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# The Instruments for Environmental Policy

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In their part of the continuing dialogue on environmental policy, economists have quite naturally stressed the role of policy tools operating through the pricing system. The case for heavy reliance on effluent charges to internalize the social costs of individual decisions is, at least in principle, a very compelling one. However, a cursory survey of potential policy instruments reveals the existence of a wide spectrum of methods for environmental control ranging from outright prohibition of polluting activities to milder forms of moral suasion involving voluntary compliance.

In spite of the economist's predilection for a central role for direct price incentives, we suspect that even he recognizes that a comprehensive and effective (and even the "optimal") environmental policy probably involves a mix of policy tools with the use of something more than only effluent fees. The purpose of this paper is a preliminary exploration of the potential and limitations of the various policy tools available for environmental protection; our concern here is what we can say in a systematic way about the particular circumstances under which one type of policy is more appropriate than another and how various policy tools can interact effectively. We stress the word preliminary, because this paper is, in effect, an interim report on a study of environmental policy.

In the first section, we enumerate and classify the available policy instruments. In the following three sections, we present a simple concep-

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tual framework for the analysis of environmental policies and a discussion of what *in principle* would appear to be the appropriate roles for the various policy tools. We turn in the fifth section to an empirical examination of the effectiveness of the different environmental policies. Our work here is in its early stages; we have at this point some admittedly fragmentary and piecemeal evidence on the efficacy of available policy instruments. In some cases, we have had to rely upon evidence that is indirect, occasionally derived from experiences other than environmental programs, to obtain some insight into the likely effectiveness of a particular policy tool.

## Policy Tools for Environmental Protection

Before examining the various active policy options available for the control of environmental quality, we want to acknowledge the case for a policy of no public intervention: we could rely wholly on the market mechanism as an instrument for the regulation of externalities, unimpeded by public programs designed to protect the environment. In fact, as Ronald Coase has shown in his classic article, it is actually possible, under certain conditions, to achieve an efficient pattern of resource use through private negotiation that internalizes all social costs or benefits. This can, at least in principle, result from the incentive for parties suffering damage from the activities of others to make payments to induce a reduction in these activities.

The difficulties besetting the Coase solution are well known, particularly the free rider problem and the role of transaction costs. The main point we wish to make here is that the Coase argument is plausible only for the small group case, for only here is the number of participants sufficiently small for each to recognize the importance of his own role in the bargaining process.[1] Note, moreover, that this requires small numbers on *both sides* of the transaction; even if the polluter is a single decision-maker, a Coase solution is unlikely if the damaged parties constitute a large, diverse group for whom organization and bargaining is costly. A quick survey of our major environmental problems—air pollution in metropolitan areas, the emissions of many industries and municipalities into our waterways—indicates that these typically involve large numbers.

1. Even in the small group case, the use of certain bargaining strategies or institutional impediments to side payments may prevent efficient outcomes.

This would suggest that the Coase solution is of limited relevance to the major issues of environmental policy.[2]

Turning to the remaining policy alternatives, we present in the following list a classification of policy tools that is admittedly somewhat arbitrary. We will examine four classes of policy instruments. The first category includes measures that base themselves on economic incentives, either in the form of taxation of environmentally destructive activities or, alternatively, of subsidization of desired actions. Under the second heading, we group programs of direct controls consisting of quotas or limitations on polluting activities, of outright prohibition, and of technical specifications (e.g., required installation of waste treatment devices). Third, we consider social pressure with no legal enforcement powers so that compliance on the part of individual decision makers remains voluntary. Finally, the fourth set of programs consists of an actual transfer of certain activities from the private to the public sector.

***Tools for environmental policy:***

1. Price Incentives[3]
   a) Taxes
   b) Subsidies
2. Direct Controls
   a) Rationing
   b) Prohibition
   c) Technical Specifications
3. Moral Suasion: Voluntary Compliance
4. Public Production

We stress at the outset that, while the list seems simple enough, it does conceal the vast number of ways in which these policy tools may be employed. Taxes, for example, may vary with time and/or place, may apply to particular inputs, or, alternatively, outputs or byproducts of productive activities, and so forth. Similarly, direct controls on polluting activi-

2. In certain instances, no intervention may, of course, be optimal for totally different reasons: not because the market will resolve the externalities itself, but because in that particular case the damage happens to be small while the social cost of regulation is large. Here we fail to intervene not because the disease will cure itself, but because the cure is worse.

3. The auctioning of pollution rights could be added here. However, considering the major environmental problems before us, the practicality of this proposal seems to us rather limited.

ties can take an enormous variety of forms, involving the courts or special regulatory agencies, permitting and sometimes encouraging citizen lawsuits, and so forth. This list is neither exhaustive nor composed of mutually exclusive policy measures. Programs of taxes and regulations, for example, can be combined to control waste emissions; we will, in fact, consider such policy mixes shortly.

## Forms of Environmental Damage

In this section, we consider, in general terms, the various forms that insults to the environment may take. More specifically, we are interested in different types of environmental damage functions. As we will argue later, the damage function that characterizes a particular type of polluting activity may be of central importance in determining the policy instrument appropriate for its control.

The first distinction is between the situation in which the current level of environmental quality is a function of the *current* level of the polluting activity and the case where it depends on the history of *past* levels of the activity. The state of purity of the air over a metropolitan area, for example, depends largely on the quantities of pollutants currently being emitted into the atmosphere. This we will call a *flow* damage function.

Alternatively, past levels of activity may build up a stock of pollutant. Therefore, the extent of environmental damage depends on the history of the activity. This we call a *stock* damage function. Such damage functions are typically associated with nondegradable pollutants, such as mercury and DDT. The pollutant accumulates over time and thus constitutes an ever increasing environmental threat. The stock and flow damage functions are pure, polar cases. In reality there is a spectrum of damage functions in which historic levels of polluting activity assume varying degrees of importance in determining the present level of environmental quality.[4] However, the distinction is a useful one for certain policy purposes.

Of equal importance is the particular form of the damage function. Economists are familiar with cost functions which exhibit monotonically increasing marginal costs; a familiar example in the literature is the case

4. For an interesting theoretical study using a more general damage function which incorporates both stock and flow elements, see C. G. Plourde.

of crowding on highways. Once costs of congestion set in, the time loss to road users resulting from the presence of an additional vehicle rises rapidly with the number of vehicles. Many environmental phenomena, however, appear to involve more complex damage functions; some exhibit important discontinuities or threshold effects. When, for example, waste loads in a river become sufficiently heavy, the "oxygen sag" may become so pronounced that the assimilative capacity of the stream is exceeded. The dissolved-oxygen content may in such cases fall to zero, giving rise to anaerobic conditions. In such cases, the cost of exceeding the threshold level of the activity may be exceedingly high. There may, moreover, exist a series of thresholds so that the damage function can be exceedingly complex. In addition, the precise form of the damage function itself may be problematic, thus injecting an important element of uncertainty into the situation.

The uncertainty element in the damage function is not a haphazard affair, but arises out of the very nature of the relationship. It is essential to recognize that damage functions are multivariate relationships, functions of a vector of variables many of them entirely outside the control of the policy maker. The effects of a given injection of pollutants into the air depend on atmospheric conditions. The damage caused by a waste emission into a stream is determined largely by the level of the water flow: it may be relatively harmless when poured into a stream that is near its crest, but very dangerous when put into the same stream when depleted by drought. Externalities in urban affairs will be more or less serious depending on the state of racial tension, the level of narcotics use, and a variety of other crucial influences.

Expressed somewhat more formally, the function describing the determination of environmental quality at time $s$, $q_s$, may be written

$$q_s = f(m_s, E_s), \tag{1}$$

where $m_s$ is the level of waste emissions and $E_s$ is a vector whose components are environmental conditions, such as the direction and velocity of the wind, the quantity of rainfall, and so forth. The important thing about $E_s$ is that it includes variables over which we have little, if any, control. The exogenous variables describing the vector, $E_s$, are themselves likely to be random variables, or at least subject to influences which can best be treated as random.

The environmental damage function may be defined as

$$z_s = g(q_s) = h(m_s, E_s). \tag{2}$$

While $q_s$ indicates the state of environmental quality (e.g., the sulfur dioxide content of the atmosphere or the dissolved oxygen level of a waterway), $z_s$ denotes the social cost associated with the value of $q_s$. For example, higher levels of sulfur dioxide in the air people breath appear to induce a higher incidence of respiratory illnesses and mortality (see Lave and Seskin); the costs associated with these repercussions are represented by $z_s$.[5]

The introduction of uncontrolled determinants of environmental quality and the associated uncertainty creates some difficult policy problems. For example, environmental control policy may have a combination of several objectives such as (a) the achievement *on average* of a level of environmental quality, $q_s$, such that the cost of environmental insults is acceptable; and (b) prevention of the attainment of some threshold level of $q_s$ at which there is discontinuity in the damage function, thus causing social costs to soar to unacceptably high levels.

If the values for the components of $E_s$ were known precisely for all future periods, we could set values of $m_s$ for each period $s$ so as to achieve these objectives, and we would look for the least cost methods of holding emissions to these specified levels. Unfortunately, we frequently do not know the values of $E_s$ in advance. Normally however, we can make some predictions about them. In fact, we almost have a kind of probability distribution for variables such as weather conditions. Often the dispersion of the distribution becomes much smaller as the pertinent point in time approaches (e.g., we have a better idea about tomorrow's weather than next week's weather).

Even so, the policy maker cannot control most of the variables in the vector, $E$, and even his ability to foresee their values remains highly limited. The science of meteorology has not yet reached a stage at which forecasts can be made with a high degree of certainty. Meteorologists are unable to determine the timing of next year's or even next month's atmospheric inversions or rainfall patterns so that plans for the intermittent crises that are likely to result may be made in advance. This phenomenon can be extremely important in the selection of policy tools. It may be that, because of limited attention to this issue in the economics

5. More realistically, we can regard $q_s$ and $m_s$ as vectors whose components represent, respectively, various measures of environmental quality and levels of discharges of different types of wastes. This, however, seems to add little to the analysis. Note that $z_s$ is a scalar, not a vector, for it represents the social cost, measured in terms of a numeraire, of the level of environmental deterioration ($q_s$) generated jointly by $m_s$ and $E_s$.

literature, we have tended to overlook the merits of policy instruments usually favored outside the profession.

## Matching Policy Tools with Environmental Conditions

Before proceeding to a more detailed empirical analysis of policy tools, we want to consider under what circumstances one policy tool is likely to be more appropriate than another. As a frame of reference, let us assume a set of standards or targets for environmental quality with an eye toward devising an effective environmental policy to realize these standards.[6]

In the case of stock damage functions with costs directly related to the accumulated quantity of the pollutant, a positive level of the polluting activity implies that the *level* of environmental damage will increase continually over time. The stock of pollutants will increase over time with the flow of emissions from one period to the next. Environmental quality will thus continue to deteriorate. Any damage thresholds may eventually be exceeded, and clearly the target level of environmental quality will not be achieved. In these cases there would appear to be a strong case for outright prohibition of polluting activities, for simply reducing the level of the activity will serve only to slow the cumulative process of environmental deterioration.[7] Outright prohibition would, therefore, seem to be an appropriate policy measure where damage functions are of the stock form. The recent ban on the use of DDT in the United States is a case in point.

Where, in contrast, environmental quality depends primarily on the current level of polluting activities, prohibition may be excessively costly. Achievement of the target level of environmental quality requires adjustment of the current levels of activities to those consistent with the target.

6. We could specify alternative types of objective functions. For example, we could assume standard utility and cost functions and, following the usual maximization procedures, derive our first order optimality conditions requiring that environmental quality be improved (or polluting activities curtailed) to the point where benefits and costs are equal at the margin. The major problem here is the difficulty of measuring benefits and costs. On this issue, see, for example, Baumol and Oates. Most of the discussion in the present paper applies, incidentally, to both of these approaches to environmental policy.

7. It might be desirable to curtail the flow of emissions gradually over time if the costs of rapid adjustment are high. This raises the interesting problem of the optimal path of reduction in the rate of flow, a problem which we note but which goes beyond the scope of this paper.

These required levels, in many cases at least, can be expected to be nonzero. A variety of the policy instruments included in our earlier list may then be appropriate to influence levels of polluting activities.

