidy are equivalent in their allocative effects, and if not, which is superior, seems legitimate. What I shall show is that they are not equivalent, and that the tax is superior, though there is a superficially plausible case for equivalence. The reasoning here is somewhat similar to that in our earlier analysis of the Coase Theorem and its application to pollution control.

Before proceeding, I should note that there is another kind of subsidy, one that is in fact a central feature of U.S. environmental policy. This is payment of part or all of the cost of pollution control. The payment can be direct, as in the case of federal grants to municipalities for the construction of wastewater treatment facilities, or indirect, as in the case of tax credits to firms for investment in certain types of control equipment. From the point of view of economic efficiency this kind of subsidy has serious drawbacks. These are considered after a discussion of the first, or Coasian subsidy.

The Coasian subsidy takes the following form. Starting from a benchmark level, the polluter is paid for each unit reduction in emissions. If the benchmark is s^*, actual emissions are s, and payment is at rate t, then the subsidy is $t(s^* - s)$. It is easy to see that this is just equivalent to a lump-sum transfer to the polluter, ts^*, coupled to a tax, $-ts$. Since behavior is presumably not affected by a lump-sum transfer, it appears that the allocative effects of a tax and a subsidy must be the same. Income distribution is affected of course by the disposition of the lump sum ts^*.

There are, however, at least two distinct difficulties with this result. One has been discussed already in connection with the Coase Theorem. Since the size of the lump-sum payment depends on the benchmark emission level, the polluter, or for that matter the *potential* polluter, has a strong incentive to misrepresent and even misallocate resources to establish a favorable benchmark. The fundamental difficulty is that the benchmark is set arbitrarily. Certainly one plausible way to do this—perhaps the only practical way—is on the basis of previous emission levels. But this creates an incentive for emissions above even what the firm would find profitable in the absence of any control for an interim period in which the benchmark is established. Moreover, setting the benchmark on the basis of observed emissions penalizes the clean firm, the one that has already installed control equipment or uses a less polluting process. It may be that an appropriate solution can be devised for determining a benchmark for each and every polluter, or potential polluter, but this is clearly not a trivial problem.[25]

A second reason for questioning the symmetry between tax and subsidy arises when the lump sum is considered more carefully. The difficulty is that, in the longer run, the lump sum *can* have an effect on the polluting firm's decisions. Because it has an effect on profits, it can influence the firm's decision as to whether to stay in business, or whether to enter a polluting business in the first place. Thus, even though a subsidy leads to a reduction in pollution by each polluter, just as a tax does, it will tend to increase the number of polluters and, correspondingly, the total amount of pollution. Over the longer run, when entry and exit are permitted, the allocative effects of a subsidy will not be the same as those of a tax.[26]

There is a qualification to this proposition, but it is not likely to be important in practice. Suppose the lump-sum payment is not made contingent on whether the firm that receives it remains in a polluting industry. That is, the firm would continue to receive the payment even if it left the industry, or shut down completely. Since this component of profit does not depend on any decision by the firm--even the decision as to whether to stay in business—the subsidy would not hold the firm in a polluting business.

The reason this is not likely to be important in practice is clear. It would simply not be feasible. The payment would have to go on indefinitely not only to the polluting firm that leaves the industry or shuts down, but also to the *potential* polluter. The objective is to keep firms from staying in or entering a polluting activity merely to qualify for the subsidy, and this requires indefinite payments to all in a position to do either.

Let us conclude the discussion of tax versus subsidy by examining briefly a different kind of subsidy. As noted earlier, current U.S. environmental policy features a direct or indirect payment by the government of a portion of the polluter's control costs. For example, the federal government now pays 75 percent of the construction costs of a municipal waste(water) treatment plant, up from about 50 percent in previous years. The difficulties with this arrangement are, first, that construction *and operation* of a plant is still a losing proposition for the municipality, and second, that the choice of control technology is biased.[27]

Unless 100 percent of the cost is paid, construction still entails a loss in revenue. If those who will benefit from the plant are largely in downstream jurisdictions, the incentive to build is weakened. Further, the incentive to operate the plant efficiently, indeed to operate it at all, is similarly weakened, since operating costs are borne entirely by the municipality. The point is, this kind of subsidy does nothing to create incentives for the efficient use of common-property water resources, as a tax or even a Coasian subsidy does.

The second objection to the subsidy as currently constituted is that it biases the choice of control technology. If capital costs are heavily subsidized and operating costs are not, one would expect capital-intensive methods of waste treatment to be popular. The results can be somewhat perverse. Current policy provides a subsidy in the form of tax credits to industrial polluters for the installation of certain types of control equipment. Recovery recycling of residuals do not qualify under this heading. Yet, in some cases at least, recycling represents the least-cost method of waste treatment.

Uniformity, Spatial Variation, and the Administrative Costs of a Tax

One of the advantages of a tax, whether designed for optimality or just cost-effectiveness in pollution control, is that it is uniform. Costly discrimination among polluters is not required to assure the Pareto-optimal or cost-effective outcome. When comparing a tax to direct controls, for example, we found that the same tax imposed on all polluters would lead to a given reduction in the total amount of pollution at least cost. In other words, the environmental authority need not tailor the tax to each polluter's individual circumstances. With direct controls, on the other hand, quotas would have to be determined based on individual control cost functions. The low administrative costs of a tax, in this respect, are one of its attractive features—though as we also saw in the comparison with direct controls, the costs of monitoring can be substantial.

But there is a problem with the uniform tax solution that casts doubt on the claim of low administrative costs. Consider two sources of pollution, one in an area where the capacity of the ambient environment to disperse or assimilate emissions is high, the other in an area where it is low. Should emissions from each really be taxed at the same rate? Intuitively, it seems the answer is no. The tax ought to be higher where emissions contribute more to pollution, to discourage polluters from locating there. This can in fact be demonstrated more formally, as I now show in the framework of our model of a cost-minimizing tax.

The only assumption in the model that needs to be changed is that emissions from individual sources are added together to produce "pollution." Instead, we shall assume that pollution is a *function*, not necessarily linear, of individual emissions. That is, where we previously defined pollution as aggregate emissions, $\sum_k s_k$, let us now define it as a function, $\phi(s_1, \ldots, s_k)$, of individual emissions. We require only that emissions by each firm contribute positively to pollution, that is, that $\delta\phi/\delta s_k > 0$, all k. Again, the point of this formulation is that it allows us to take account of differences among sources in the contribution of their emissions to pollution.

Constraint (19) now becomes

$$\phi(s_1, \ldots, s_k) \leq s^* \qquad \ldots (19)'$$

and the necessary condition (22) becomes

$$p_i + \lambda\phi_g k g_i^k = 0. \qquad \ldots (22)'$$

The other necessary conditions, including (26), are not affected, so that the tax on firm k, t_k, must be set equal to $\lambda\phi_g k$, which is obviously not the same in general as the tax on firm k', $t_{k'} = \lambda\phi_g k'$. The tax on emissions by each source, in other words, is no longer uniform, and is instead weighted by the contribution of emissions by that source, $\phi_g k$, to pollution.[28]

How significant is this modification to the comparative assessment of pollution taxes? Clearly, if there are a large number of sources in a region, and something like our $\phi_g k$ term must be assessed for each, a tax loses some of its appeal. A practical solution to the dilemma might be to make a fairly broad cut at discriminating among sources. In the simplest case, for example, just two classes of sources might be defined—those characterized by high assimilative capacity of the receiving medium, and those characterized by low—and a uniform tax set within each. The study of taxes versus direct controls on water pollution in the Delaware estuary (note 16) represents a considerably more ambitious approach, in that it also distinguishes between a uniform tax and one that varies by zone, for some 30 different zones. The additional flexibility introduced by this variation does have an impact on control costs, though the major impact is still produced by the move from uniform direct controls (equal percentage reductions) to a uniform tax. In other words, fairly substantial spatial differentiation appears to be computationally feasible and would yield a savings in control costs but even without this a tax is much superior to direct controls.

Note that we are in any case not talking about evaluating the *damages* from pollution, or indeed even determining them. The environmental authority would only need to know something of the influence of emissions from each source on aggregate pollution levels. Moreover, to the extent this information must be taken into account in setting a tax, it is equally relevant in determining the values of other policy instruments, such as direct controls or a subsidy. The case for a tax, then, does not appear to be seriously weakened by the complications introduced by spatial variation. A uniform tax still leads to substantial savings (if the Delaware study is at all representative), further savings can be had by varying the tax in a realistic way, and the complications are, in any case, complications for the use of other instruments as well.

Still another such instrument, the sale of pollution permits or rights, is sometimes advocated as being superior to a tax on several grounds, including the ability to deal with spatial variation. In the remainder of this section, I consider the relative merits of tax and pollution rights schemes.

Tax Versus Pollution Rights: Price Versus Quantity Rationing

In principle, a tax and a rights auction ought to lead to the same result. The tax is set to cut emissions to some desired level, whereas the auction sells rights to produce the same emissions. In either case, polluters have an incentive to pursue controls to the point where the cost reaches the price they would pay for polluting. But a number of economists have suggested that the rights auction might have some advantages in practice.[29]

One alleged advantage, as just indicated, is a superior ability to deal with spatial variation. The idea is that fewer permits would be auctioned in "bad" areas. Alternatively, of course, the tax could be set higher in such areas, but Baumol and Oates (1979) argue that this sort of discrimination would be politically difficult. Note that both the number of permits and the tax could also be manipulated to shift the *time* distribution of emissions. I suggested earlier that this would not be practical for short periods, such as those associated with atmospheric inversions. But for longer periods, such as a season, it might well be. In any event, I find it difficult to choose between tax and auction on the basis of the political difficulty of spatial variation. Perhaps Baumol and Oates are right, but it is not clear to me why, if polluters are going to complain about paying a higher tax price than their competitors in other areas, they will not complain about being offered fewer rights.