What *in principle* can we say about the relative effectiveness of these policy instruments? The efficiency-enhancing properties of taxes (effluent charges) are widely recognized and need little discussion here.[8] In terms of our objective, the realization of a set of specified standards of environmental quality, we have shown elsewhere (Baumol and Oates) that, assuming cost-minimizing (not necessarily profit-maximizing) behavior by producers, effluent charges are the least-cost method of attaining the target: the proper effluent fee will generate, through private decisions, the set of activity levels which imposes the lowest costs on society. Any other set of quotas determined by regulatory authorities and consistent with the specified environmental standards will thus involve a higher opportunity cost.

This would appear to establish a presumption at the conceptual level in favor of price incentives over regulatory rationing, and to make a system of fees an ideal standard with which others should be compared and judged as more or less imperfect substitutes. However, the proof of the superiority of the tax instrument involves a number of simplifying assumptions (and typically utilizes a static analytic model); there are several other critical considerations without which it is impossible to understand fully the inclination toward other policy instruments on the part of many noneconomists who are demonstrably well informed and well intentioned.

Once we enumerate these elements, their relevance is obvious. We will show that on economic grounds they may often call for measures other than the tax instruments that receive primary attention in the economic literature. This list includes the following.[9]

8. See, for example, Kneese and Bower, and Upton.

9. We might consider adding to this list the "political acceptability" of the program. This is not without an important economic dimension. Suppose we are given two programs *A* and *B* the first of which is shown capable of yielding an allocation of resources slightly better than that which would be produced by the latter. However, suppose that *B* can be "sold" to a legislature with little expenditure of time and effort, while the enactment of *A*, if it can be secured at all, would require a highly costly and time consuming campaign. In such a case, *purely economic considerations* may favor the advocacy of *B* in preference to *A*, if we are willing to take the predisposition of the legislature as a datum in exactly the same way we take the production function for a particular product as given for the problem of determination of outputs.

1. Administrative and enforcement costs (playing a role analogous to transactions costs elsewhere in theoretical analysis).

2. Exclusion or scale problems, which may make it difficult for the private sector to provide activities appropriate for the protection of the environment. (If one wishes, this can be classified as a special case of the problem of high administrative costs, the costs of collecting payment for an environmental service or of assembling the large quantity of capital needed to supply it efficiently.)

3. Time costs. Here we include not only the interval necessary to design a program and put it into effect, but also the period of adjustment of activities to the program.

4. Problems of uncertainty.

Let us now explore how these considerations, in the context of the objective of allocative efficiency, influence the choice among the basic types of policies listed in "Tools for Environmental Policy."

### *Pollution taxes*

Beginning once more with the tax measures we see that, in addition to their desirable allocative properties, effluent charges possess a further major attraction: their enforcement mechanism is relatively automatic. Unlike direct controls, they do not suffer from the uncertainties of detection, of the decision to prosecute, or of the outcome of the judicial hearing including the possibility of penalties that are ludicrously lenient. Like death, taxes have indeed proved reasonably certain. Few are the cases of tax authorities who neglect to send the taxpayer his bill, and that is the essence of the enforcement mechanism implicit in the tax measures. They require no crusading district attorney or regulatory agency for their effectiveness.

However, once we leave this point, we are left with considerations in terms of which tax measures generally score rather poorly. We will defer the issue of time costs to a later point where its role will be more clear. It is true that *enforcement* costs are likely to be relatively low, although like any other taxes we can be confident that they will provide work for a host of tax attorneys employed to seek out possible loopholes. Perhaps more important in many cases are high monitoring or metering costs. One of the major reasons additional local telephone calls are supplied at zero charge to subscribers in small communities is the high cost of devices that record such calls, and the same is apparently true of communities in which water usage is not metered universally. This is particularly to the point when we recognize that allocative efficiency requires tax charges to vary by season of the year, time of day, or with unpredictable changes in environmental conditions (e.g., the charge on smoke emissions should presumably rise sharply during an atmospheric "inversion" that produces a serious deterioration in air quality). Moreover, in many cases there is no one simple variable whose magnitude should be monitored. Waste emissions into waterways should ideally be taxed according to their BOD level, their content of a variety of nondegradable pollutants, their temperature, and perhaps their sheer volume. Obviously, the greater the number of these critical attributes, the more costly will be the monitoring program required by an effective tax policy. This, of course, increases the complexity of other types of regulatory programs as well.[10]

A special problem may arise from the structure of the polluting industry. Under pure competition, fees will, in principle, work ideally; in addition, it is easy to show that they tend to retain their least-cost properties in any industry in which firms minimize cost per unit of output. However, under oligopoly or monopoly, management's interests may conflict with such a goal, and taxes on polluting activities may fail to do their job with full effectiveness. If an industry routinely shifts virtually all of the cost of such fees without attempting to reduce waste emissions in order to lower its tax payments, much of the intended effect of the tax program will be lost.

From all this we do not conclude that economists have been ill-advised in their support of tax measures. On the contrary, we continue to believe strongly that in many applications they will in the long run prove to be the most effective instrument at the disposal of society. However, it is clear that certain environmental and industrial characteristics can impair

10. The technology of monitoring industrial waste emissions appears still to be in its infancy; metering devices which provide reliable measures of the composition and quantities of effluents at modest cost are (to our knowledge) not yet available. Environmental officials in New Jersey, for example, rely heavily on periodic samples of emissions which they subject to laboratory tests, which involve costly procedures. However, there is a considerable research effort underway to design effective and inexpensive metering mechanisms. This may well reduce substantially the administrative costs of programs whose effectiveness depends on measurement of individual waste discharges. In this connection, William Vickrey has stressed, in conversation with us, the dependence of the cost of metering on the degree of accuracy we demand of it. In many cases, high standards of accuracy may not be defensible. As Vickrey points out, a ten-hour inspection of an automobile will undoubtedly provide a more reliable and complete description of its exhaust characteristics than a half-hour test, but it is surely plausible that the former exceeds the standard of "optimal imperfection" in information gathering!

their effectiveness. This, as we will suggest shortly, may point to the desirability of a mixed policy of fees and controls.

### *Subsidies*

An obvious alternative to taxes is the use of subsidies to induce reductions in the levels of these activities; what can be accomplished with the stick should also be possible with the carrot. Kneese and Bower, for example, have argued that "Strictly from the point of view of resource allocation, it would make no difference whether an effluent charge was levied on the discharger, or a payment was made to him for not discharging wastes" (p. 57). However, in addition to some extremely important differences at the operational level between taxes and subsidies, Bramhall and Mills have pointed out a fundamental asymmetry between the effects of fees and payments. While it is true that the price of engaging in a polluting activity can be made the same with the use of either a tax or subsidy, the latter involves a payment to the firm while taxes impose a cost on the firm. As a result, the firm's profit levels under the two programs differ by a constant. We have shown formally that, in long-run competitive equilibrium, subsidies (relative to fees) will result in a larger number of firms, a larger output for the industry, and a lower price for the commodity whose production generates pollution. Moreover, it is plausible the net effect will be an *increase* in total industry emissions over what they would be in absence of *any* intervention. Subsidies tend to induce excessive output. Thus, at least at a formal level, taxes are to be preferred.[11]

### *Direct controls*

Direct controls often seem to score poorly on most of our criteria, in spite of their appeal to a curiously heterogeneous group composed largely of activists, lawyers, and businessmen. They are usually costly to administer,

11. Subsidies may be desirable if there is reason to suspect that direct controls constitute the only alternative that is feasible politically. Two reasons for this are obvious to the economist: a) direct controls are likely to allocate pollution quotas among polluters in an arbitrary manner while taxes *or* subsidies will do this in a manner that works automatically in the direction of cost minimization; b) a direct control that prohibits a polluter from, say, emitting more than $x$ tons of sulfur dioxide per year, under threat of punishment, offers that polluter absolutely no incentive to reduce his emissions one iota below $x$ even though the private cost of that reduction to him is negligible compared to its social benefits. Thus, subsidies may sometimes be preferable to direct controls even though both of them produce misallocations.

because they involve all the heavy costs of enforcement without avoiding entirely the costs of monitoring in whose complete absence violations simply cannot be detected. We have already noted their tendency to produce a misallocation of resources. Moreover, experience suggests that their enforcement is often apt to be erratic and unreliable, for it depends largely on the vigor and vigilance of the responsible public agency, the severity of the courts, and the unpredictable course of the public's concern with environmental issues.

Yet direct controls do possess one major attraction: *if enforcement is effective,* they can induce, with little uncertainty, the prescribed alterations in polluting activities. We cannot expect controls to achieve environmental objectives at the least cost, but they may be able to *guarantee* substantial reductions in damages to the environment, a consideration that may be of particular importance where threats to environmental quality are grave and time is short. This points up two limitations of effluent charges: first, the response of polluters to a given level of fees is hard to predict accurately, and second, the period of adjustment to new levels of activities may be uncertain. If sufficient time is available to adjust fees until the desired response is obtained, the case for effluent charges becomes a very compelling one. However, environmental conditions may under certain situations alter so swiftly that fees simply may not be able to produce the necessary changes in behavior quickly (or predictably) enough. Where, for example, the air over a metropolitan area becomes highly contaminated because of extremely unfavorable weather conditions, direct controls (perhaps involving the prohibition of incineration or limiting the use of motor vehicles) may be necessary to avoid a real catastrophe.

There *may* be a further role for direct controls in industries dominated by a few large firms whose market power enables them to pass forward taxes on polluting activities without much incentive to undertake major adjustments in production techniques to reduce environmental damage.[12] This is frankly a difficult case to evaluate. Perhaps the best example is the ongoing attempt to impose technical standards for exhaust discharges on new automobiles. Because of the highly concentrated character of the auto industry, it is not clear that taxes on motor vehicles (perhaps graduated according to the level of exhaust emissions) would have much effect

12. Of course, it is normally desirable that some portion of the tax be passed forward in the form of price increases, as a means to discourage demand for the polluting output. The issue is that an oligopoly whose objectives are complex may not always minimize the costs of producing its vector of outputs.

on automobile design or usage.[13] A more promising approach may consist of legislated emission standards that will compel alterations in the design of engines so as to reduce the pollution content of vehicle discharges. However, the use of standards also involves difficult problems: witness the protracted "bargaining" between auto-industry representatives and federal legislators over the level of the standards and the timing of their implementation. Moreover, there is always the danger of adopting standards approaching complete "purity" that impose enormous costs; the reduction of polluting activities typically involves marginal costs that increase rapidly as the required reductions in waste discharges approach 100 per cent. The setting of emission standards without adequate regard for the costs involved may produce some highly inefficient results.

### *Hybrid programs*

Even those policy makers who have come to recognize the merits of a system of charges as an effective instrument of control seem normally unwilling to rely exclusively on this measure. Rather they typically prefer a mixed system of the sort in which each polluter is assigned quotas or ceilings which his emissions are in any event never to be permitted to exceed. Taxes are then to be used to induce polluters to do better than these minimum standards and to do so in a relatively efficient manner.

While this may at first appear to be a strange mongrel, some of the preceding discussion suggests that, under certain circumstances, such a mix of policies may have real merit. If taxes are sufficiently high to cut emissions well below the quota levels, the efficiency properties of the tax measure will be preserved. Moreover, it retains the advantage of the pure fiscal method in forcing recognition of the very rapidly rising cost of further purification as the level of environmental damage is reduced toward zero. It is all too easy to set quotas at irresponsibly demanding levels, paying no attention to the heavy costs they impose. But it is hard not to take notice when tax rates must be raised astronomically to achieve still further improvements in environmental quality.

On the other hand, the quota portion of the program can make two important contributions, safety and increased speed of adjustment and implementation. Suppose, for example, there is a threshold in the damage

13. As Roger Noll points out, the case for effluent fees is the weakest "when regulators must deal with firms with considerable market power, and, at the other extreme, individuals with very little freedom of choice arising either from a lack of economic power, lack of knowledge, or lack of viable technical options" (pp. 34–5).

function so that a form of environmental abuse imposes a serious threat, but only beyond some point that is fairly well known. In this case, a hybrid policy can make considerable sense, since the quotas it utilizes can be employed to make reasonably certain that damages never get beyond the danger point. Taxes can be unreliable for this purpose, since, as noted earlier, the tax elasticities of pollution output are generally not well known and these fees may not induce changes in activity levels with sufficient rapidity. Thus, reliance on tax incentives alone may impose unacceptable risks, which can be prevented by a set of direct controls that set ceilings on levels of polluting activities.

Controls can, moreover, introduce additional flexibility into an environmental program. In terms of our illustrative case, urban air pollution, we noted that authorities may be able to invoke temporary prohibition, or at least limitations, on polluting activities when environmental deterioration suddenly reaches extremely serious levels.

Hybrid programs of taxes and controls thus represent a very attractive policy package. The tax component of the program functions to maintain the desired levels of environmental quality under "normal" conditions at a relatively low cost and also avoids the imposition of uneconomically demanding controls. The controls constitute standby measures to deal with adverse environmental conditions that arise infrequently, but suddenly, and which would result in serious environmental damage with normal levels of waste emissions.[14] Such a mixed program should not involve notably higher administrative costs than a pure tax policy, since much of the monitoring structure used for the latter should also be available for enforcement of the controls. In sum, where threshold problems constitute a serious environmental threat and where levels of polluting activities may require substantial alteration on short notice, which is not a rare set of circumstances, a hybrid program using both fees and controls may be preferable to a pure tax-subsidy program.

### *Moral suasion: voluntary compliance*

We come next to the cases in which it seems appropriate to rely on appeals to conscience and voluntary compliance. As economists, we tend to be somewhat skeptical about the efficacy of long-run programs which

14. In this volume, Lave and Seskin report evidence that the mortality danger of air pollution crises may have been exaggerated. Nevertheless, it remains true that, during periods of stagnant air, the social cost of a given emission level will be high, because a great proportion of the polluting element remains over the city for a protracted period.

require costly acts of individuals but offer no compensation aside from a sense of satisfaction or the avoidance of a guilty conscience. In fact, the appeal to conscience can often be a dangerous snare. It can serve to lure public support from programs with real potential for the effective protection of the environment. Later, we will provide some evidence that suggests this to be a real possibility.

There is nevertheless an important role for voluntary programs. In particular, in an unanticipated emergency there simply may be no other recourse: the *time cost* of most other instruments of control may be too high to permit their utilization under such circumstances. A sudden and dangerous deterioration of air quality allows no time for the imposition of a tax or for the drawing up and adoption of other types of regulatory legislation. There may be no time for emergency controls, particularly if they have not previously been instituted in standby form, but there can be an immediate appeal to the general public to avoid the use of automobiles and incinerators until the emergency is passed. Moreover, as we shall indicate in a later section, there is evidence to indicate that the public is likely to respond quickly and effectively to such an appeal. Perhaps social pressures and a sense of urgency lie behind the efficacy of moral suasion in such cases.[15]

Casual observation suggests that the sense of high moral purpose is likely to slip away rather rapidly and thus implies little potential for long-term programs that rest on no firmer base than the public conscience. However, that is no reason to reject this instrument where it can prove effective, particularly since no effective alternative may be available. We suspect that we have not yet experienced the last of the unforeseen emergencies and, in extremis, time cost is likely to swamp all other costs in the choice of policy instruments.

### *Public provision of environmental services*

The direct public "production" of environmental quality may be justified in two types of situations. The first is the case where the current

15. There is another precondition for the efficacy of moral suasion, even in an emergency. We can usually expect a few individuals not to respond to a public appeal. Thus, voluntarism cannot be relied upon in a case where universal cooperation is essential, as during a wartime blackout where a single unshielded light can endanger everyone. However, in most environmental emergencies as long as a substantial proportion of the persons in question are willing to comply with a request for cooperation, a voluntary program is likely to be effective. For example if, during a crisis of atmospheric quality, an appeal to the public may lead to a temporary reduction in automotive traffic of some 70 or 80 per cent, that may well be sufficient to achieve the desired result.

quality of the environment is deemed unsatisfactory (i.e., falls below the specified standard) as a result of "natural" causes and where this cannot be corrected through market processes because the particular environmental service is a public good. It is hard to find a perfect illustration, but natural disasters such as periodic droughts or flooding come close. Here the problem is not one of restricting polluting activities on the part of the individual; it is one of providing facilities such as dams and reservoirs to prevent these catastrophies. The private sector of the economy may handle such situations adequately if the commodity needed to avert the disaster is not a public good—that is, if exclusion is possible (or, more accurately, not too costly) and consumption is rival. However, where exclusion is difficult and/or consumption is joint, as in the case of protection from flood damage, the public sector may have to take direct responsibility for the provision of the good.

The second type of situation in which direct public participation *may* be appropriate is that involving large economies of scale and outlays. An example may be the case of a large waste treatment facility used by a multitude of individual decision makers. The reduced cost of treatment of effluents made possible by a jointly used plan may not be realized if left to the private sector.

This example, incidentally, suggests a further type of environmental service that the public sector must provide, namely the planning and direction of systems for the control of environmental quality. The need of reaeration devices, for instance, depends upon water flows (influenced by reservoir facilities), the levels of waste emissions (determined in part by current fees or regulations), and so forth. The point is that the control of water quality in a river basin or atmospheric conditions in an air shed requires systematic planning to integrate effectively the use of quality-control techniques. Kneese and Bower stress the need for river basin authorities to plan and coordinate a program of water-quality management. Urban areas require similar types of authorities to develop integrated air quality programs. Thus, public agencies must not only directly provide certain physical facilities, but must also exercise the management function of coordinating the variety of activities and control techniques that serve jointly to determine environmental quality. Such agencies need not be federal, but must be sufficiently large so that their jurisdiction includes those activities that influence environmental conditions in a given area. This implies jurisdictions sufficiently large to encompass systems of waterways and areas whose atmospheric conditions are dependent on the same activities.

## Optimal Mixed Programs: A Simple Model

The logic of the argument in the preceding section for the use of hybrid programs in the presence of random exogenous influences can be more clearly outlined with the aid of a simple illustrative model. Such a model can indicate not only the potential desirability of such a hybrid as against a tax measure or a program using direct controls alone, it can also illustrate conceptually how one might go about selecting the optimal mix of policy instruments.

A relationship apparently used frequently in the engineering literature to describe the time path of environmental quality is (in a much simplified form)[16]

$$q_s = k_s q_{(s-1)} + m_s, \tag{3}$$

where:

- $q_s$ is a measure of environmental quality during period $s$,
- $k_s$ is a random exogenous variable (call it "average wind velocity") during time $s$, and
- $m_s$ is the aggregate level of waste emissions in period $s$.

In the presence of a tax program, the level of waste discharges will presumably be determined in part by the tax. Let us define

- $m_{is}$ = waste emissions of firm $i$ in period $s$,
- $c_i(m_{is}, \ldots)$ = the total cost function of firm $i$, and
- $t$ = tax per unit of waste emission.

Then, if the firm minimizes its costs, we will presumably have in equilibrium

$$\frac{\partial c_i}{\partial m_{is}} = -t. \tag{4}$$

That is, the firm will adjust waste discharges to the point where at the

16. Other forms of this relationship are obviously possible. For example, $k_s$ and $q_{(s-1)}$ may be additive rather than multiplicative. The facts will presumably vary from case to case, but within wide limits the choice of functional form does not affect the substance of our discussion.

margin the cost increase resulting from a unit reduction of emissions (e.g., the marginal cost of recycling) is equal to the unit emission charge. Using the cost function for the firm and its cost-minimizing emission condition, (4), we can derive a relationship expressing the level of waste discharges of the $i$th firm as a function of the unit emission tax:

$$m_{is} = h_i(t_s). \tag{5}$$

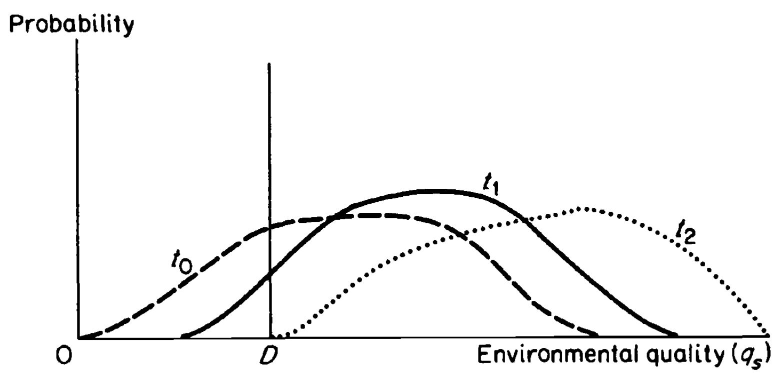
Aggregating over all $i$ firms, we get an aggregate waste-emission function

$$m_s = h(t_s) = \sum h_i(t_s). \tag{6}$$

From equation (6), we can thus determine the total level of waste discharges into the environment in period $s$ associated with each value of $t$, the effluent fee.

Next, suppose we know the probability distribution of $k_s$, our random and exogenous environmental variable ("average wind velocity") in equation (3). For some known value of environmental quality in period $(s - 1)$, we can then determine the distribution function of environmental quality in time $s$ associated with each value of the emission tax, $t$. Figure 1 depicts some probability distributions corresponding to different tax rates.

***Figure 1***




We see that a reduction in the emission tax from $t_1$ to $t_0$ shifts the distribution leftward. Once a lower tax rate is instituted, higher levels of waste emissions become profitable, thereby increasing the likelihood of a period of relatively low environmental quality.

Assume, moreover, that the environmental authority cannot readily change $t$ in response to current environmental conditions so that $t$ is essentially fixed for the period under analysis.[17] Let there also be some accepted "danger standard" (i.e., a minimum acceptable level of environmental quality). We designate this danger standard as $D$ in Figure 1 and assume that the environmental authority is committed to maintaining the level of environmental quality above $D$ at *all* points in time.

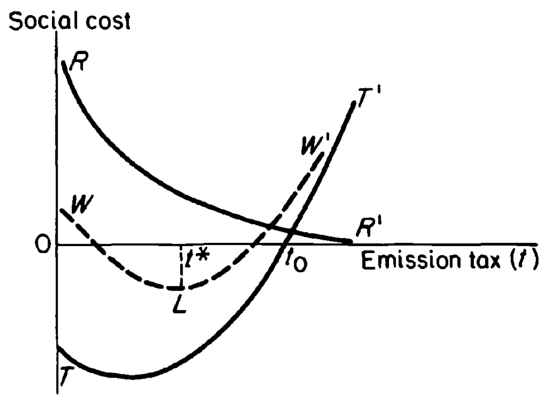
How can the authority achieve this objective at the least cost to society? One method of guaranteeing that $q_s$ will never fall below $D$ is to set the tax rate so high that waste emissions can never, regardless of exogenous environmental influences, reach a value sufficiently high to induce environmental quality to deteriorate to a level less than $D$.[18] In terms of Figure 1, this would require an emissions tax of $t_2$, which shifts the environmental probability distribution rightward until its horizontal intercept coincides with $D$. However, as we suggested earlier, this method of achieving the objective may be an excessively costly one, because it is likely to require unnecessarily expensive reductions in waste discharges during "normal" periods when the environment is capable of absorbing these emissions without serious difficulty. It may be less costly to set a lower emission tax (less than $t_2$ in Figure 1) and to supplement this with periodic introductions of controls to achieve additional reductions in waste discharges during times of adverse environmental conditions (periods of "stagnant air").

In Figure 2, we illustrate an approach to the determination of the optimal mix of emission taxes and direct controls. Let the curve $TT'$ measure the *total* net social cost associated with each value of $t$. There are two components of this social cost. The first is the added costs of production that higher taxes impose by inducing methods of production consistent with reduced levels of waste emissions. This cost naturally tends to rise with tax rates and the associated lower levels of waste discharges. However, we must subtract from this "production" cost a negative cost (or social gain) which indicates the social benefits from a higher level of environmental quality. Over some range of values for $t$ (up to $t_0$ in Figure 2), we might expect the sum of these costs to be negative, that is, the social benefits from improved environmental quality may well exceed the in-

17. Alternatively, we can assume that the response of waste emissions to changes in $t$ is not sufficiently rapid for the tax adjustments in period $s$ to influence significantly waste discharges during that period.

18. It may, of course, be impossible to achieve such a guarantee with any finite tax rate, no matter how high.

*Figure 2*




creased costs of production. However, as tax rates rise and waste discharges decline, the *marginal* net social cost will typically rise. The marginal production cost of reductions in waste emissions (equated in value to $t$) will obviously increase, while we might expect diminishing social gain from positive increments in environmental quality.[19] The $TT'$ curve will, therefore, typically begin to rise at some point and, for values of $t$ in excess of $t_0$ in Figure 2, the net social cost of the tax program becomes positive.