Another alleged advantage of an auction is its superior ability to achieve the desired degree of control. We saw earlier that, to achieve this, the environmental authority must know something of the aggregate control cost function. Where this knowledge is lacking, there is the risk that the target will not be achieved, in particular that too much pollution will result. The situation is represented in Figure 4. Suppose the target is q^*. If the environmental authority believes marginal control cost are approximately MC_1, the appropriate tax is t_1. But if marginal control costs are really more like MC_2, then only $q' < q^*$ will be achieved. Baumol and Oates suggest this is one

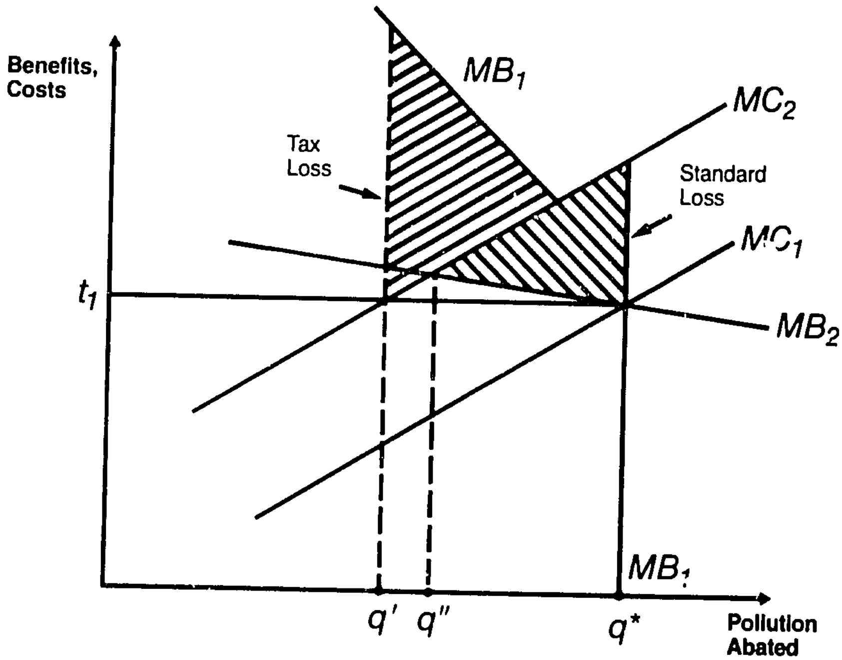

Figure 4. Tax and standard compared.

reason why taxes, though recommended persistently by economists, are viewed with skepticism by policymakers. Of course, a tax is not set in concrete. If it does not achieve the desired objective, it can be moved around until it does. Still, changing the tax, especially raising it, could be politically difficult, and there is also the question of *ex ante* and *ex post* control costs. The initial tax presumably will lead to investments in control. Once these investments are in place, the cost of adjusting them in response to a change in the tax could be substantial.[30]

The skepticism of policymakers—and some economists—may be well founded, then. On the other hand, setting a standard and sticking by it carries a risk of its own. The costs of compliance could reach unacceptable levels. This possibility is also illustrated in Figure 4. Suppose, again, the target is q^*, set because the environmental authority believes marginal control costs are in the neighborhood of MC_1. If they are really nearer MC_2, achieving the target will entail substantially higher costs, which may imply unacceptable sacrifices of other social objectives.[31]

It appears, then, that either a tax or a standard can be set, and with either one society runs the risk of much larger than anticipated losses in environmental amenities or other goods and services. The source of the difficulty, along with the control cost uncertainty, is that neither tax nor standard is set with regard to the relationship between costs and benefits in the framework we have adopted. It follows that some knowledge of benefits may be helpful. The question is, what kind of (imperfect) knowledge can in fact be helpful?

Suppose we have reason to believe that marginal damages from the pollution in question rise sharply at some point, or, in other words, that marginal benefits from control fall sharply. Then the marginal benefit curve would look very much like MB_1 in Figure 4, which becomes inelastic at around q^*. In this case, the environmental authority ought to auction off rights just sufficent to attain q^*, rather than take a chance on a tax that could lead to an inefficiently low level of environmental quality, if control costs have been underestimated. On the figure, tax t_1 based on a cost (under) estimate of MC_1 results in large losses, as measured by the area between curves MB_1 and MC_2 from q^1 to q^*.

Now suppose the marginal benefit function is believed to be quite elastic, like MB_2 in the figure. Again estimating control costs as MC_1, the environmental authority sets a standard q^*. If costs are really MC_2, losses are once again incurred, measured by the area between curves MC_2 and MB_2 from q' to q^*. This time, however, the losses result not from too much pollution, rather from "too little," in the sense that more is being spent on control than it is worth.

To sum up, where the marginal control cost curve is uncertain, knowledge of the *shape* of the marginal benefit curve can be helpful in choosing between a pollution tax and a standard-and-auction approach to avoid the risk of large efficiency losses. An inelastic benefit curve would favor a standard, which it resembles, whereas an elastic curve would favor a tax, which *it* resembles.[12]

Whether it is realistic to expect that an environmental body will have at its disposal even the limited knowledge of benefits called for in this approach, I do not know. But in view of the potential for very large losses if it does not, research to determine whether, or where, benefit curves exhibit sharp drops (or damage curves sharp rises) similarly has a potential for a large payoff. Lacking such knowledge, the choice of tax or standard might simply be based on avoiding what appears to the decision maker to be the larger risk. Where there is concern that environmental quality reach at least a certain minimal level, for example, the standard-and-auction approach seems indicated. Where the concern is more for the possibly excessive costs of reaching a standard, on the other hand, a tax is appropriate.

Thus far, a case has not been made, in my judgment, for the *general* superiority of a pollution rights auction to a tax. Either might be varied for cost-effectiveness, where time and politics permit. And uncertainty about control costs can cut in favor of either one, depending, as we have just seen, on the nature of benefits. But two considerations from outside the realm of static efficiency analysis do seem to pose special difficulties for a tax.[13]

In a growing economy, tax rates would have to be adjusted frequently to maintain a desired quality of the environment. With a rights market, the price of a right to pollute would rise automatically, that is, without government intervention. As the demand for rights increases, this *should* be reflected in a higher price, just as for other scarce resources. A tax can be adjusted to reflect this, but the point is that the rights market will do so automatically.

A closely related argument concerns the effect of inflation on environmental quality under the two regimes. Again, without frequent adjustment of rates, quality will be eroded inadvertently under a tax. A permit system, however would maintain quality, while the price of a permit or right simply shares in the general inflationary rise. In a dynamic setting, then, where growth and inflation may be significant, a rights auction is likely to do a better job of protecting the environment than a tax. Still, we probably should not overlook entirely the advantage of a tax in holding the line on costs.

POLLUTION DAMAGES AND CONTROL COSTS

In order to use effectively any of the instruments for pollution control that we have just described, something of the damage done by pollution must be known. And the more ambitious the target, the more must be known. This section is about methods for assessing damages, or, as we should put it where a change for the better is under consideration, the benefits of control. Some attention is also given to the relatively more straightforward, though still challenging, problem of assessing the costs of control. Rather than simply presenting a bewildering variety of results from literally hundreds of very diverse empirical studies, I shall stress some of the more important and interesting theoretical issues that arise in the formulation and interpretation of these studies. Some key results are also presented. The discussion will be especially relevant to air pollution, because the theory and practice of damage estimation has been mainly directed to this. It should be obvious, as we go along, where the discussion applies also to other types of pollution, or related disamenities such as noise.

Damage Estimation

To understand how damages are estimated, it will be helpful to place them in a larger framework. This is done in Figure 5. Starting on the left on the figure, the pattern of economic activity in a region leads to a pattern of residuals discharge—so many tons of particulates emitted to the atmosphere, so many gallons of raw or treated sewage dumped into streams, and so on.[34]

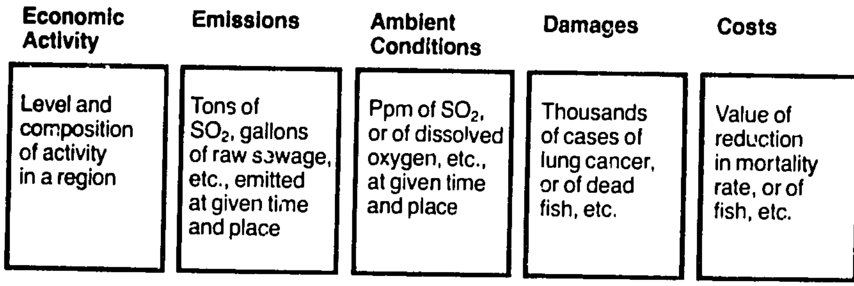

Figure 5. Steps in going from activity to costs.

These waste residuals move through the receiving medium, possibly undergoing some physical or chemical transformation in the process, and appear in concentrations varying with the time and distance from the source of the discharge.[35] Ambient concentrations in turn produce physical damages—crop loss, increased human deaths, and so on.[36]

Our problem is to *evaluate* the damages. One way to do this is obviously to first determine the physical magnitudes and them impute a value to each. An alternative, somewhat neater way, if it can be done, is to infer values directly from pollutant concentrations. This avoids the risk, in the first method, of failing to capture all of the separate effects. For example, some of the disutility of pollution is clearly aesthetic. Yet the aesthetic damage is hard to measure. What are the appropriate units? Alternatively, aesthetic damage will be reflected in the value of a location specific private good, such as a house in a polluted area. Other things equal, we would expect a house in a polluted area to sell for less than one in an unpolluted area, and the difference is just the value of damage, inlcuding aesthetic damage.

Actually assessing values is more complicated than this suggests and will ordinarily require a combination of methods. In the current, fairly primitive state of our knowledge, it appears that some effects, such as aesthetic losses, and perhaps materials and some vegetation damage, can be better evaluated by means of a sophisticated version of the comparison of property values just described. Risks to human health, on the other hand, may not be captured in this fashion, at least in part because the risks are not accurately perceived. A separate assessment of health damage would be required.