We recall that the environmental authority is committed to the maintenance of a level of environmental quality no lower than the danger point, $D$. We will thus assume that, whatever the level of the emission tax, environmental officials will introduce direct controls whenever necessary to maintain $q$ above $D$. One relationship is immediately clear: the higher the emission tax, the less frequently will environmental quality threaten to fall below $D$ and hence the less often (and less "intensely") will the use of direct controls be required. Controls, like taxes, impose increased costs of production by forcing reductions in waste emissions. Therefore, the more frequent and extensive the use of direct controls,

19. We have drawn $TT'$ with a "smooth" shape ( a continuous first derivative), but there could easily be flat portions of $TT'$ corresponding to ranges of values of $t$ over which the level of waste emissions remains unchanged. Note, however, that even in this instance $TT'$ would still exhibit the general shape depicted in Figure 2 and, most important, would still possess a well defined minimum for some value (or continuous range of values) of $t$.

the greater the increment in production costs they will generate. We depict this relationship in Figure 2 by the curve $RR'$, which indicates that the higher the tax rate the less the reliance and, hence, the lower the costs associated with the periodic use of direct controls to maintain $q$ above $D$.[20]

When we sum $TT'$ and $RR'$ vertically, we obtain the net social cost ($WW'$) associated with each level of the emission tax ($t$) *supplemented* by a program of direct controls which prevents environmental quality from ever falling below the danger point ($D$). In Figure 2, we see that the lowest point ($L$) on the $WW'$ curve corresponds to the cost-minimizing or optimal tax rate ($t^*$) and determines residually the optimal use of direct controls.[21]

We stress that the treatment in this section is purely illustrative. It indicates an approach to the determination of the optimal mix of emission taxes and direct controls. A rigorous solution to this problem requires an explicit recognition of the stochastic element in the curves in Figure 2. The social costs generated by a given tax program depend in part on the values taken by our random exogenous environmental variable ("wind velocity"), so that the curves in this diagram must be regarded in some sense as "averages." More formally, the solution involves the minimization of a stochastic social cost function subject to the constraint that $q \geq D$. Elsewhere we will show how this can be formulated as a nonlinear programming problem, whose solution yields the optimal mix of effluent taxes and direct controls.

## Environmental Policy Tools in Practice

In this section, we want to present some preliminary evidence on the effectiveness of the various tools of environmental policy. Since evidence in the form of systematic, quantifiable results is scarce, we have had to resort in some instances to case studies suggesting only in qualitative

20. Unlike the tax-cost function ($TT'$), the social cost of direct controls does not include a *variable* component related to the benefits from varying levels of environmental quality. Direct controls in this model are used solely to maintain $q$ above $D$. We can treat the social benefits derived from the guarantee that environmental quality never falls below $D$ as a constant (independent of the level of $t$), and we can, if we wish, add this constant to $RR'$ (or to $TT'$ for that matter). The essential point is that we can expect $RR'$ to be a function that decreases monotonically in relation to $t$.

21. Note that the curve $WW'$ may possess a number of local minima. It need not increase monotonically to the right of $L$.

terms the nature of the response to the programs. Many of the findings, however, do seem roughly consistent with the preceding discussion.

### *Price incentives*

While economic theory suggests an important role for price incentives, particularly effluent fees, for environmental control, we really have limited experience with their use. The opposition to proposals for effluent charges has been strong, in some measure, we suspect, because people realize they will be effective and wish to avoid the inevitable costs of environmental protection.[22] Nevertheless, there has been some use of charges, and what evidence is available suggests that effluent fees have in fact been quite successful in reducing polluting activities.

The most striking and important case appears to be the control of water quality in West Germany's Ruhr Valley. The site of one of the world's greatest concentrations of heavy industry, the rivers of the Ruhr Valley could easily have become among the most polluted rivers in Europe. However, since the organization of the first *Genossenschaft* (river authority) in 1904 along the Emscher River, the Germans have been successfully treating wastes in cooperatives financed by effluent charges on their members. There are presently eight *Genossenschaften*. Together they form a closed water-control system which has maintained a remarkably high quality of water. In all but one of the rivers in the system, the waters are suitable for fishlife and swimming. Together, the eight cooperatives collect approximately $60 million a year, mainly from effluent charges levied on their nearly 500 public and private members. The level of charges is based largely on a set of standards for maintaining water quality, although the formulas themselves are rather complicated. As Kneese and Bower point out, the fee formulas do not correspond perfectly to the economist's version of effluent fees ("they violate the principle of marginal cost pricing," p. 251).[23] Nevertheless, the charges, in conjunction with an integrated system of planning and design for the entire river basin, "is a pioneering achievement of the highest order" (Kneese and Bower, p. 253).

There has been a scattered use of effluent fees for environmental protection in North America, and these, to our knowledge, exclusively for

22. For an excellent survey and evaluation of the most frequent arguments directed against programs of effluent fees, see Freeman and Haveman.

23. For a more detailed discussion of the Ruhr experience, see Kneese and Bower, Chapter 12.

the control of water quality. However, this evidence does again point to the effectiveness of fees in curtailing waste emissions. Kneese and Bower cite three instances in which the levying of local sewer charges induced striking reductions in waste discharges.[24] C. E. Fisher reports similar responses to a local sewerage tax in Cincinnati, Ohio. Fees were established in 1953 with the proviso that a rebate would be given to anyone who met a specified set of standards by a certain date. Subsequently, some 23 major companies invested $5 million in pollution control in less than two years to meet these standards.

There also exist three more systematic studies of industrial responsiveness to sewerage fees. Löf and Kneese have estimated the cost function for a hypothetical, but typical, sugar beet processing plant in which cost is treated as a function of BOD removal from waste water. Their results suggest, assuming the firm stays in business, that a very modest effluent charge would induce the elimination of roughly 70 per cent of the BOD contained in the waste water of their typical plant. Likewise, a recent regression study by D. E. Ethridge of poultry processing plants in different cities imposing sewerage fees indicates substantial price responsiveness on the part of these firms. In a total of 27 observations from five plants, Ethridge found that "The surcharge on BOD does significantly affect the total pounds of BOD treated by the city; the elasticity of pounds of BOD discharged per 1,000 birds with respect to the surcharge on BOD is estimated to be −0.5 at the mean surcharge" (p. 352).

The most ambitious and comprehensive study of the effects of municipal surcharges on industrial wastes in U.S. cities is the work of Ralph Elliott and James Seagraves. Elliott and Seagraves have collected time-series data on surcharges, waste emissions, and industrial water usage for 34 U.S. cities. They have put these data to a variety of tests and their findings indicate that industrial BOD emissions and water consumption do indeed appear to respond negatively to the level of surcharges on emissions. In one of their tests, for example, they have pooled their cross-section and time-series observations and, using ordinary least squares, obtained the following estimated equations:

$$T = 13.1 - \underset{(8.5)}{14.6S} - \underset{(79.3)}{120.0G} + \underset{(22.6)}{36.2P} \qquad (7)$$

$$R^2 = .17 \qquad N = 190,$$

24. These involved sewerage fees in Otsego, Michigan, in Springfield, Missouri, and in Winnipeg, Canada. See Kneese and Bower, pp. 168–70.

$$W = 2.2 - 5.2S - 36.8N + 8.6P + 75.1F \qquad (8)$$
$$\qquad\quad (2.9) \quad (24.7) \quad (7.2) \quad (26.0)$$
$$R^2 = .32 \qquad N = 179,$$

where;

$T$ = pounds of BOD per $1,000 of value added in manufacturing;
$S$ = surcharge per pound of BOD in 1970 dollars;
$G$ = price of water (per 1,000 gallons) in 1970 dollars;
$P$ = the real wage rate (per hour) in 1970 dollars;
$N$ = net cost of additional water (per 1,000 gallons) in 1970 dollars;
$F$ = proportion of value added in manufacturing in the city contributed by food and kindred products.

The coefficients on the surcharge variable ($S$) possess the expected negative sign and are statistically significant using a one-tail test at a .05 level of confidence. Using typical values for the variables, the authors estimate the elasticity of industrial BOD emissions with respect to the level of the surcharge to be $-0.8$, and the surcharge elasticity of water consumption at $-0.6$.[25] We are thus beginning to accumulate some evidence indicating that effluent fees can in fact be quite effective in reducing levels of industrial waste discharges into waterways.

In contrast, our experience with charges on waste emissions into the atmosphere is virtually nil. However, there is one recent and impressive study by James Griffin of the potential welfare gains from the use of emission fees to curtail discharges of sulfur dioxide into the air. Using engineering cost data, Griffin has assembled a detailed econometric model of the electric utility industry.[26] The model allows for desulfurization of fuel and coal, substitution among fuels, substitution between fuel and capital (using more capital allows more energy to be derived from a unit of fuel), and for the substitution away from "electricity-intensive" products by consumers and industry. Griffin then ran a series of nine alternative simulations involving differing effluent fees and other assumptions

25. The explanatory power ($R^2$) of the Elliott-Seagraves' equations is not extremely high. Among other things, this reflects the difficulties of accounting for varying industrial composition among cities and for intercity differences in the fraction of waste emissions that enter the municipal treatment system. Ethridge's equations, which use observations on only a single industry (poultry-processing), have much higher $R^2$ (of about .5).

26. In 1970 "power plants contributed 54% of the nation's sulfur dioxide emissions" (p. 2).

based on the estimates provided by the Environmental Protection Agency of the social damage generated by emissions of sulfur dioxide. In all the simulations, substantial net welfare gains appeared. The results were somewhat sensitive to assumptions concerning the availability and cost of fuel gas desulfurization processes about which there is some uncertainty. However, with such techniques available at plausible costs, Griffin's average annual welfare gains ranged from $6.5 to $7.7 billion, and these estimates do not allow for possible shifts to nuclear power sources.

The evidence thus does suggest that effluent fees can be an effective tool in reducing levels of waste emissions. This, of course, is hardly surprising. We expect firms and individuals to adjust their patterns of activity in response to changes in relative costs. It has often been observed that in less developed countries, where wages are relatively low, more labor intensive techniques of production are typically adopted than in higher wage countries. Moreover, in a regression study of the capital labor ratios across the states in the U.S. for 16 different manufacturing industries, Matityahu Marcus found that factor proportions did indeed vary systematically in the expected direction with the relative price of capital in terms of labor. There does seem to be sufficient substitutability in relevant production and consumption activities for modest effluent charges to induce pronounced reductions in waste emissions.[27]

What would be even more interesting is some measure of the relative costs of other control techniques (for example, the imposition of uniform percentage reductions in the waste discharges of all polluters). Evidence on this is scarce. However, one such study has been made, a study of the costs of achieving specified levels of dissolved oxygen in the Delaware River Estuary.[28] A programming model was constructed using oxygen balance equations for 30 interconnected segments of the estuary. The next step was to specify five sets of objectives and then to compare the costs of achieving each of these objectives under alternative control policies. Although effluent charges were not included specifically as a policy alternative in the original study, Edwin Johnson headed a subsequent study using the same model and data. This made possible the comparison of four alternative programs for reaching specified levels of dissolved oxygen in the estuary. The results for two D.O. objectives are presented

27. For a useful summary of estimates of price elasticities for polluting activities, see the paper by Robert Kohn.

28. Federal Water Pollution Control Administration, *Delaware Estuary Comprehensive Study: Preliminary Report and Findings* (1966); a useful summary of this study is available in Kneese, Rolfe, and Harned, Appendix C.

in Table 1, where LC is the least-cost programming solution, UT is a program of uniform treatment requiring an equal percentage reduction in discharges by all polluters, SECH is a program consisting of a single effluent charge per unit of waste emission for all dischargers, and ZECH is a zoned charge in which the effluent fee is varied in different areas along the estuary. As indicated by Table 1, the substantial cost savings of a program of effluent fees relative to that of uniform treatment is quite striking. Moreover, it should be noted that the least-cost programming solution involves a great deal more in the way of technical information and detailed controls than do the programs of fees. The reduced costs from the use of fees instead of quotas thus appear to be potentially quite sizable.

**TABLE 1**
***Cost of Treatment Under Alternative Programs***

| D.O. Objective (ppm) | Program | | | |
|---|---|---|---|---|
| | *LC* | *UT* | *SECH* | *ZECH* |
| | (million dollars per year) | | | |
| 2 | 1.6 | 5.0 | 2.4 | 2.4 |
| 3–4 | 7.0 | 20.0 | 12.0 | 8.6 |

Source: Kneese, Rolfe, and Harned, p. 272.

As we mentioned in the preceding section, effluent fees are, in theory, a more efficient device for achieving standards of environmental quality than subsidies. Fees appear, moreover, to possess a number of practical advantages as well. The design of an effective and equitable system of subsidies is itself a difficult problem. If a polluter is to be paid for reducing his waste emissions, it then becomes in his interest to establish a high level of waste discharges initially; those who pollute little receive the smallest payments.

In practice, subsidies have been used far more extensively in the United States than fees. The federal government has relied heavily on a program of subsidization of the construction of municipal waste treatment plants and on tax credits to business for the installation of pollution control equipment. The serious deficiencies in the first program are now a matter of record in the 1969 Report of the General Accounting Office. The failure to curtail industrial pollution; the subsidization of plant construction but not operating expenses (resulting in many instances of incredibly ineffective use of the facilities); and the inappropriate location of many plants have resulted in the continued deterioration of many major U.S. waterways despite an expenditure of over $5 billion.[29]

Although we have been unable to find any direct evidence on the tax credit program, there is a simple reason to expect it to have little effect. As Kneese and Bower (pp. 175–78) point out, a firm is unlikely to purchase costly pollution control equipment which adds nothing to its revenues; the absorption of $k$ per cent (where $k < 100$) of the cost by the government cannot turn its acquisition into a profitable undertaking.

Thus both theory and experience point to the superiority of effluent charges over subsidies as a policy tool for environmental protection. Finally, we might also mention that, from the standpoint of the public budget, fees provide a source of revenues, which might be used for public investments for environmental improvements, while subsidies require the expenditure of public funds.

### *Direct controls*

As James Krier points out, "Far and away the most popular response by American governments to problems of pollution—and indeed, to *all* environmental problems—has been regulation . . ." (p. 300). Three general types of regulatory policies for environmental control: quotas, prohibition, and the requirement of specified technical standards are stated in the list of tools for environmental control. However, this classification does not indicate the vast number of ways in which these direct controls may be implemented. The directive for polluters to cease certain activities or to install certain types of treatment equipment may come from an empowered regulatory authority, may result from a court order, or might be forced by the citizenry itself through a referendum. Even this is an oversimplification. There are, for example, several methods by which action through the Courts may be initiated (see Krier). Our category of "direct controls" thus encompasses an extremely broad range of policy options. It is beyond the scope of this paper to examine in detail, for instance, the potential of various forms of litigation for effective environmental policy. We shall rather examine somewhat more generally the success or failure of each of these approaches with particular attention to the circumstances which appear to bear on their effectiveness.

The record of regulatory policies in environmental control is not very impressive. This stems at least as much from administrative deficiencies

29. For further documentation of the ineffectiveness and abuses under this subsidy program, see Marx, and Zwick, and Benstock.

in the application of regulatory provisions as in the establishment of the provisions themselves. A successful regulatory policy generally requires at least three components.

(1) A set of rules that, if practiced, will provide the desired outcome. In this case, satisfactory levels of environmental quality achieved at something reasonably close to the least cost.

(2) An enforcement agency with sufficient resources to monitor behavior.

(3) Sufficient power (the ability to impose penalties) to compel adherence to the regulations.

The design of an efficient set of rules is itself an extremely difficult problem. As mentioned earlier, effluent charges have important efficiency enhancing properties. Moreover, the specification of an efficient set of regulatory provisions will generally require at least as much, and frequently more, technical information than the determination of schedules of fees.[30] In addition, experience suggests that substantial transaction costs in terms of resources devoted to bargaining (as noted earlier in the case of the continuing controversy over auto emission standards) may be involved in the rule selection process.

Even an effective set of regulations can only achieve its objective if it is observed. Unfortunately, the history of environmental regulation in the United States is not encouraging on this count. Regulatory agencies have frequently been understaffed and unable, or unwilling, to enforce antipollution provisions. An interesting historical example is the River and Harbors Act of 1899 which prohibits the discharge of dangerous substances into navigable waterways without a permit from the Army Corps of Engineers. As of 1970, only a handful of the more than 40,000 known dischargers had valid permits. Moreover, the newspapers abound with accounts of huge plants which have paid trivial sums (sometimes a few hundred dollars) for serious violations of pollution regulations. Many of the provisions simply have not given the agencies the power they require for enforcement.

Action through the courts has also not proved very effective. Environmental lawsuits, where a plaintiff can be found, have often stretched over years or even decades without resolution. However, even if judicial proceedings were prompt, it is difficult to envision how suits by individual plaintiffs for damages could lead to an efficient environmental policy.

30. In an interesting paper, Karl Göran-Mäler has shown recently that the determination of an efficient set of effluent standards (or quotas) among activities requires at least as much information as that necessary to solve for an optimal set of effluent charges.

Kneese and Bower, while acknowledging the potential of some support from the judicial process, conclude simply that ". . . efficient water quality management cannot be achieved through the courts" (p. 88).

Nevertheless, where enforcment is effective, and it surely has been in a significant number of cases, direct controls can lead to substantial reductions in polluting activities. A variety of regulations in various metropolitan areas have generated large reductions in waste discharges into the atmosphere. The banning of backyard incineration and of the use of sulfur bearing fuels over several months of the year led to significant reductions during the 1950's in smoke, dust, and sulfur oxide discharges into the air shed over the Los Angeles basin. Likewise, tough new regulations in Pittsburgh during the 1940's, requiring the switch from coal fuels to natural gas for heating purposes, resulted in notable improvements in air quality. Strong regulations combined with aggressive enforcement *can* clearly raise the level of environmental quality.[31] The difficulties, of course, are that the improvements may come at an unnecessarily high cost, or, alternatively, may come not at all, if the regulations are themselves inadequate or are ineffectively enforced.

### *Moral suasion and voluntary compliance*

We suggested earlier that, while moral suasion is likely to be an ineffective policy tool over longer periods of time, it may prove quite useful in times of emergency. An interesting illustration of this pattern of response involves voluntary blood donations. In September of 1970, New York City hospitals were facing a blood crisis in which reserves of blood had fallen to a level insufficient for a single day of operation. The response to a citywide plea for donations was described as "fantastic" (*New York Post*, September 4, 1970, p. 3); donors stood in line up to 90 minutes to give blood. The statements by some of the donors were themselves interesting:

"I've never given blood before, but they need it now. That's good enough reason for me."

"I was paying a sort of personal guilt complex."

"It's the least I could do for the city."

31. Direct controls in the form of "technical specifications" for polluting activities may be the only feasible policy instrument, where the monitoring of waste emissions is impractical (or, more accurately, "excessively costly"). For example, if difficulties in metering sulfur dioxide emissions into the atmosphere were to preclude a program of effluent fees (or quotas, for that matter), it might well make sense to place requirements on the quality of fuel used, on the technical characteristics of fuel burners, etc.

And yet within a few months (*New York Times,* January 4, 1971, p. 61), the metropolitan area's blood stocks were again down to less than one day's supply. It was also noted that many donors who promised to give blood had not fulfilled their pledges.[32]

A somewhat similar fate seems to have characterized voluntary recycling programs. Individuals and firms greeted these proposals with substantial enthusiasm and massive public relations efforts. Many manufacturers agreed to recycle waste containers collected and delivered by nonprofit volunteer groups. While the initial response was an energetic one, it seems to have tailed off significantly. "Many (of the groups) disbanded because of a lack of markets or waning volunteer interest" (*New York Times,* May 7, 1972, p. 1 and p. 57). The Glass Manufacturers Institute announced that used bottles and jars returned by the public were being recycled at a rate of 912 million a year, but this represents only 2.6 per cent of the 36 billion glass containers produced each year. Similar reports from the Aluminum Association and the American Iron and Steel Institute indicated recycling rates of 3.7 per cent and 2.7 per cent respectively for metallic containers. The reason for the failure of these programs to achieve greater success is, according to several reports, "that recycling so far is not paying its own way" (*New York Times,* May 7, 1972, p. 1). Experience with recycling programs also points to a danger we mentioned earlier: that these types of programs will be instituted instead of programs with direct individual incentives for compliance. There are a wealth of examples of businesses providing active support for voluntary recycling as parts of campaigns *against* fees or regulations on containers. The *New York Times* (May 7, 1972, p. 57), for instance, cites a recent case in Minneapolis in which the Theodore Hamm Brewing Company and Coca-Cola Midwest, Inc. announced that they would sponsor "the most comprehensive, full-time recycling center in the country." This pledge, however, was directed against a proposed ordinance to prohibit local usage of cans for soft drinks and beer.

A final example of some interest involves a recent attempt by General Motors to market relatively inexpensive auto-emission control kits in Phoenix, Arizona. The GM emission control device could be used on most 1955 to 1967 model cars and could reduce emissions of hydrocarbons, carbon monoxide, and nitrogen oxides by roughly 30 to 50 per cent. The cost of the kits, including installation fees, was about $15 to $20.

32. Other cases we are currently investigating are the formation of car pools both in emergency and "normal" periods to cut down on auto emissions, and the extent of voluntary reductions in usage of electricity during periods of power crises.

Despite an aggressive marketing campaign, only 528 kits were sold. From this experience, GM has concluded that only a mandatory retrofit program for pre-1968 cars, based upon appropriate state or local regulation, can assure the wide participation of car owners that would be necessary to achieve a significant effect on the atmosphere. The Chrysler Corporation has had a similar experience. In 1970 Chrysler built 22,000 used car emission control kits. More than half remain in its current inventory. In fact after 1970 Chrysler had experienced "negative" sales. About 900 more kits were returned than shipped.

The role of moral suasion and voluntary compliance thus appears to promise little as a regular instrument of environmental policy. Its place (in which it may often be quite effective) is in times of crisis where immediate response is essential.

## Concluding Remarks

Our intent in this paper has been a preliminary exploration of the potential of available tools for environmental policy. There is, as we have indicated, a wide variety of options at the policy level with differing instruments being appropriate depending upon the characteristics of the particular polluting activity and the associated environmental circumstances. The "optimal" policy package would no doubt include a combination of many approaches including the prohibition of certain activities, technical specifications for others, the imposition of fees, etc. We hope that the analysis has provided some insight into the types of situations in which certain policy instruments promise to be more effective than others.

Our own feeling, like that of most economists, is that environmental policy in the United States has failed to make sufficient use of the pricing system. Policies relying excessively on direct controls have not proved very effective in reversing processes of environmental deterioration and, where they have, we would guess the objective has often been achieved at unnecessarily high cost. Moreover, to the extent that environmental authorities have used price incentives, they have typically adopted subsidies rather than fees. These subsidy programs have often been ill-designed, providing incentives only for the use of certain inputs in waste treatment activities and by absorbing only part of the cost so that investments in pollution reducing equipment continue to be unprofitable. We still have much to learn at the policy level about the proper use of price incentives in environmental policy.

What emerges from all this is the conclusion that there is considerable validity to the standard economic analysis of environmental policy. There is good reason for the economist to continue to emphasize the virtues of automatic fiscal measures whose relative ease of enforcement, efficiency enhancing properties, and other special qualities are too often unrecognized by those who design and administer policy.

On the other hand, we economists have often failed to recognize the legitimate role of direct controls and moral suasion, each of which may have an important part to play in an effective environmental program. These policy tools may have substantial claims in terms of their efficiency, particularly under circumstances in which the course of events is heavily influenced by variables whose values are highly unpredictable and outside the policy-maker's control. In environmental economics we can be quite certain that the unexpected will occur with some frequency. Where the time costs of delay are very high and the dangers of inaction are great, the policy-maker's kit of tools must include some instruments that are very flexible and which can elicit a rapid response. A tightening of emission quotas or an appeal to conscience can produce, and has produced, its effects in periods far more brief than those needed to modify tax rules, and before any such change can lead to noteworthy consequences. Where intermediate targets, such as emission levels, may have to be changed frequently and at unforeseen times, fiscal instruments may often be relatively inefficient and ineffective.

In sum, as in most areas of policy design, there is much to be said for the use of a variety of policy instruments, each with its appropriate function. Obviously this does not mean that just any hybrid policy will do, or that direct controls are always desirable. Indeed, there are many examples in which their use has provided models of mismanagement and inefficiency. Rather, it implies that we must seek to define particular mixes of policy that promise to achieve our environmental objectives at a relatively low cost.

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## COMMENT

**Charles Upton, The University of Chicago**

The authors consider a variety of policy instruments for regulating environmental quality. For a variety of reasons which they—and I—find compelling, they reject proposals such as subsidies and moral suasion and suggest instead a mixture of pollution taxes and direct controls. They conclude that although "environmental policy in the United States has failed . . . to make sufficient use of the pricing system . . . economists have too often failed to recognize the legitimate role of direct controls . . . which may have an important part to play in an effective environmental program."