In summary, then, there are two methods of evaluating damages. The first, a two-step method, first measures separate physical effects of pollution and then imputes a value to each. The second estimates a relationship directly between ambient concentrations and a measure of value, ordinarily residential property value. I shall discuss each briefly.[37]

Measurement of Damages, Imputation of Values: Impacts on Vegetation and Materials

In principle, valuation of nonhuman impacts, such as those on livestock, crops, commercial marine life, and so on, seems straightforward. The observed loss in units of biomass is simply multiplied by the per unit price to obtain a measure of value. Something like this has indeed been done in countless studies of local impacts of particular pollutants, and the results may be reasonably accurate. There are pitfalls even here, however, suggested by economic and econometric theory.

In the first place, how is the loss "observed"? Two methods are available: statistical field study, in which actual crop yields, say, are statistically related to a variety of influences including differences in pollutant concentrations; and controlled dose-response experiments, in which the effect of a substance on a laboratory specimen is studied. An obvious difficulty with the statistical approach is the presence of other influences on yield. Suppose one or more of these is related also to pollution. If they are left out of the regression equation, the estimated relationship between pollution and yield will be biased. If they are included, all the estimated coefficients are tainted by multicollinearity, which reduces the likelihood that precise estimates of the effects of particular pollutants will be identified. It is also difficult to disentangle the effects of different types of pollution, some of which tend to appear in concert, and which may act synergistically.

Another pitfall in interpreting the statistical results is suggested by our theoretical analysis of the general equilibrium adjustments to pollution. For example, instead of suffering heavy crop damage, a farmer might plant a less valuable, but more pollution-resistant strain, and in so doing limit the damage. The real loss from pollution in this case is the reduction in new crop yield plus the difference in value between old and new crops, but only the former would tend to be captured in the statistical analysis.[38]

Fortunately, in the case of nonhuman impacts, such potentially incomplete or biased results can be supplemented by laboratory experiment. Thus, damage to the original crop could be studied in a controlled environment. But note that this would tend to produce an *over*estimate of the loss from pollution, since possibilities for defensive adjustments are ignored.

Whether biomass and materials losses are estimated from statistical field studies or dose-response experiments, or perhaps, best of all, from a mixture of both, the problem of imputing values remains. Although market price is the obvious measure, at least a couple of rather subtle pitfalls must be avoided. One is the effect of a pollution-induced quantity change on price. If the quantity change is substantial, and demand is inelastic, market price could be affected. Further, in a general equilibrium system other prices will in turn be affected—for commodities related in consumption, and for factors of production. This is a potentially troublesome issue, since the price changes imply in each case changes in consumers' or producers' surpluses. Clearly, the researcher must hope price effects can be safely ignored, and some evidence suggests they can.[39]

A different problem is presented by effects of pollution other than simple reductions in yield. Substantial evidence exists that both the quality of crops is also changed, generally for the worse, and that vegetation is made more susceptible to damage by insects and disease. The compounding effect probably cannot be ignored. A study by the Stanford Research Institute (1973) estimates the value of annual damage to vegetation from air pollution in the United States at US$134 million, but another study suggests that taking the indirect damages into account would put the figure at more than US$1 *billion.*[40]

It appears, then, that even the relatively straightforward task of valuing the nonhuman impacts of pollution must proceed with a great deal of care, with an eye on pitfalls suggested by economic and statistical theory. In saying this, I certainly do not wish to give the impression that results obtained to date are not significant. On the contrary, taken together, the hundreds of statistical and experimental studies clearly document large and costly impacts on vegetation, on (commercial) marine life, on materials, and so on. But challenging theoretical issues must be faced in refining and interpreting the results. My impression is that actual damages are probably substantially *greater* than even the studies suggest, for two reasons. First, they would tend to be based on postadjustment, high-pollution equilibria, where the bulk of the damage may be invisible. Second, many of the effects of pollution, including synergistic effects such as lowering the resistance of vegetation to pest attack, are not yet well understood.

Evaluating Impacts on Human Health

Lack of knowledge is a problem especially for a class of effects we have not yet discussed—effects on human health. Measurement and evaluation here run into all of the difficulties already noted, and then some. For example, one reason it is hard to estimate the effect of pollution on human health is that controlled experiments cannot be carried out in the same way they can on plants or mice. The researcher must then rely almost exclusively on statistical regression analyses of public health data. There has been a great deal of work in this area, probably the best known (to economists, at least) being the careful and comprehensive statistical analyses of the relationship between air pollution and human health by Lave and Seskin (1970, 1977). The results are not free of controversy, but I think it is fair to say that Lave and Seskin, and others, have demonstrated that there *is* a relationship between the main stationary source pollutants, sulfates and particulates, and human mortality rates.[11]

But the most difficult aspect of evaluating the damage done by pollution to human health is not determining the extent of the damage. Rather, it is imputing a value. For impacts on commercial plant and animal species, and on materials, market prices can serve as measures of value, subject to the qualifications noted. When it comes to evaluating changes in human mortality rates, however, the researcher is confronted with the lack of a measure of value, a willingness to pay analogous to the price for a bushel of wheat or a pound of shrimp. A number of indirect methods for valuing lives have accordingly been suggested. In my judgment none is entirely satisfactory, but let us briefly review them.

At the outset, it ought to be clear that we are talking "statistical" life, as opposed to the life of a known individual. Obviously I would be willing to pay (if I had it) an infinite amount to prevent my certain loss of life tomorrow. And there is considerable evidence that society is similarly willing to go to enormous expense to save or prolong the life of a known individual. But this is not germane to the evaluation of pollution damages. What is to be evaluted in this case is not the certain loss of life of a known individual, but rather a relatively modest increase in the *probability* of loss of life for each individual member of a larger population at risk: in short, statistical life. It is clear that individuals and governments routinely make choices that involve trading off money, time, or other goods for small changes in the probability of loss of life. The methods we shall discuss seek in one way or another to infer, from these tradeoffs, the value of statistical life.

A commonly suggested source of information about this value is expenditure on public programs to save lives. From data on expenditures and lives saved it is possible to calculate the expenditure per life saved, which might be assumed the value attached by society to a statistical life. There are problems, however. Most important, the procedure is circular. The relevant value, instead of being determined by analytical methods and then given to the political process, to use as it chooses in assessing and deciding on programs, is itself extracted from the political process. Thus, one is simply looking at the outcomes of past decisions and feeding them back into current assessment. Not surprisingly, since the decisions generally have not reflected any sort of optimization, a very wide range of values (expenditures per life saved) has been observed, spanning three orders of magnitude (see Table 1).[12]

On the other hand, in the few cases where public agencies have adopted an explicit benefit-cost framework for making these decisions, the values are just those calculated by other methods, so here too, inferring value from the political process is circular.[13]

Table 1. Estimates from Assorted Sources for the Value of Saving a Statistical Life and Averting Associated Illness and Disability

Source of Evidence	Estimated Value (US$ thousands)	Reference
Human capital		
Discounted future earnings plus total medical costs	$89	Cooper and Rice (1976)
Surveys		
Willingness to pay for emergency coronary care	28–43	Acton (1973)
Willingness to pay for flight on airline with better safety record	5,000	Jones-Lee (1976)
Political process		
Office of Science and Technology	140	U.S. OST (1972)
National Academy of Sciences	200	NAS (1974)
Federal Highway Administration	250	Hapgood (1979)
National Highway Traffic Safety Administration	287	Hapgood (1979)
U.S. Air Force	270–4,500	Usher (1973)
Occupational Safety and Health Administration	1,900–625,000	Bailey (1978)
Consumer Product Safety Commission	240–1,920	Bailey (1978)
Labor Market		
Extra wages of workers in risky occupations	136–260	Thaler and Rosen (1975)
Extra wages of workers in risky industries	1,500–5,000	R.S. Smith (1974 and 1976)
Extra wages for underground miners	68–318	Usher (1973)
Hazard pay for pilots	161	Usher (1973)
Other evidences		
Seat belts and time preference	160–551	Blomquist (1977)

SOURCE: Hamilton (1979).

Perhaps the most common approach, and the one in fact taken by Lave and Seskin in valuing their estimated health effects, is the "human capital" approach. The idea is that the death of an individual causes losses to society in the form of both medical costs and foregone future contributions to the national product, the latter measured by the individual's wage or salary. One difficulty with this approach is its failure to capture losses in the form of pain and suffering, by the affected individual *and* those who care for him. The failure is particularly serious where the individual is not in the labor force.

A more basic difficulty is that foregone earnings do not provide information about what an individual would be willing to pay to obtain a given reduction in the probability of death, which is after all what we are interested in. For example, suppose I am offered a safer widget, one that will reduce the probability of my suffering a fatal accident during its use from, say, 0.01 to 0.0001, that is, by a factor of 100. The human capital approach implies that I would be willing to 1 percent of the present value of my future earnings for this opportunity. Yet, I might, depending on my preferences, be willing to pay a good deal more than this.[14] The human capital approach thus appears to be conservative, likely to underestimate the value of statistical life. It may be useful, as a lower bound, where no better information is available.

If willingness to pay is the measure of value, why can't we simply ask people what they would be willing to pay for a product or program carrying a specified reduction in probability of loss of life? I am aware of three or four such surveys, and results vary widely (see Table 1). There are, in addition, the usual reasons for concern about the accuracy of responses to hypothetical questions, and about distortions due to strategic behavior by the respondents.

The final approach I shall discuss also focuses, correctly, on willingness to pay, but on the basis of observed behavior generally in the labor market. People routinely make choices about jobs carrying different degrees of risk. This approach seeks to infer the value attached to an increment of risk of loss of life from the resulting pattern of wage differences. The method used is statistical regression analysis of wages on a variety of influences, such as age, education, region, and of course degree of risk. The estimated risk coefficient then gives a measure of the extra compensation required for the individual to bear extra risk, or his willingness to pay for reduced risk.[15]

In principle, this is an appropriate method for valuing impacts on health, because it seeks the right value—willingness to pay for a reduction in risk—and does so on the basis of observed behavior. In practice, there are a number of difficulties. In the first place, much of the modern theory of the labor market questions the assumptions of perfect mobility and of perfect competition required for observed wage differences to reflect faithfully attitudes toward risk. For example, if mobility is restricted, wages will not be bid up to attract or hold workers to a risky job.