According to Oates and Baumol, an important reason for using direct controls is the stochastic nature of environmental quality. Since a fixed tax will result in periods of low environmental quality, direct controls should—again, according to the authors—be employed on those occasions. Yet this argument is an invalid comparison between controls which can be varied and a tax structure which cannot. Their argument essentially rests on the quite strong assumption that pollution taxes cannot be changed to deal with "emergencies," but the level of direct controls can be changed.

But one can change the level of taxes. Indeed, one should. For example, air quality in urban areas is usually lower in winter than in summer, suggesting the use of a two-part emissions tariff, and not a uniform emissions tax throughout the year supplemented by direct controls during the winter and summer months.

The notion of a differential tax can be further extended to other cases. For example, air quality drops during "thermal inversions" and so presumably does the optimal level of emissions. Oates and Baumol call for direct controls under such circumstances. But a temporary rise in the emission tax—of sufficient magnitude—could achieve the same reduction in emissions as direct controls. To be sure, one could not impose a "thermal inversion surcharge" until it could be determined that an inversion had occurred. Hypothetically if this took one day, taxes would not be useful on the first day. But then one could not impose emergency controls until it was determined that an inversion had occurred.[1]

In sum, it is important to distinguish between cases in which one knows that a parametric shift in the environmental quality has or will occur and those which are unpredictable. In the first case, either taxes or controls can be used; the well-known efficiency properties of taxes to which the authors allude suggest that taxes are appropriate. In the second case, there is no emergency policy which will be able to affect emissions.

As the authors admit, their argument rests on the assumption that the taxes cannot be changed as rapidly as direct controls can be imposed. However, they do not address the question of how one could institute controls and have them effective if it is impossible to use taxes.[2]

Integral to Oates and Baumol's discussion of the uncertainty issue is their analysis of the environmental authority's objective, which they take to be meeting some prescribed standard of environmental quality. The true objective of the authority is to maximize the value of environmental quality net of emission treatment costs. Since this rule may prove difficult to implement, it may sometimes be useful to adopt as a proxy an objective of meeting a prescribed environmental quality standard.[3] However, if the shifts in the parameters such as wind conditions that affect environmental quality are truly stochastic and emission levels cannot be changed in response, it may be impossible to meet any standard with certainty.

Even if it is possible to change emissions to meet a given environmental quality standard, optimal social policy may be to accept variations

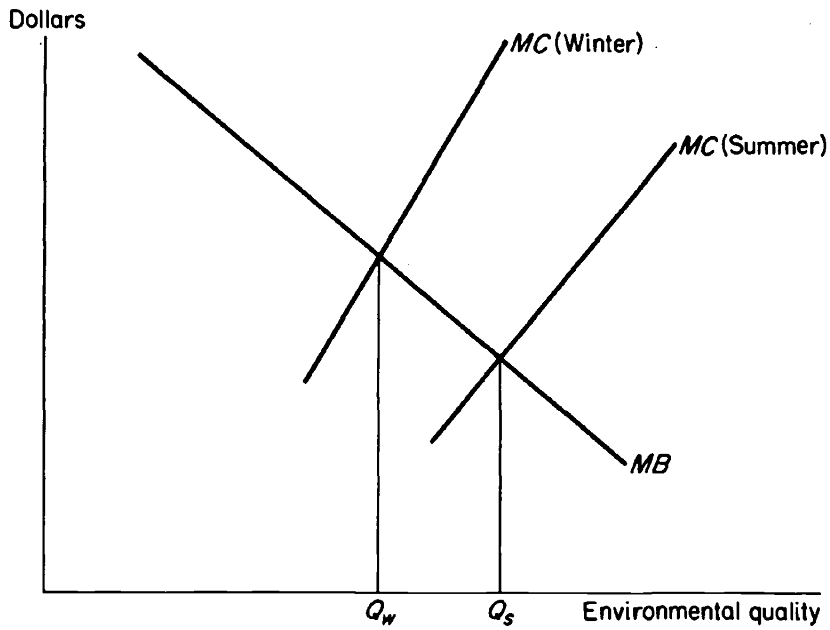
1. One could however announce a tax schedule which would be applied whenever inversions occur, even though there might be a delay in determining that an inversion had occurred. But unless one assumes that firms have superior ability to recognize the start of an inversion, this plan will not make the tax any more effective since firms will respond only when they believe an inversion has begun.

2. Two caveats on this point. First, firms will set the short run marginal cost of reducing pollution equal to the tax. So if only a short-run reduction in emissions is desired, one may require a tax higher than the one which would be required if a permanent reduction in emissions was desired (assuming, of course, that the short-run marginal cost of reducing emissions is higher than the long-run marginal cost).

Second, there is another problem if the regulatory authority is unsure of the effects any given tax or control schemes will have on emissions. This is a difficult problem which has no simple solution. However, this is not the problem taken up in the paper.

3. To be more precise, it is sometimes useful to analyze pollution control as the dual tasks of meeting a quality standard at minimum cost and determining the optimal standard.

*Figure 1*




in environmental quality. For a simple example, consider again the case of winter and summer months. One factor behind the difference is the wintertime demand for heating. If we interpret this as meaning that the marginal cost curve for environmental quality shifts to the left in winter months and if, for simplicity, we assume that the marginal benefit curve is the same for both seasons, it is optimal to have seasonal changes in environmental quality standards. As Figure 1 illustrates, the optimal level of environmental quality is $Q_w$ in the winter and $Q_s$ in the summer.

Another difficulty with emission taxes raised by Oates and Baumol lies in their application to oligopolies. Since the authors attach only minor importance to this issue and since an oligopoly is an ill-defined concept, my comments will be brief. First, note that in the simple case of a profit-maximizing monopolist, emission taxes are more efficient than direct controls. An emission tax will induce any profit-maximizing firm to reduce emissions and substitute hitherto more costly factors of production, thus minimizing the total social cost of producing output. Direct controls probably will not do that. To be sure, a monopoly will not necessarily pass along the full cost of emission control to the consumer (as would a competitive firm), but this will be true whether the costs arise from direct controls or control via emission taxes. So, on balance, the differences lie in favor of emission taxes.

Two additional cases are those of the regulated industry and an oligopoly which has objectives other than maximizing profits. However, there is a question of their relevance: to what extent does regulation matter and do oligopolies exist? But these are old issues and there seems little point in repeating the arguments here.

Another regulatory device which Oates and Baumol do not consider is nonintervention. Indeed they begin their paper by specifically ruling out this possibility, claiming that the transactions costs involved in the provision of environmental quality by private action make such a solution impossible. The transactions costs involved in private action do not constitute an absolute barrier to the provision of public goods by private action; they mean public goods might thereby be undersupplied. Although little is known about the economics of political processes, it is possible that political control of environmental quality could mean an oversupply of environmental quality. If so, a policy of nonintervention which results in an undersupply of environmental quality may well be preferable to a policy of government intervention which provides an oversupply. The expected cost of an undersupply must be weighted against the costs of a possible oversupply and inefficient production of environmental quality possible with a nonmarket solution.[4]

It is even more difficult to reject *a priori* a policy of nonintervention by the federal government when one considers the possibility of local control. Regional differences in factor endowments suggest that there should be regional differences in the provision of environmental quality. Indeed, even were there no differences in factor endowments, differences in individual tastes would argue for cities providing different levels of environmental quality.

Surely, almost all of the externalities from, for example, Pittsburgh's air pollution are internalized within Pennsylvania, and there would seem little necessity for federal intervention to set air quality standards. To be sure, there are some cases like the Chicago SMSA where problems cross state lines. However, the number of negotiators required to inter-

4. Or to put it another way: most economists would agree *a priori* that there is some inefficiency in a water pollution control act which called for zero effluents, and it is conceivable that the social welfare would be lower than it would be under a policy which permitted unlimited discharges of pollutants.

nalize interstate externalities is sufficiently small (the governors of Illinois and Indiana) that the Coase solution seems appropriate.

Thus, it is difficult to rule out a policy of nonintervention at the federal level. Although a case can be made for economies of scale implicit in federal control,[5] these gains must be weighted against the welfare loss from the provision of uniform levels of environmental quality (which seems implicit in federal control).

5. A common example of these economies is the possible cost to the automobile industry of dealing with fifty state automobile emission standards.

# The Resource Allocation Effects of Environmental Policies

G. S. Tolley, *The University of Chicago*

Once, if you asked an economist what to do about externalities, the answer was sure to be: tax them. A number of questions have been raised about the traditional tax approach, and nontax approaches have continued to find more favor in actual policy. These developments help explain why interests of economists have widened to direct limitations on outputs and inputs, zoning, salable rights, legal recourse and a variety of other formal and informal arrangements (see Bohm, Buchanan, Ciriacy-Wantrup, Clarke, Dales, Kamien, Kneese, Mishan, Tideman, Tolley, Turvey, Upton, Wolozin, Wright, Zerbe).

The traditional economics literature on taxes and most of the recent literature on nontax policies have been qualitative. How to measure the benefits and costs of the policies has been neglected. The measurement task is often taken to be the obvious gathering of facts, not recognizing deficiencies in concepts needed for their collection and interpretation. Previous literature has tended to deal with one policy at a time. Different forms of control on polluting firms, procrusteanism of imposing uniform requirements, and spatial arrangements have been particularly neglected. A framework is needed for systematically comparing policies and indicating how effects depend on underlying demand and production conditions.

With these concerns in mind, the first section of this paper considers benefits from reducing a single negative externality. Results are obtained on how to use information on physical effects of the externality, on de-

NOTE: Helpful comments were made by Gardner Brown, Charles Upton, Richard Zerbe and University of Chicago urban economics workshop participants.

fensive acts of those harmed, and on factor rewards. Several needs for modifying benefit estimation practices emerge.

The second section considers the costs of reducing an externality through (a) emission regulation, (b) requirement of emission control equipment, (c) restrictions on inputs and (d) restriction on output. General cost expressions are developed, the policies are compared using algebraic forms, and applications of current interest are discussed.

After a third section on how to bring together benefits and costs with identical factors, the fourth section considers losses from identical requirements where there are uncertain multiple externalities with nonuniform factors. This section gives most attention to nonuniformity within a shed where physical effects are interrelated. Quantitative restrictions, taxes, salable rights and zoning—all of which are the same for a single externality under certainty—are compared. The final major section deals with location of activity between sheds giving attention to land bids needed for optimum location incentives.

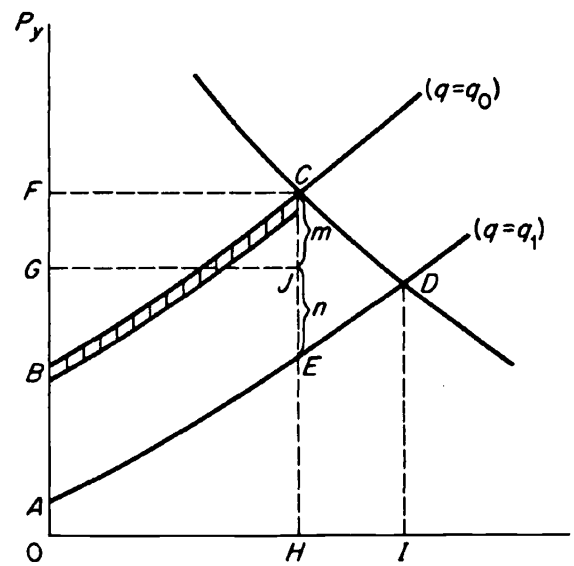
## Damages and Defensive Acts

### *Firms*

If reducing effluent will lower production costs of downstream firms, one part of benefits is the lowering in costs of producing the prevailing level of output downstream. Since the change in the total cost of producing the output is the sum of changes in marginal costs, this part of the benefit is equal to the sum of changes in the marginal cost of production from zero output up to the prevailing output downstream. If the demand curve facing the downstream firms is not completely inelastic, the lowering of marginal cost curves will increase the output at which marginal cost equals price. On each increment of increased output, there is a net gain equal to the difference between the demand value of the increment and marginal cost of production. The part of the benefit resulting from increased output of the downstream firms is the sum of the differences on each increment between the old and new output.

In Figure 1, $H$ is output of the downstream firms prevailing before the reduction in the externality. The part of the benefits which is the change in cost of producing the prevailing output is the difference in marginal costs from zero up to $H$, or the area $ABCE$. As a result of reducing the externality, output expands to where the new marginal cost curve intersects the demand curve at output $I$. The part of the benefit due to additional output is the sum from $H$ to $I$ of incremental differ-

*Figure 1*




ences between demand value and marginal cost, or the area *CDE*. The total benefit from reducing the externality is the total of the areas *ABCE* and *CDE*, or *ABCD*. This is the standard result that benefit is equal to the change in producer plus consumer surplus [10].

Let the demand curve facing producers of a commodity $y$ which is adversely affected by pollution be

$$p_y = F(y). \tag{1}$$

The production function is

$$y = y(z, q), \tag{2}$$

where $z$ refers to inputs controlled by the producer. The variable $q$ is a public good such as quality of water or air and is not controllable by the producer of $y$. The system is completed familiarly by equating marginal value of $y$ to input price times inputs required to produce an extra unit of $y$:

$$p_y = p_z/y_z(z, q) \tag{3}$$

where $y_z(z, \hat{q})$ is the partial of (2) with respect to $z$ and is the marginal product of $z$.

The right side of (3) is the marginal cost of producing $y$. Solving (2) for $z$ and substituting into (3) gives marginal cost of producing $y$ as a function of $y$ itself, for different amounts of the public good $q$. Correspondence with the graph is established by noting that Figure 1 shows two of these marginal cost schedules and the demand curve (1).

### *Households*

A first procedure possible for households would be to let environmental quality enter the utility function. A second procedure, to be followed here, is to exclude environmental quality from the utility function and let it be an input shifting the production for other goods which do enter the utility function. For instance, instead of entering the utility function, air quality is a production function shifter affecting goods which enter the utility function such as condition of buildings, clothing cleanliness, and freedom from respiratory and eye symptoms.

Under this second procedure the household problem is to maximize satisfaction from goods affected by environmental quality. Air quality affects inputs devoted to obtaining the goods. Among several advantages of this procedure, the analysis of benefits from improving environmental quality for the household becomes identical to that just given for the firm, permitting an institutionally neutral approach not arbitrarily affected by whether activity, such as laundering, takes place in the firm or household.

As applied to the household, Figure 1 shows how a lowering of environmental quality raises the marginal cost curve for attaining the goods on the $x$-axis which are affected by environmental quality. Defensive measures and other time and money responses to pollution are cost outlays devoted to the goods. The total cost outlay is the sum of the marginal costs up to the output achieved, or *OADI* at the higher level of environmental quality and *OBCH* at the lower level. The costs include housewife time in the case of cleanliness, and they include medical bills and time lost from work in the case of health. Even under adverse environmental conditions, medical bills and time lost from work are subject to choice since options would be not to have medical treatment and not to stay away from work, in which case health would be reduced below the low best level *OH* achievable with the reduced environmental quality.

To derive equations (1), (2) and (3) for the household, note that in contrast to the firm problem max $p_y y - p_z z + \mu_y[y - y(z, q)]$, the problem for the household is max $U(y, z') + \mu_y [y - y(z, q)] + \mu_z(Z - p_z z - p_{z'}z')$ where $Z$ is total wealth and $z'$ is all other goods. For the firm, given the demand curve (1) and the production function (2), the derivation of (3) by Lagrangian maximization is straightforward. For the household, the production function (2) is given, but no price of $y$ or demand curve are given. Letting $p_y$ be the internal demand price or amount of money the household is willing to give up to get an extra unit of $y$, this amount of money must be such that the utility from spending an extra dollar on $y$ and $z'$ are the same, $U_y/p_y = U_{z'}/p_{z'}$ (see Becker), or rearranging $p_y = (U_y/U_{z'})p_{z'}$. Using the budget constraint to substitute out $z'$ and taking $p_{z'}$ as given, the foregoing price condition gives the internal demand curve ($p_y$ as a function of $y$) which is equation (1) for the household. Using the price condition together with the Lagrangian solution to the household maximization problem gives $p_y = p_z/y_z$ which states that marginal valuation is equated to marginal cost of producing $y$ and is equation (3), completing the demonstration that the same system is obtained for the firm and household.[1]

### *The expenditure approach*

The effects of air quality on household expenditures are often estimated to gain an idea of the benefits of air pollution reduction (see Ridker).

1. The Lagrangian solution to the household maximization problem in the text can be written $U_z = U_y y_z$. Indicating the variables appearing in each function and using the production function $y = y\ (z, q)$ to eliminate $y$, the equilibrium condition for the text formulation—where environmental quality does not enter the utility function—is $U_{z'}(z', z, q) = U_y(z', z, q)y_z(z, q)$.

If environmental quality does enter the utility function, expenditures on things $z$ affected by environmental quality are considered to be expenditures on goods with utility, instead of being expenditures on inputs. The formulation of the household maximization problem becomes max $U(z', z, q) + \mu_z(Z - p_z z - p_z z')$ for which the equilibrium condition is $U_{z'}(z', z, q) = U_z(z', z, q)$.

Compare the right sides of the equilibrium conditions under the two different formulations. If environmental quality enters the utility function as in the formulation just given in this footnote, the marginal utility of things affected by environmental utility is seen, in terms of the text formulation, to be a product whose unobserved components are the marginal utility of the output affected by environmental quality times the marginal productivity of inputs in producing the output.

If environmental quality enters the utility function, activities which are responses to pollution must enter as related goods. They have to be analyzed in terms of "substitutability" with environmental quality, which seems arbitrary and prevents consideration of the more ultimate household satisfactions $y$. Because of the suppression of ultimate satisfactions, information on health and other physical measures of well-being cannot be used in benefit estimation using the formulation, given in this footnote, where environmental quality enters the utility function.

The change in expenditures is the difference between $OADI$ and $OBCH$. Since the two costs have $OAEH$ in common, the change in expenditure is $ABCE$ minus $HEDI$. $ABCE$ is the change in costs necessary to maintain the level of $y$ at $H$, and is equal to $ky\Delta C$ where $\Delta C$ is the vertical shift in marginal cost at $H$ and $k$ is the ratio of the average of vertical shifts at all the previous values of $y$ relative to the shift at $H$. Expressed as a percentage of the value of $p_y y$, $ABCE/p_y y$ is $k\Delta C/C$ since at the margin price $p_y$ equals marginal cost $C$. The area $HEDI$ is $p_y\Delta y$ minus the area $EJD$, which in turn is $n\Delta y/2$. Making use of the fact that the elasticity $\delta$ of the marginal cost curve is $(\Delta y/n)(C/y)$ and again expressing results as a per cent of $p_y y$, $HEDI/p_y y$ equals $(\Delta y/y)[1 - (\Delta y/y)/2\delta]$. To find $\Delta y/y$, making use of the fact that the elasticity of demand $\beta$ is $(-\Delta y/m)\ (p_y/y)$ and of the expression for $\delta$, obtain $m + n = \Delta C$ as a function of $\Delta y$. Solving for $\Delta y$ and dividing by $y$ gives $(\Delta y/y) = (\Delta C/C)$ $\{1/[(1/\delta) - (1/\beta)]\}$. These results may be combined to obtain changes in expenditures as a percentage of value

$$\Delta E/p_y y = (\Delta C/C)\{k + \beta[1 + \beta(\Delta C/C)/2(\delta - \beta)]/[1 - (\beta/\delta)]\}. \quad (4)$$

The special case of a horizontal marginal cost curve is

$$\Delta E/p_y y = (\Delta C/C)(1 + \beta) \quad \text{if} \quad \delta = \infty \quad \text{and} \quad k = 1. \quad (4\bar{C})$$

Extra effort over a wide range should continue to yield substantial effects on physical characteristics defining cleanliness, thus suggesting that marginal cost is fairly constant for attaining these attributes and thereby proving ($4\bar{C}$) a good approximation for cleanliness. A commonly reported finding is that higher pollution does not lead housewives to devote more effort to cleaning. Contrary to the inference one might be tempted to draw that there are no cleanliness benefits, a possibility is that the elasticity of demand for cleanliness is unity ($\beta = -1$) since this is the only condition making the right side of ($4\bar{C}$) zero. Even with error in answers, the lack of perceptible expenditure response under extreme pollution conditions suggests a downward response of cleanliness demanded to a rise in its cost ($\beta < 0$).

The area $ABCE$ is $-k\Delta C/C$ as already noted. The additional benefit area $CDE$ is $(\Delta y)(\Delta p)/2 - (\Delta y)(\Delta C - \Delta p)/2$ or $-(\Delta y)(\Delta C/2)$. Adding the two areas, making use of the solution for $\Delta y/y$ and dividing by $p_y y$ gives benefits as a fraction of product value:

$$\Delta B(y)/p_y y = -(\Delta C/C)\{k + (\Delta C/C)/2[(1/\delta) - (1/\beta)]\}, \quad (5)$$

which reduces to

$$\Delta B(y)/p_u y = -(\Delta C/C)[1 - \beta(\Delta C/C)/2] \quad \text{if} \quad \delta = \infty \quad \text{and} \quad k = 1. \qquad (5\overline{C})$$

### *Cleanliness*

Comparing ($4\overline{C}$) and ($5\overline{C}$) makes clear that zero change in expenditure ($\beta = -1$) does not indicate that benefits are zero. Under conditions that seem typically satisfied of rises in marginal costs less than one hundred per cent and absolute value of elasticity of demand of one or less, benefits in ($5\overline{C}$) are the same order of magnitude as the use in marginal cost.

Suppose marginal cost of maintaining household cleanliness is raised twenty-five per cent due to heavy pollution in a neighborhood. Assuming $\beta = -1$, $\delta = \infty$, and $k = 1$, ($4\overline{C}$) indicates change in expenditures is zero while ($5\overline{C}$) indicates costs (negative benefits) are 28.1 per cent of the total expenditures for cleanliness. If the yearly value of materials and time expended on cleanliness is \$1,000 per household, the pollution costs are \$281 per household of \$2.81 million per year for a neighborhood of 10,000 households showing that pollution costs may be substantial even in the absence of an observed expenditure response.

### *Medical services*

Instead of being horizontal, marginal cost curves may be upward sloping and may be shifted nonuniformly. For a disease, the abscissa is an index of freedom from the disease symptoms. In the absence of pollution, rising marginal costs might be encountered only at a health level far to the right. With air pollution, the marginal cost curve would be shifted up and could become more steeply sloped at a lower level of health. For a disease with high treatment costs or debilitating effects, the relative rise in marginal cost $\Delta C/C$ may be high at $H$, and change in marginal cost at $H$ may be greater than average change in marginal cost on the units of $x$ to the left of $H$. At the lower level of cost, which Figure 1 indicates to be the relevant cost curve for the calculation, the supply curve might be highly elastic. The fact that expenditures are observed to increase is suggestive that the demand elasticity is less than one. If $\Delta C/C = .10$, $k = 2$, $\delta = 7.5$ and $\beta = -.5$, (4) and (5) give $\Delta E/p_u y = .153$ and $-\Delta B/p_u y = .205$.

At the extreme, if no defensive expenditures are possible, the marginal cost curves become vertical lines. With no observed changes in expenditures, the benefits are determined entirely by the slope of the demand curve ignored in the expenditure approach.

### *Mortality*

The model of (1) – (3) can guide studies of physical effects of pollution. The benefit of a one-unit change in environmental quality is the sum of the effects on marginal costs of all units of $x$ up to the observed level, illustrated as the sum of the small quadrangles in Figure 1. In view of (3), the sum is $\int_0^y [d(p_z/y_z)/dq]dY$. Carrying out the differentiation under the integral sign, substituting in $p_y = p_z/y_z$ and making use of $dY = y_z dZ$ to change the variable of integration gives as the sum of quadrangles $\int_0^z p_y y_{zq} dZ$ which equals $p_y y_q$ and says that the benefit from a one-unit change in environmental quality is the value of a unit of $y$ times the effect of the environmental change on $y$. Another way of representing the benefit area *ABCE* plus *CDE* thus is

$$B(y) = \int_{q_0}^{q_1} p_y y_q \, dQ, \tag{6}$$

which suggests how measures of pollution effects $y_q$ on physical attributes should enter benefit estimation. Note that $y_q$ is a marginal productivity concept holding all other inputs $z$ constant.

Suppose the only health effects of air pollution are small effects on probability of survival, which probability is the good measured as the abscissa. Suppose the change in probability is so small that the marginal value of survival is not affected (demand curve flat over the range being considered) and there are no defensive measures (marginal cost curves perfectly vertical). Then equation (6) indicates the appropriate measure of benefits is the observed change in survival expectancy times $p_y$, a measure of the value of life.

### *Morbidity*

If the demand curve is not flat or if defensive expenditures are undertaken, as is the rule for morbidity, in applying (6) one must first allow for changes in marginal value $p_y$ along the demand curve. Econometric studies are conceivable estimating sacrifices people are willing to make to avoid physical effects as a way of facing this valuation problem. Second, the effect of the expenditures on physical attributes needs to be subtracted out to obtain the sole effect $y_q$ of pollution or physical attributes. The observed association between morbidity and pollution understates the benefits from pollution reduction since morbidity is reduced by defensive expenditures. Clinical data might throw light on effects of defensive measures and might also be used to directly estimate $y_q$ if situations can be found of the same defensive measure under different pollution levels. For damages to materials, as opposed to human beings, controlled observations are promising.