Second, the wrong attitudes may be reflected, for purposes of evaluating effects of pollution on health. People who take risky jobs probably do require some compensation for bearing the extra risk, but less than the average person affected by pollution would require for bearing the same risk from the pollution. The risk in a risky job is often quite glamorous, but there is nothing glamorous about the risk of sickening and dying from air pollution. Again, observed wage differences would underestimate willingness to pay for a reduction in risk from pollution.

Finally, it must be assumed that workers correctly perceive risks. For example, with risks of latent development of cancers from prolonged exposure to certain industrial materials only now coming to light, it is not likely that they have been accurately perceived by the workers. For this reason too, wage differences would underestimate the value of statistical life. Misperception of risk could, of course, cut in either direction; workers might be unduly concerned about the risk of exposure to a substance that they would in fact be effectively shielded from, or that would turn out to be relatively harmless.

In raising these questions about the labor market approach, I do not wish to deny its potential usefulness. Again, I believe it is appropriate in principle. But it needs to be used with care, and with an eye on qualifications suggested by labor market theory. For example, wage-risk differences *within* occupations probably would be superior to differences *between* occupations, since the former are not impaired by restrictions on mobility.[10]

Some estimates of the value of statistical life from one or another kind of labor market evidence are presented in Table 1, along with the human capital and government expenditure estimates. Note also an estimate based, correctly, on observed willingness to pay for reduced risk in a different situation. In Table 2, a few estimates of the value of pollution damages are presented. Note that since the damage to health is valued on the basis of the human capital method, the figures in the table are lower bounds.

Discussion of Empirical Results

A number of tentative conclusions can be drawn from the results reported in both tables. With respect to the value of statistical life (Table 1), the human capital value does indeed generally fall below the value estimated from labor market and other observed behavior. One would, therefore, certainly not be guilty of overvaluing life in employing the human capital figure. Further, since even the labor market figures tend to be biased downward, they are probably preferable, as furnishing a tighter lower bound on the true value. A commonly suggested central tendency for the labor market value is in the neighborhood of US$300,000 (1979 dollars). The

Table 2. Selected Estimates of U.S. Air and Water Pollution Damages

Type of Damage	Value (Annual US$billions)	Source
Stationary source air pollution	$10.8 ($4.3 health, $1.1 materials, $5.4 aesthetics and soiling.)	Waddell (1974) for U.S. Environmental Protection Agency
Automotive air pollution	5	National Academy of Sciences (1974)
Air pollution: health benefits of 58% abatement of particulates, 88% abatement of sulfates, consistent with 1979 compliance with 1970 Clean Air Act Amendments	16.1 (1973)	Lave and Seskin (1977)
Air pollution damage to vegetation	2.9	Heintz, Hershaft, and Horak (1976) for U.S. Environmental Protection Agency
Water pollution: benefits of Clean Water Act Amendments of 1972	5.5 by 1985	National Commission on Water Quality (1976)
Water pollution	10.1 (60% due to loss of recreation opportunities, 17% due to production losses)	Heintz, Hershaft, and Horak (1976) for U.S. Environmental Protection Agency

results in Table 2 are very sketchy. The health damages are probably more firmly established and a good deal larger—even though they are underestimates—than damages to vegetation or structures. Note also that adjustments for inflation would increase all figures somewhat.

An interesting question, in view of the motivation for this whole discussion, is whether the calculated values tell us anything about pollution control policies. Specifically, we might ask whether suggested ambient standards for particular pollutants are justified on efficiency grounds. To answer this, we of course need to know something of the costs of attaining the standards. In one case at least, that of air pollution from sulfates and particulates, there appears to be sufficient information about both costs and benefits. Lave and Seskin use U.S. Environmental Protection Agency estimates of the costs and come up with a total in the neighborhood of US$9.5 billion (1973 dollars). This is compared to their estimate of US$16.1 billion in benefits, again in 1973 dollars, from the same standards. The standards are justified, then, in a rough way, especially if we bear in mind that only health benefits have been included, and probably conservatively. Further calculations would be required to determine "optimal" standards, those that would result in marginal benefits just equal to marginal costs.

Direct Estimation of Values: Pollution and Property Values

An alternative to the two-step, piecemeal approach to estimating values is to estimate them directly as a function of differences in ambient concentrations. As noted earlier in this section, this is normally done by relating differences in land or property values to differences in air pollution levels. Well over a dozen studies of this type have been carried out over the last decade.[47] Results are hard to characterize with precision, because quite different measures of the key variables, pollution and property values, have been used, and the data are drawn from different times and places.[48] But it is probably fair to say that the existence of a relationship between air pollution and residential property values, at least, has been demonstrated.[49]

One of the potentially very attractive features of this approach is that, in principle, it captures all of the separate effects of pollution—on aesthetics, on health, on materials, and so on. As noted earlier, however, it seems doubtful that health effects, at least, are reflected in residential property values, because they probably have not been accurately perceived.

Another difficulty, which this approach shares with all of the examples of statistical estimation we have discussed, is the presence of other variables that may bias the estimate. Clearly, land values are affected by a variety of factors aside from pollution. And we cannot look to experimental data to disentangle all the effects of pollution, as we can, for example, when attempting to infer its effect on, say, crop yields.

But there is a positive side to the story, which deserves further discussion here because it is both important and special to the property-value method. Researchers believed originally that, to estimate the damages from pollution—or as we shall say here in conformity with the literature, the benefits from a reduction in pollution—the change in property values that would result from the reduction would have to be predicted. If correct, this raises the question of how to account for general equilibrium adjustments to property values everywhere in the system. Even assuming no prices were affected outside the area experiencing the reduction, as could be the case if the area were sufficiently small, the supply of low-pollution sites would have increased, and

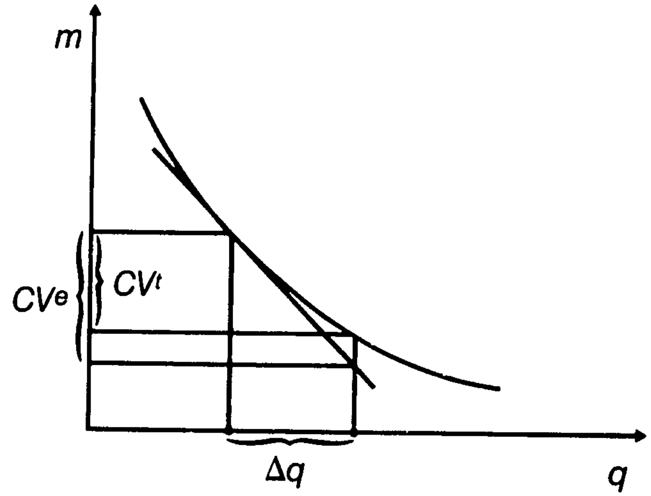

Figure 6. Compensating variation measure of the value of an environmental improvement.

the price of such sites presumably decreased. And, if outside prices were affected, demand for the improved sites would also shift, influencing price in an undetermined direction.

Fortunately, it can be shown that prediction of a new set of property values—even for the directly affected sites—is not required to estimate benefits.[50] There is sufficient information in the existing property-value-pollution relationship to infer a correct, compensating variation measure of the benefits of an improvement. We show this by proceeding indirectly, through the relationship between income and a reduction in pollution, or an improvement in environmental quality.

Figure 6 displays a consumer's indifference curve for a numeraire, income net of land rent (where rent is the amount paid per period for the site, a flow measure related to the site's capital value by an appropriate discount factor), and environmental quality. The numeraire represents an aggregate private good. For a marginal change in quality, dq, the compensating variation is the change in net income, dm, which would keep the consumer on the same indifference curve. For a sufficiently small change, this is approximately by the slope of the tangent to the curve at the appropriate point.

There is a qualification, easily demonstrated on the figure. Suppose we are considering a nonmarginal change, say Δq. The true compensating variation, read from the indifference curve, is CV^t. But if the compensating variation is computed from a point estimate of the income-quality relation-

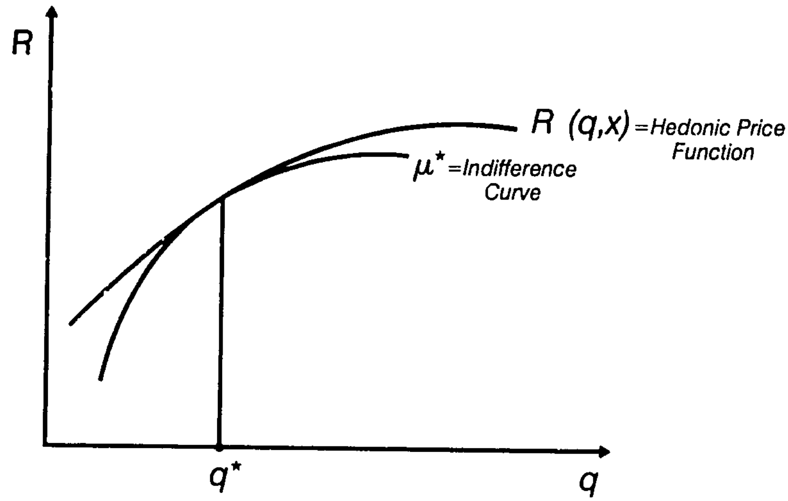

Figure 7. Hedonic price function equilibrium.

ship, such as the slope of the tangent line, an overestimate, CV^e on the figure, will result. For a nonmarginal improvement, then, a technique such as the one we are about to discuss, based on a point estimate, will yield an upper bound to the value of the improvement. Conversely, the value of a nonmarginal *deterioration* in quality, the amount of the numeraire that would be required in compensation, will be *underestimated.*

Now let us redefine the indifference curve in Figure 6 in terms of land rent R, instead of the numeraire $m = Y - R$ (where Y is income). The new curve is a mirror image of the old one, as indicated on Figure 7. Next, we draw in an opportunity locus for the individual, that describes the relationship between land rent and environmental quality, keeping constant other site characteristics that might influence rent. This relationship—between the price of a site and its characteristics—is often called a hedonic price function.[51] Although we would normally expect the partial relationship between rent and quality to be positive, as indicated on the figure, noncorner solution requires only that some indifference curves lie below the rent-quality locus.

Where is the equilibrium, then? Clearly, at the point of tangency where quality is q^*. Any other point on the opportunity locus yields inferior utility. And points on indifference curves to the right of the one shown, though preferable, are unattainable.

The value of a change in quality (around q^*) is then given by the slope of the tangent line at q^*, which is just the value of derivative of the opportunity locus, or hedonic price function, at q^*. The value of a change that affects several sites—as any conceivable change in the public good, environmental quality, will—is the sum of the individual site values. The full empirical procedure is to (1) estimate, by statistical regression techniques, the hedonic price function, in particular the relationship between price and the relevant measure of environmental quality; (2) take the derivative; (3) multiply it by the change in quality for each site; and (4) sum the result over all affected sites. This yields a measure of benefits (for an improvement) or costs (for a deterioration) directly from the relationship between quality and property values, without the need for an intermediate determination of the physical consequences of the change in quality. Nor is there any need to predict the new equilibrium configuration of rents or property values.

An important qualification, noted earlier, is that the consequences be perceived accurately by those making the location decisions. To the extent they are not, as is almost certainly true for at least some consequences to health, separate estimates would be required to capture the full value of a change.

One other qualification, or perhaps we should call it an assumption needed for the procedure to yield sensible results, is that the area experiencing the change be "open," that is, that there be no restrictions on mobility. Suppose pollution is decreased in an area. This represents a consumers' surplus benefit to residents, but the benefit will not be capitalized into property values unless there is some mechanism to transfer the surplus from residents to property owners. Competition from potential in-migrants from the improved sites normally would do this. Where there are barriers to entry, however—and note that even significant costs of migration would fall into this category—some part of the surplus may not be captured in rents and property values. In this case, the estimated property value–pollution relationship will be biased downward.

There is another potential source of (downward) bias of considerable theoretical interest. Thus far, we—along with most of the researchers who have studied the relationship between pollution and property values—have ignored the role of wage differences. This is not unreasonable. Within a single urban labor market, the type of area that has been studied, differences in pollution levels cannot be reflected in differences in wage compensation. Subject to the qualifications noted, only rent provides a site-specific measure of value related to pollution. On the other hand, it seems plausible that individuals might be attracted to a polluted area in a *different* labor market by higher wages there.[12]

The question is whether the compensation required to hold an individual at a polluted site comes in the form of lower rents, higher wages, or both. There are a few empirical studies of the relationship between wages and environmental quality across urban areas, but they do not really address this question, any more than do the more numerous studies of the relationship between property values and quality within urban areas.[53] The very recent theoretical analyses of Freeman (1979) and Scotchmer (1979) suggest that differences in both rents *and* wages will contribute to the required compensation, over a broad range of conditions.

Clearly, the econometric problems involved in an attempt to disentangle and identify both components of value would be formidable. This is probably one good reason why no such study exists, to my knowledge.[54] Yet, to the extent that wage differences are relevant, the value of a change in quality will be underestimated by an approach that takes into account only differences in intraurban rents or property values. Still a further source of downward bias, even if wage differences are appropriately counted, is the existence of cost or other barriers to labor mobility, exactly as in the property-value estimation.

We have identified a number of theoretical pitfalls—sources of bias that have nothing to do with econometric or data problems—in using comparative property values to infer environmental values.[55] But let me reaffirm the usefulness of this approach. It is rooted in economic theory. It depends on observed behavior. And each of the difficulties we have identified can be characterized as leading unambiguously to an under- or overestimate, usually an overestimate, of the environmental value at stake. Where a deterioration in quality is concerned, all effects are unambiguously negative. An estimated relationship between quality and property values can be interpreted as a lower bound, subject to the identification of other, conflicting, sources of bias. Where an improvement is concerned, if it is nonmarginal, the direction of bias is theoretically indeterminate, though all but one of the identified sources would lead to an underestimate of the value. In an actual case, the researcher well might have sufficient feel for the data to at least determine the direction of bias.[56]

If one is nevertheless unsatisfied with this and all of the other approaches considered thus far, there remains the possibility of simply asking people what an improvement in quality would be worth to them. The difficulties with surveys here are the same as noted briefly in connection with surveys designed to elicit information about the value of life. First, people may not know how to respond to a hypothetical question. Second, they will ordinarily have an incentive to behave strategically, to not reveal the "truth," even if they know what it is. Still, given the difficulties with the alternative approaches, the use of surveys ought not to be rejected out of hand. And a number of clever schemes, designed to elicit honest responses, have been suggested for valuing different kinds of public goods—though only a couple are specifically directed to valuing pollution abatement.[57]

Before moving on to discuss the estimation of abatement costs, I should acknowledge that the discussion of damages has neglected the question of how they are distributed. Since policy decisions will be importantly affected by this, it is clear that empirical studies ought to try to develop information about the distribution of damages, or the benefits from abatement, as well as the costs, along with information about the magnitudes. Though the literature has not always addressed the distributional impacts, several studies of these have been made, especially with reference to air pollution.[58] I shall return to the role of damage estimation, and benefit-cost analysis in environmental decision making, in my conclusion.

Control Cost Estimation

Control costs are those entailed by changing in some respect the pattern of economic activity that gives rise to pollution. For example, a polluting firm might invest in waste treatment facilities, relocate, or change its product mix—or pursue some combination of these and still other measures. Whatever it does, the consequences will show up on the firm's balance sheet in dollars and cents. As such, they are much easier to grasp, and certainly to evaluate, than the damages done by pollution. This is, as we noted earlier, one reason why some environmental economists prefer to focus on the administrative and control costs associated with the alternatives (taxes, subsidies, etc.) for achieving a reduction in pollution specified without regard to the value of damages.

In order to determine these costs, it helps to have a theory or model of the way a polluter will respond to, say, a tax. We have outlined such a theory in earlier portions of this study, but this was done for the purpose of drawing some qualitative conclusions—about the optimal tax, about the cost of reduction under a tax as opposed to other policy instruments, and so on. Here we are interested more in the detailed modeling of adjustments of the sort mentioned just above—investment in treatment facilities, changes in input and product mixes, and so on.

Such modeling has in fact been done, especially for water pollution. One approach taken is extension of the neoclassical (smooth isoguant) model of the firm to include decisions about how, and how much, to reduce pollution in response to one or another kind of charge. Within this framework, pollution has been considered as both an input to production, along the lines of our optimal tax model (see Pollution Externalities and Economic Efficiency) and a by-product amenable to treatment, somewhat along the lines of our cost-effective tax model (see The Cost-Effectiveness of a Tax).[59] The other approach taken is a still more detailed engineering–economic analysis of discrete process options, at the plant level, for responding to a charge or

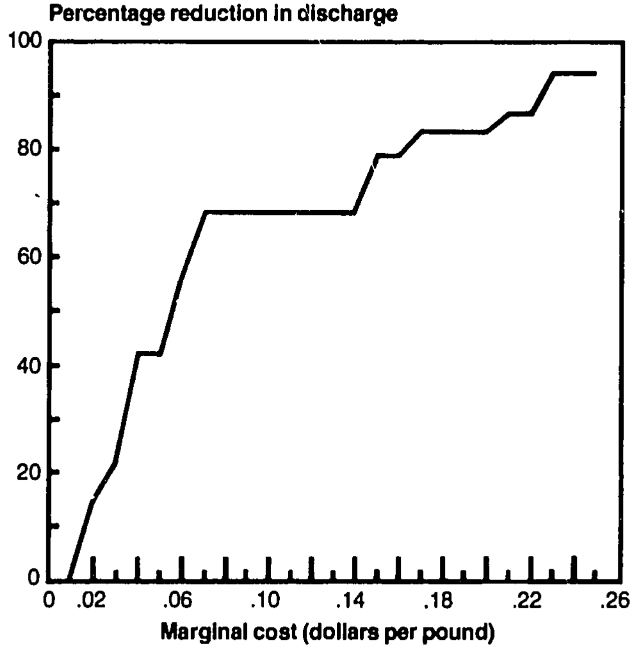

Figure 8. Marginal cost of BOD discharge reduction in petroleum refining (Russell 1973).

other control on pollution in a given industry. In the more recent applications, a formal optimizing procedure, linear programming (LP), is often used to select the options and their levels.[60]

Whatever the underlying model, the key question is: How can the costs be estimated in an actual case? Here the engineering–economic process model has an advantage, in that it is already in a computational format. The effect on a cost or profit function of a tax or other constraint on pollution is readily determined in an LP model. For such an abstract representation of a production process to yield usable results, though, obviously a great deal of very detailed technical information is required. The difficulty in acquiring this information may be compounded by the fact that some of it will be proprietary.

An alternative way of proceeding in these circumstances, indicated in any case to give empirical content to the neoclassical model, is by means of statistical regression analysis of industry data. The idea here is to estimate changes in inputs, outputs, and costs of production in response to a tax or some other control on pollution.[61] Much of the interesting detail of the LP-

Table 3. Cost of Water Pollution Abatement under Taxes and Direct Controls, Delaware Estuary, United States (US$ millions/per year)

Dissolved Oxygen (ppm)	Program		
	Uniform Treatment Controls	Uniform Tax	Zone Tax
2	5.0	2.4	2.4
3–4	20.0	12.0	8.6

SOURCE: Kneese (1977).

process model is lost, of course, but it may not have been available to begin with, and the econometric model may do reasonably well in tracing the movements of broad aggregates. Indeed, econometric models have been used to predict the effects of environmental policies on the broadest aggregates: gross national product, the price level, unemployment, and so on.[62]

Several results stand out from the many and varied studies. First, there is in most cases considerable scope for reducing pollution, and by a variety of methods in addition to "end-of-pipe" treatment of wastes. A detailed look at the alternatives for a number of important industries in the United States, including pulp and paper, petroleum refining, steel, and coal-electric energy, is provided in Kneese and Bower (1979), a review of work by Resources for the Future in this area.[63]

A second important point that emerges from much of this same work, however, is that, beyond a point, the marginal cost of control rises steeply. Fortunately, this generally occurs at high levels of control. For example, as shown in Figure 8, the marginal cost of BOD discharge reduction in petroleum refining begins to rise steeply only after a 70 percent reduction has already been achieved.

A third finding is that a given reduction in aggregate discharges, or improvement in environmental quality in a region, can be brought about more cheaply by a tax on the discharges than by uniform direct controls on them. This is shown in Table 3 for water pollution in the Delaware estuary. To achieve, for example, a 3–4 ppm level of dissolved oxygen in the water, the cost would be US$20 million annually under uniform treatment controls (each source reducing discharges by the same percentage). A uniform tax on discharges would accomplish the same result, but at a cost of just US$12 million. Finally, a tax that varied by zone, over 30 zones along the river, would produce the cleanup at a cost of US$8.6 million. All of this is, of course, consistent with our theoretical discussion of the cost-effectiveness of a tax.

Table 4. Impact of a Pollution Control Policy on Macroeconomic Variables, Expressed as the Percentage Difference Between the Economy Without the Policy (BASE or FULL) and with the Policy (CEQ or HC), 1976–1983 (percentage)

Macroeconomic Variables	Years							
	1976	1977	1978	1979	1980	1981	1982	1983
Real GNP								
BASE-CEQ	0.09	-0.48	-1.03	-1.16	-1.42	-1.70	-1.97	-2.17
BASE-HC	0.14	-0.59	-1.28	-1.40	-1.73	-2.09	-2.44	-2.68
FULL-CEQ	0.11	-0.53	-0.93	-1.16	-1.41	-1.74	-1.95	-2.27
Consumer price index								
BASE-CEQ	1.56	2.26	2.72	3.17	3.64	4.05	4.47	4.71
BASE-HC	1.82	2.74	3.40	3.90	4.53	5.03	5.59	5.94
FULL-CEQ	1.54	2.32	2.78	3.39	3.84	4.41	4.77	5.34
Growth rate of consumer price index								
BASE-CEQ	0.7	1.1	1.0	1.03	0.8	0.7	0.7	0.7
BASE-HC	0.9	1.4	1.2	1.0	1.1	1.1	0.9	0.9
FULL-CEQ	0.8	1.2	0.9	0.9	0.9	0.8	0.8	0.8
Unemployment rate								
BASE-CEQ	-5.56	-7.35	-2.41	-2.02	-1.15	0.00	1.64	3.64
BASE-HC	-8.33	-10.29	-3.61	-3.03	-2.30	-1.43	0.00	1.82
FULL-CEQ	-5.48	-7.94	-3.64	-2.13	-2.27	0.00		4.55

SOURCE: Evans (1973).

A final interesting empirical result is that the macroeconomic effects of current U.S. environmental policies—and also those of at least a couple of other countries for which studies have been done—are likely to be relatively modest. That is, the studies do not lend support to either of two extreme positions that have been advanced in the political debate about environmental policies: (1) that current policies will lead to sizable reductions in output, or rises in prices, as opponents claim; or (2) that they will greatly stimulate employment, as proponents claim. Projections from the Chase macroeconometric model for the United States (Evans 1973) are shown in Table 4. The figures of the table indicate percentage deviations from a baseline projection of the economy without existing environmental regulations. The deviations are not negligible, but neither are they dramatic. Output and prices are adversely, though modestly, affected. Employment is stimulated presumably by investment in the needed control equipment but falls back toward the end of the forecast period once the equipment is in place and growth has slowed. The moral of this story, I think, is that environmental policy need not be overly influenced by macroeconomic considerations, though some coordination with stabilization policy, and perhaps assistance to adversely affected areas, is certainly appropriate.

CONCLUSION: THE ROLE OF BENEFIT-COST ANALYSIS IN ENVIRONMENTAL DECISION MAKING

Following the discussion of the estimation of pollution damages, we noted that the question of their distribution is not generally addressed in the empirical literature—though several studies have been made, especially for air pollution (see note 58). Again, given the importance of this question in the policy arena, it seems clear that future studies ought to try to develop information about how damages—and the costs of control—are distributed across relevant groups in the population.

Another issue we touched on briefly, and one often raised against the benefit-cost approach to policy and analysis, is whether environmental impacts are in fact capable of being evaluated. The preceding section's discussion has dealt with particular methods of evaluation and the difficulties as well as the promise, attached to each. But let us now consider the issue more generally.

In doing an empirical study we might, for example, estimate readily the value of pollution damage to crops or livestock, but what about risk to human life? Perhaps this is indeed impossible to value. In any case, a study of pollution damages should certainly report such crucial physical impacts as, say, an expected increase in human mortality rates. But before we reject any attempt at evaluation, we ought to recognize that it is in fact carried out routinely by individuals—in choice of transport mode, of neighborhood, of job, and so on. In each of these and other everyday situations, money or time or both are traded off for a reduction in risk. The values implied by these trade-offs are precisely the ones we seek.

Government agencies, in deciding on programs that can affect human health—and other sensitive elements of the environment—necessarily make "value" judgments, as indeed they should. I would suggest only that these judgments are likely to be better, in the sense of getting closer to efficiency in resource allocation, if they are informed by estimates of the values individuals themselves place on things that affect their health and well-being. This is not solely a matter of academic concern. In a world where environmental standards and related programs increasingly need to be capable of passing muster at cost- and efficiency-minded agencies such as (in the United States) the Office of Management and Budget, and the Council on

Wage and Price Stability, serious attention to commensurate measures of value does not seem out of place.

Further, systematic consideration of the benefits and costs of a project need not involve quantitative estimation of *all* of these, to be useful in decision making. Suppose we find that just the readily estimated losses due to adverse impacts on the environment exceed the gains. Then we need not worry about our inability to evaluate more elusive damages. Of course, it is important to indicate the unevaluated damages in a qualitative way, to assure that the quantitative estimate is indeed a lower bound.

Finally, difficult questions are raised by two other often-related objections to benefit-cost analysis of environmental decisions: that the evaluation techniques do not deal adequately with intergenerational impacts, and that they do not deal adequately with uncertainties about impacts. The intergenerational problem can be viewed as a rather intractable form of the distribution problem. The difficulty is that future generations are not around to register their preferences, nor can they be readily compensated for damages suffered as a result of decisions taken in the present. Where future costs—and benefits—of such decisions are appropriately incorporated into the evaluation procedure, the objection often takes the form of disgreement with the discount rate used to reduce these future values to present values.

One way of dealing with differing views about the discount rate, hence the weight accorded future impacts, is to examine the effects of varying the rate. Where there is uncertainty or controversy about the magnitude of an important parameter, such as the discount rate, this sort of sensitivity analysis is particularly appropriate. Less formally, information about the distribution of benefits and costs over time is likely to be relevant to a political decision and ought to be included in the evaluation of a proposed environmental standard or policy. The suggestion here is just the same as the one for dealing with concerns about contemporaneous distributions. Nothing in the methods used to evaluate impacts precludes presenting the findings in some richness of detail.

The uncertainty objection is obviously related to the one that claims some impacts cannot be evaluated. It is also related to the future generations problem, since more distant events ordinarily would be less certain. Once again, I would agree that stochasticity, in nature or in the economy, may make the information contained in a single number, such as the expected value of a benefit or cost, inadequate as the sole input to a decision. And once again, I would suggest that attention be given to higher moments of a distribution where relevant. If an energy technology, for example, exhibits some probability, however small, of a catastrophic impact on the environment, surely this is relevant to a decision on how to regulate it or set standards for its use and ought to be included as part of a complete evaluation. This can be done informally by simply presenting the information, or more formally by folding it into models of decision making under uncertainty. The sensitivity of results to variations that span the range of uncertainty about the key influences can and should also be examined.

Obviously, a great deal more could be said about each of these objections to evaluation of environmental impacts. My purpose in raising them here has been simply to indicate that environmental economists generally are aware of them, that they have merit in some cases, but that they are not fatal to evaluation. On the contrary, where valid they call for the development of supplementary information, about distributions of costs and benefits, about elements of probability distributions, about the results of sensitivity analyses, and so on.

But I would also argue that even a more restricted benefit-cost analysis can play a role in environmental decision making. How big a role? Clearly, this is a question that can be answered only by the concerned decision maker, and only in a given set of circumstances. One possible guideline, which I put forward in a tentative way, is the following. Where the decision in question is "small," for example, whether to set an ambient standard for a pollutant at x ppm or at $(x+\Delta x)$ ppm, or whether to set a tax on emissions at \$$y$ or \$$(y+\Delta y)$ per pound, a fairly straightforward consideration of benefits and costs may suffice. In such a case the analysis is trying to substitute for the market where the market has failed to do something it ordinarily does well: setting a price based on the interaction of demand (benefit) and supply (cost).

Where, on the other hand, the decision is "big," for example, whether or not to proceed with development of nuclear power, considerations of intergenerational equity, of the potential for low-probability catastrophic events, and so on, may loom larger than considerations of simple efficiency in resource use. Even in this case, though, efficiency is not irrelevant—just as equity may not be irrelevant in setting an emissions charge.

NOTES

1. In addition to the classic article by Samuelson (1954), see also Head (1962) for a discussion of the attributes of public goods.
2. Problems for pollution control policy raised by spatial variation in pollution concentrations are considered later in Pollution Control Policies: A Comparative Analysis.
3. The model developed and used here is based fairly closely on one in the volume on the theory of environmental policy by Baumol and Oates (1975a), though these are differences. Other general equilibrium models include Ayres and Kneese (1969); Kneese, Ayres, and d'Arge (1970); Meyer (1969); Tietenberg (1973, 1974a); Page (1973a); and Mäler (1974). Kneese and his collaborators do take account of materials balance, but not substitution in production, including substitution of other factors for pollution. Maler's analysis is a good deal more abstract than the others, employing the methods of algebraic topology now standard in the general equilibrium literature. More recently, models combining general equilibrium and dynamic features have been developed (see Gruver 1976; Comolli 1977; and Forster 1977). Dynamic models are discussed later and in the next note.
4. The accumulation of waste over time is introduced in a highly aggregated materials balance model that includes resource extraction, by d'Arge (1972) and d'Arge and Kogiku (1973). Several other dynamic models of waste accumulation have also been developed, though these do not always include extraction and full materials balance. See Keeler, Spence, and Zeckhauser (1972); Plourde (1972); V.L. Smith (1972); and Mäler (1974). Maler's analysis does account explicitly for materials balance. These dynamic models are, in essence, optimal growth models extended to consider the residuals of pollution generated by consumption. As in the case of optimal growth models with an extractive resource constraint, the key question is whether a steady state exists. And again, substitution possibilities, here for pollution, are clearly decisive. The question, in other words, is whether and at what rate pollution per unit of output, and also pollution accumulations, can be reduced. Other questions, relating to the composition of investment and of output and to the stability of a tax-adjustment scheme, are also treated by Gruver (1976), Forster (1977), and Comolli (1977), respectively.
5. Notice that this is just Samuelson's (1954) condition for the optimal supply of a public good: the marginal cost is equated to the sum of marginal rates of substitution between the good and a numeraire private good. In this case, of course, the good is a bad, pollution, so it is

the marginal *benefit* from its use that is equated to the sum of (positive) marginal rates of substitution.

6. The distinction between a tax on pollution, as an input to production, and a tax on output, was made by Plott (1966), who showed that if pollution were an inferior input it would be increased by a tax on output.
7. The discussion here, like the result, is drawn from Baumol and Oates (1975a). See also Page (1973) and Mäler (1974).
8. Our framework does not explicitly allow for public production, but as pointed out by Kneese and his collaborators, the optimal provision of a public good externality may require this, along with fiscal incentives for individuals. In the case of pollution control, public investment in treatment facilities can complement a tax on polluters. The optimal mix of these control elements is studied by Bohm (1972a).
9. The view of nonconvexity developed here is based on that of Starrett and Zeckhauser (1974). A more rigorous, abstract analysis is presented by Starrett (1972). Other treatments of the connection between externality and nonconvexity include Portes (1970), Kolm (1971), Baumol and Bradford (1972), Baumol and Oates (1975a), Kohn and Aucamp (1976), and Gould (1977).
10. These and other alternatives are emphasized, under the general heading of "averting behavior," by Zeckhauser and Fisher (1976). Averting behavior is simply an aspect of the general equilibrium adjustment of an economy to a disturbance, such as an increase in pollution.
11. Coase's original article is much richer in detail than this suggests, and there is a bit more to the theorem. Coase may indeed have been the first to emphasize the potential for the kind of averting behavior or adjustment to externality we discussed in the preceding section. For a very clear presentation of Coase's analysis, as well as extensions and criticisms, see Randall (1972) and Page (1973a).
12. This argument—that publicness and the large numbers associated with it make the Coase Theorem inapplicable—was developed originally by Wellisz (1964), and by Kneese (1964) with special reference to water pollution. Schulze and d'Arge (1974) provide a detailed analysis of the ramifications of transaction costs. For more on the effects of transaction costs on the bargaining behavior of large and small groups, not confined to externality situations, see Olson (1964). Buchanan and Stubblebine (1962) show that a pollution tax can lead to too *little* pollution because the victims will bribe the polluters to reduce pollution beyond the optimal point induced by the tax. The significance

of this result is clearly weakened, it seems to me, by the prohibitive transaction costs in the typical large-number pollution case. The transaction costs argument has been turned around and used in *favor* of Coase in an imaginative way by Demsetz (1964). His point is that, where transaction costs block a Coasian solution the status quo must be optimal, in the sense that the benefits from moving are less than the costs. The difficulty with this argument is that it proves nothing about the desirability of an alternative solution, such as a tax or other collective action. Thus, we can turn around the transaction costs argument once again, and say that, where transaction costs block formation of a market, the relevant comparison is between doing nothing, letting the damage take its course, and imposing some sort of collective control. It is by no means obvious that the former will always be preferred. A useful analytical framework here is that of Arrow (1969), who observes that comparative transaction costs can affect the mode of economic organization. Thus, the cost of learning and communicating information, through prices, is low in a market system. On the other hand, the cost of exclusion may be high for some public goods, which is why they normally are not left to the market.

13. The insight into the potential for strategic behavior even in a two-party setting is due to Wellisz (1964). Mumey (1971) discusses the possibility that resources will be channeled into threatening actions or processes.
14. Income effects are analyzed by Dolbear (1967) and Mishan (1967). For a very amusing critique of the Coase Theorem and extensions as applied to pollution, see Mishan's (1971) "Pangloss on Pollution."
15. Methods of estimating damages are discussed in detail in the next section.
16. A version of this result has been obtained or discussed by many people. See for example Kneese (1964); Ruff (1970); Baumol and Oates (1971, 1975a); Baumol (1972); and Mishan (1974). The clear, nontechnical discussion by Ruff can be particularly recommended to noneconomists. A detailed empirical study of the comparative costs of taxes or effluent charges as opposed to uniform controls (discussed in the text that follows) to achieve a desired level of water quality in the Delaware estuary is discussed by Kneese (1977). The conclusion of the study is that the desired quality can be achieved for about half the cost with taxes.
17. For an approach that treats pollution as an input, but is similar in other respects to ours, see Baumol and Oates (1975*a*).
18. We must also assume that the firms are price takers in factor markets, importantly including the market for pollution. That is, the tax *rate* is not influenced by firm activities. This issue is further discussed by

Bohm (1970) and Baumol and Oates (1975a). A potential difficulty with the factor price assumption is that, after imposition of a tax, the prices may either be changed, or may no longer reflect real factor scarcities (assuming they did so in the original problem of social cost minimization). My guess is that this difficulty is likely to be of very little empirical importance.

19. It must also be true that $\alpha_k = \lambda_k$. Since the equations and parameters are the same in both cases (provided $t = \lambda$), the solution values of the variables, including α_k, must be the same. Away from equilibrium α_k is in general not equal to λ_k.
20. The reader seeking a discussion of some of the theoretical efficiency issues treated in this section, especially taxes versus direct controls, in a detailed, realistic setting might wish to consult the Kneese-Bower volume on the economics, technology, and institutions of water quality management (1968).
21. Kneese and Schultz (1975), in a nontechnical discussion of the history of air and water pollution policies in the United States, and desirable changes in these policies, argue that the incentive to technical change in pollution control may be the most important criterion for judging a policy. Discussiosn of the effect of a tax on control technology are found in Smith (1972), Orr (1976), and, most rigorously and comprehensively, Magat (1978). For a comparison of technical change under a subsidy for pollution control as opposed to a tax, see Wenders (unpublished). The conclusion is that a tax provides superior incentives.
22. For example, recycling, considered by many to be the ideal control technology, is not among the mandated technologies that qualify for water pollution control subsidies (Kneese and Schultz, 1975), with the result that the choice of technology is biased away from recycling. Similarly, low-sulfur western (U.S.) coal is discriminated against by the proposed New Source Performance Standards (NSPS) for coal-burning plants that mandate scrubbers. The advantages of the low-sulfur coal is that a plant using it does not need scrubbers to meet any reasonable ambient air quality standard, and it is this natural advantage that is impaired by the mandate.
23. For a detailed discussion of the alternatives for dealing with toxic substances, see Portney (1978). The Portney article appears in an RFF book, edited by him, containing articles by RFF researchers on several aspects of U.S. environmental policy.
24. This suggestion is due to Baumol and Oates (1975b).
25. It is recognized in a number of early contributions to the tax versus subsidy literature or, as it is also known, the bribes versus charges

literature. See for example Kamien, Schwartz, and Dolbear (1966), Freeman (1967), and Mills (1968).

26. The differing implications of tax and subsidy for firm profits are noted by Bramhall and Mills (1966). For an analysis of long-run effects on resource allocation among industries, see Porter (1974) and Baumol and Oates (1975a).
27. For a detailed critique of current subsidy policy along these lines see Kneese and Bower (1968) and Kneese and Schultze (1975). A variety of issues involving more efficient and equitable operation of the subsidy program is discussed by Renshaw (1974). He also suggests an argument *for* a subsidy, namely that a tax could be regressive in its impact on income distribution.
28. A result like this is obtained in the more richly detailed analyses of Tietenberg (1973, 1974a, b) and Hamlen (1978). An important contribution of these analyses, especially Hamlen's, is the modeling of spatial diffusion of emissions. Atkinson and Lewis (1976) consider some issues that arise in the setting of standards and taxes in a theoretical and empirical model of air pollution in the St. Louis area. See Rose-Ackerman (1973) for discussion of a variety of difficulties with a uniform tax. The spatial dimension may have been used first in formal externalities models by Førsund (1972).
29. The rights auction is perhaps first and most prominently associated with the work of Dales (1968). For further discussion of the advantages (and some disadvantages) see Ferrar and Whinston (1972), Tietenberg (1974c), and Baumol and Oates (1979).
30. For a formal analysis of adjustment costs in pollution control see Harford (1976).
31. This is recognized also by Baumol and Oates (1979) in their discussion of the advantage of a rights auction over a tax.
32. For a more formal derivation of this and other results on the effect of uncertainty on the choice of control instruments, see Adar and Griffin (1976). Formal analyses of control under uncertainty are also provided by Fishelson (1976) and Yohe (1976).
33. These considerations have been raised by several of the authors who discuss the merits of the rights auction.
34. The connection between the level and composition of economic activity and the pattern of residuals is provided by augmented input-output models. Along with conventional materials flows, these show residuals flows, and include a pollution abatement "sector." The original suggestion of a model of this sort is probably due to Cumberland (1966). An operational version, which takes account also of materials

balance, is in Cumberland and Korbach (1973). During this period a somewhat different model, which features a pollution abatement sector but does not account for materials balance, was developed by Leontief (1970). More complete models, which seek to account for materials flows back and forth from the natural environment to the economy, have been suggested by Isard (1969) and Victor (1972). Victor develops such a model and also provides a detailed review of the literature. More recently, dating from about 1974, an expanded and improved version of the early models (the SEAS model), has been developed and used by the U.S. Environmental Protection Agency. For a detailed (and sometimes critical) discussion of the properties of SEAS and related models, see Holdren, Harte, and Tonnessen (1978).

35. These processes are described with the aid of physical diffusion models. For some discussion and use of diffusion models by an economist, see Hamlen (1978).

36. There are literally hundreds of studies of these impacts of pollutant concentrations, for the most part, naturally enough, by noneconomists. An extremely useful published reference and guide to these for economists is the recent volume by Freeman (1979) on the evaluation of damages—or benefits from environmental improvement. An even more detailed review of the scientific literature on effects of air pollution is provided in an unpublished study by Hamilton (1979). Much of the discussion in the following text is drawn from these two excellent references, and a third (Scotchmer 1979) is described in the next note. For a comprehensive survey of studies linking air pollution and human health, see Lave and Seskin (1977). A review of evidence linking environmental factors to cancer, and suggestions for policies to deal with this problem, are found in Kneese and Schultze (1976).

37. Both methods are discussed by Freeman (1979). A useful feature of his discussion is a treatment of the welfare foundations of damage or benefit estimation. Empirical results are also reviewed. A very detailed review of both of the steps in the first method is provided by Hamilton (1979). Hamilton's work is part of a study for the Air Resources Board of California of methods of estimating and evaluating pollution damages. Another part of the study is a review and analysis of the second method, by Scotchmer (1979). As mentioned in the previous note, much of this section is based on these three references. Another useful source is the collection of studies on the valuation of social cost edited by Pearce (1978). For a discussion of issues in the benefit-cost analysis of water quality programs, see the studies in Peskin and Seskin (1975).

38. This problem is discussed further in Hamilton (1979), with references

to studies of actual crop shifts in response to pollution. In principle, a way to overcome the problem is to take the property-value capitalization approach. As we discuss later, virtually all such studies have been of *residential* property values. I am aware of one study of the effect of pollution on the price of agricultural land, by Crocker (1971). Advantages and disadvantages of the property-value approach are considered in the following text. One special disadvantage in the agricultural setting is the possible correlation between air pollution, which presumably depresses values and encroaching urban development, which presumably raises the

39. An estimate of crop damage from air pollution in California, though alarming in some absolute sense, represents less than 1.00 percent of the total value of California crops, and less than 0.25 percent of the total value of U.S. crops (Millecan 1976).
40. Studies describing effects on various quality characteristics are discussed in Hamilton (1979). The estimate of over US$1 billion in damages to vegetation is due to Heck and Brandt (1977).
41. The first in a series of publications by Lave and Seskin is a 1970 *Science* article. Their 1977 book provides a much more comprehensive analysis and discussion of results. For a guide to the extensive literature on air pollution and human health, see Lave and Seskin (1977) and Hamilton (1979).
42. For example, Bailey (1978) has inferred values ranging from US$1.9 million to US$625 million for a statistical life from standards promulgated by the U.S. Occupational Safety and Health Administration.
43. For example, the U.S. Federal Highway Administration and the National Highway Traffic Safety Administration both use a figure of about $250,000 (Hapgood 1979), derived from explicit "risk-benefit" analyses.
44. This conjecture is proved by Conley (1976), who shows that willingness to pay necessarily would exceed the present value of earnings.
45. Probably the best-known work here is by Thaler and Rosen (1976), who provide a theoretical and empirical analysis of interoccupational wage differences, especially as related to risk differences. There have been a number of other studies as well, however. For references, see Hamilton (1979), Table 1.
46. One study I am aware of that looks at intraoccupational differences (for miners) is that of Usher (1973). Interestingly, his estimates are in the same range as Thaler and Rosen's (see Table 1).
47. The pioneering work here, to my knowledge, is due to Ridker (1967) and Ridker and Henning (1967). For references to and brief descriptions of the many studies undertaken since, see Freeman (1979). For an

application to noise pollution consistent with the theory described in the following text, see Nelson (1978); and for a review of studies of the relationship between noise and property values and an application to airport siting in the London area, see Walters (1975). An estimate of the relationship between lakeshore property values and lake water quality is made by David (1968). Freeman (1979) suggests an adaptation to water quality of the theory originally developed to evaluate differences in air quality.

48. A concise "guided tour" of data, methods, and results for each study is provided by Freeman (1979).
49. Two of the early theoretical analyses of the relationship between pollution and property values, by Strotz (1968) and Lind (1973), focus on land as a productive input, rather than a residential site. Other theoretical analyses (including Freeman, 1974, 1979; Polinsky and Shavell, 1975, 1976; and Polinsky and Rubinfeld, 1977) consider residential property values, as do most of the empirical studies.
50. The discussion that follows is based on the theoretical analyses of Scotchmer (1979) and Freeman (1979).
51. For further discussion of the measurement and interpretation of hedonic prices, see Rosen (1974).
52. At first blush, the persistence of wage differences seems inconsistent with the factor price equalization theorem. But as Freeman (1979) and Scotchmer (1979) show, conditions needed for the theorem to hold probably are not met in this situation.
53. For estimates of the relationship between urban amenities or dis-amenities and wage rates, see Hoch (1972), Nordhaus and Tobin (1973), Tolley (1974), and Meyer and Leone (1977).
54. For a discussion of how a study might be set up, the kinds of data needed, and the econometric considerations, see Scotchmer (1979).
55. A potential source of bias of an indeterminate nature that involves both theory—under what conditions will surplus be capitalized in property values—and econometric procedure, is housing-market segmentation. That is, if an urban housing market is really a set of separate markets, with barriers to mobility *between* them, separate hedonic price functions would have to be estimated. This issue was first raised by Straszheim (1974) and is discussed by Freeman (1979). A study by Harrison and Rubinfeld (1978) suggests substantial variation in estimated benefits from an air-quality improvement in the Boston area depending on how the market is stratified. On the other hand, Nelson (1978) finds no significant difference between urban and suburban hedonic price functions in the Washington, D.C., area.

56. One other pitfall here that is not really behavioral, rather has to do with the form in which the data are likely to come, as suggested by Niskanen and Hanke (1977), is the existence of income and (especially) property taxes. See also Freeman (1979) for a detailed discussion and some estimates of the size and direction of bias in studies that ignore tax effects.
57. Both Freeman (1979) and Scotchmer (1979) provide discussions, with references, of survey approaches. The studies directed specifically to valuing pollution abatement are Randall, Ives, and Eastman (1974), and Brookshire, Ives, and Schulze (1976).
58. An early empirical study of some aspects of the distribution of air and water pollution damages in the United States is by Freeman (1972). More recent studies include those by Zupan (1973) for air quality in the New York area; Harrison (1975) for costs of air pollution control; Dorfman and Snow (1975) for costs of pollution control generally; Dorfman (1976) for benefits and costs of environmental programs; Spofford, Russell, and Kelly (1976) for benefit and costs of controlling air and water pollution in the Delaware estuary; Freeman (1977) for costs of controlling automotive air pollution; Gianessi, Peskin, and Wolff (1977) for air pollution policy in the United States; and Peskin (1978) for the U.S. Clean Air Amendments of 1970.

    Distributional considerations have been introduced into models of representative or legislative environmental decision making by Haefele (1973), and Dorfman and Jacoby (1972). For a review and further analysis, see Portney, Sonstelie, and Kneese (1974), and Kneese and Bower (1979).
59. For an example of the former, see Sims (1979), and for the latter, Ethridge (1973).
60. Early RFF studies of industrial water use, such as the one by Löf and Kneese (1968) for the beet sugar industry, exemplify the first, relatively informal phase of this line of research. Later RFF studies expanded the scope of the analysis to take into account all residuals, not just waterborne ones. In this category are studies of petroleum refining (Russell, 1971, 1973); steel production (Russell and Vaughn, 1974, 1976); pulp and paper (Bower, Löf, and Hearon, 1971); and steel scrap recycling (Sawyer, 1974).

    The linear programming approach in the Russell studies has been further developed by Thompson and his collaborators (Thompson and Young, 1973; Calloway, Schwartza, and Thompson, 1974; Singleton, Calloway, and Thompson, 1975; Calloway and Thompson, 1976). The Calloway and Thompson study is noteworthy in that it considers several related industries in a region (the Texas Gulf Coast): petroleum refining, electric power production, and chemicals. Finally, an explicitly reginal approach, focusing on all residuals in a geographic area, is taken in the RFF studies of the Delaware estuary by Russell and Spofford (1972); Spofford, Russell, and Kelly (1976): and Russell and Spofford (1977).

Much of this work is reviewed in a recent volume by Kneese and Bower (1979). For a further review of industrial water pollution control studies in the RFF tradition, see Hanke and Gutmanis (1975).

61. For such studies of a tax on the sulfur content of fuels in the electric power industry, see Griffin (1974a, 1974b) and Chapman (1974). For an application to an effluent charge in the Canadian brewing industry, see Sims (1979).

62. See Evans (1973), and for a review and discussion of the Evans study and a couple of others, see Haveman and Smith (1978).

63. The range of choice in water pollution control is emphasized in an early RFF study by Davis (1968).

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