### *Land and labor returns*

The problems of goods definition encountered in analyses of expenditures and physical effects do not arise in the factor rewards approach. Since any environmental effect which is less than nationwide can be escaped by moving, given consumer knowledge the shaded benefit area *ABCE* plus *CDE* can be expected to show up as a factor reward difference, estimable without the conceptual problems surrounding Figure 1. The idea that air quality differences within a city are reflected in land values, provides a rationale for benefit estimates based on econometric studies of pollution effects on residence values (see, for example, Crocker and Anderson).

Environmental effects pervading an entire city are not mutable by a residence change within the city. However, because they are mutable by moving between cities, they can be expected to show up in differences in wages between cities. In contrast to work on land values, there has been little estimation of environmental effects on wages. To indicate possibilities, a preliminary result by Oded Izraeli is a regression of deflated wages of laborers in SMSAs on human capital, public expenditure and environmental variables. The $R^2$ is .81. Regarding air pollution, the elasticity of wages with respect to sulfates is .09 and with respect to particulates is .01. Both signs are as expected, and the coefficient of sulfates is significant above the 5 per cent level.

## Productivity of Pollution

Turning from benefits to costs, the costs of pollution reduction consist of losses in satisfaction from commodities whose production causes pollution. In the absence of incentives to control pollution, pollution can be ignored as a consideration in production of these commodities. The traditional theory of production without controls suffices. If pollution is reduced from the point of no control, losses may be incurred because less of the product is produced and it is produced in a higher cost way. While the existence of pollution control costs has been recognized, the reasons for losses have not often been considered explicitly. At most, even in theory, a cost schedule for reducing emissions is usually assumed as a starting point without being derived.

To find out why and by how much the costs of different methods of control differ, in addition to needing to know about product demand and the traditional production function for product output, knowledge is needed about an additional production function indicating how pollutant emissions depend on producer decisions. Specifically, emissions depend on waste producing inputs and pollution control inputs. In this section, it will be shown that the production function for emissions is a key determinant of differences in policy costs. Under an emission regulation policy, producers can choose between adjusting waste producing inputs and pollution control inputs. Because they can choose, this policy is least costly. Under requirement of pollution control devices, producers have incentives to reduce emissions using the devices but not to adjust waste producing inputs; whereas under regulation of waste producing inputs, these incentives are reversed. The relative costs of the latter two policies depend on the marginal effects on emissions of pollution control devices and waste producing inputs. The most costly policy of all is restriction of product output, under which the only reason for emission reduction is a fall in output, with no action being taken to reduce emissions caused by any given output.

Policy effects can be analyzed as responses to incremental exogenous changes. The marginal emission benefit is achieved by allowing emissions to increase one unit through incremental changes in a policy, as for example, the benefit from relaxing restriction on waste producing inputs just sufficiently to allow emissions to increase by one unit. The cost of a policy (measured as benefit foregone) is the sum of marginal benefits from allowing emissions to increase from their level under the policy up to the uncontrolled level. Since uncontrolled emissions are pushed to the point where they have no further value, marginal benefit is zero from allowing emissions to increase at the no control equilibrium under any policy. The magnitude of total benefits foregone depends on how rapidly marginal benefits decline in approaching the no control equilibrium. Thus comparing policies requires comparing *change in marginal benefits* as emissions are allowed to increase. After presenting the no control model, a model of producer decision will be set up for each policy, from which will be derived marginal benefits, change in marginal benefits and the resulting policy costs.

### *No control*

Let the demand curve for a commodity whose production causes pollution be

$$p_x = D_x(x) \tag{7}$$

where $p_x$ is price or value of an extra unit of $x$. For a producer having no effect on price, $p_x$ is given implying the slope $D_{xx}$ is zero. If output affects price, as for a local utility, it will be assumed that regulation enforces marginal cost pricing, leaving for future analysis other pricing policies. If the polluting entity is a household, the price is marginal valuation within the household. The incentive is then to maximize the area under the curve less foregone expenditures in producing the commodity.

In the absence of expenditures to reduce emissions, the only physical relation of concern to the producer is the traditional production function explaining product output:

$$x = x(u, f), \tag{8}$$

where $f$ consists of inputs such as coal or gasoline which are polluting and $u$ consists of all other inputs that increase the production of $x$. The assumed demand conditions imply familiar incentives to make output price times marginal physical product equal to input price. The problem is max $\int_0^x D(X)dX - p_f f - p_u u + \lambda_x[x - x(u, f)]$, whose solution by Lagrangian maximization gives:

$$p_x x_u = p_u, \tag{9}$$

$$p_x x_f = p_f, \tag{10}$$

where $x_u$ and $x_f$ are the partials of (8) and $p_u$ and $p_f$ are the input prices.

Equations (7)–(10) determine commodity price, output, and the two inputs in the absence of efforts to control pollution. They describe market behavior toward pollution assuming there are free rider and other impediments to private negotiations. To consider how changes will affect this system, a generalized displacement can be represented by taking the differential of each equation. The resulting coefficients of differential changes are:

$$
\begin{array}{cccc}
dx & dp & du & df \\
\end{array}
\qquad\qquad dE
$$

$$
\begin{bmatrix} D_{xx} & -1 & 0 & 0 \\ -1 & 0 & x_u & x_f \\ 0 & x_u & px_{uu} & px_{uf} \\ 0 & x_f & px_{fu} & px_{ff} \end{bmatrix} = \begin{bmatrix} e_x \\ e_p \\ e_u \\ e_f \end{bmatrix}. \qquad \begin{matrix} (7') \\ (8') \\ (9') \\ (10') \end{matrix}
$$

The determinant on the left hand side will be denoted $M$. On the right hand side $dE$ refers to any exogenous change. The coefficients $e_x$, $e_p$, $e_u$ and $e_f$ indicate the effect, if any, of the change in each equation. With no controls, exogenous changes refer to shifts in demand function, production function or factor prices. With controls, the exogenous changes can also refer to incremental changes in a policy control.

### *Emission regulation*

The production function specifying emissions is:

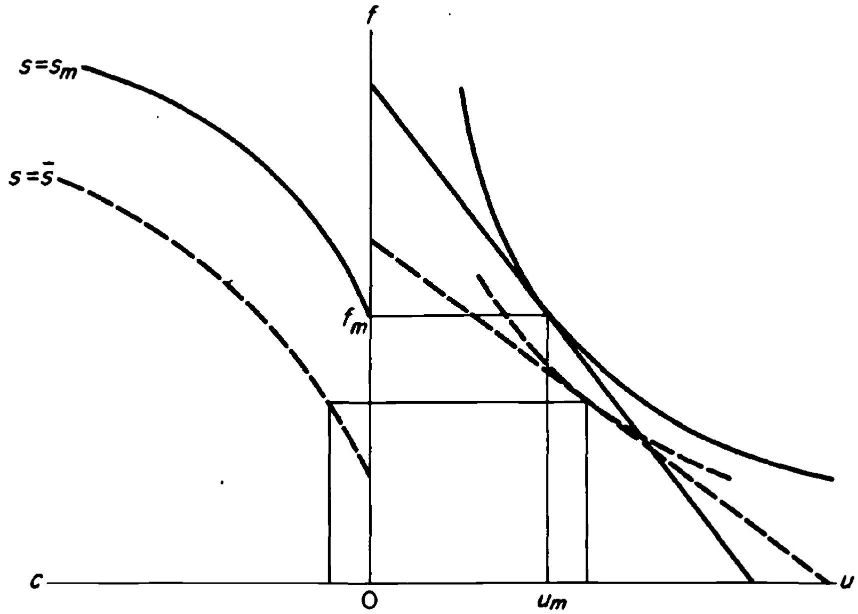
$$s = s(f, c) \tag{11}$$

where $c$ refers to inputs devoted to controlling emissions. The polluting inputs $f$ increase emissions, and control inputs $c$ decrease them.

One set of policies of interest theoretically and practically operates on emissions $s$; e.g., $s$ is limited to some maximum amount. This type of policy induces producers to use less polluting inputs and to incur emissions control expenditures. With a given limit of allowable emissions, the marginal cost to the producer of adding a unit of polluting input is the input price *plus* the cost of controlling the emissions from the extra input. The extra emissions are given by the partial of the emission relation (5) with respect to $f$, or $s_f$. For example $s_f$ is pounds of smoke resulting from an extra ton of coal. Since adding one unit of precipitator inputs will reduce pounds of smoke emitted by $-s_c$, precipitator inputs required per pound of smoke reduction are $-1/s_c$. Multiplying the precipitator inputs required per pound of smoke by the extra pounds of smoke gives $-s_f/s_c$, the control inputs required to keep emissions from increasing. The magnitude $-s_f/s_c$ is the marginal rate of substitution between control inputs and polluting inputs and will be denoted $\sigma$. The cost of controlling emissions from an extra unit of polluting inputs is this amount times the price of control inputs or $p_c\sigma$. In contrast to (10), the conditions governing use of polluting inputs becomes

$$p_x x_f = p_f + p_c\sigma. \tag{10s}$$

*Figure 2*




The control costs $p_c\sigma$ add to the marginal cost of using polluting inputs, giving incentives to use less of them.

Equations (7)–(9), (10s) and (11) describe the system under the regulation controlling $s$. As compared with the free market system, there is an additional endogenous variable $c$. In the free market system, there are no incentives to use control inputs ($c = 0$). If the regulation is effective, $c$ will take on a positive value.

Free market and the control situation are compared in Figure 2. The right side contains iso-product curves for $x$. The free market inputs $f_m$ and $u_m$ are determined in the usual manner by tangency between an iso-product curve and factor cost line having slope $-p_u/p_f$. With an emission standard, the slope of the factor cost line is the dashed line $-p_u/p_f + \sigma p_c$. The left side of Figure 2 contains iso-emission curves. Taking the differential of (11) holding $s$ constant and solving for $df/dc$ gives slope of iso-emission curve $-s_c/s_f$, the reciprocal of $\sigma$. If allowed emissions are lowered from the free market level $s_m$ to $\bar{s}$, the producer contemplates positions along the new iso-emission curve, each position implying a different slope of marginal factor cost line on the right side. For any choice of $c$ on the left side, an optimum production decision for $x$ on the right side can be found. Suppose the producer was temporarily at some non-equilibrium point on the iso-emission curve. This would determine $\sigma$ and hence the slope of the marginal factor cost line whereupon, dividing (10s) by (9) an expansion path for $x$ and $u$ could be found. The producer would proceed along the expansion path until marginal cost equalled marginal gain. Having found this position, he could ask whether further gains could be made by changing emission control expenditures $c$ thus changing allowable fuel use. Since $\sigma$ units of $c$ are required to increase fuel use by one unit while still being able to meet the emission standard, the emission control cost required to expand fuel use by one unit is $p_c\sigma$. The gain from the expansion of fuel use is the marginal revenue from additional fuel use less the resource cost of the fuel or $p_x x_f - p_f$. The producer will be in full equilibrium in the use of fuel only when he has moved out the iso-emission curve to where (10s) is satisfied. Fuel use and control expenditures are thus simultaneously determined by the factor use condition (10s) and the requirement not to exceed allowable emissions (11).

To find effect of changing allowable emissions, take the differentials (7) – (9), (10s) and (11). If adjustments are too small to affect variable input prices, the only exogenous change will be the change $d\bar{s}$ in allowable emissions. The solutions for induced changes in fuel use and control expenditures are

$$df/d\bar{s} = p_c\sigma_c M_{ff}/M_s, \tag{12s}$$

$$dc/d\bar{s} = (M - p_c\sigma_f M_{ff})/M_s, \tag{13s}$$

where $M_s = s_c M + (s_f\sigma_c - s_c\sigma_f)p_c M_{ff}$. The first subscript of a double subscript for $M$ indicates the deletion of a row, and the second indicates deletion of a column.

The benefits from producing $x$ are $b(x) = \int_0^x D(X)dX - p_u u - p_f f - p_c c$, that is, the consumption benefits less the input costs. The change in benefits from imposing an incremental adjustment in $s$ is obtained by differentiating benefits with respect to $s$ to obtain $b(x)_s{}^{\bar{s}} = p_x(dx/ds) - p_u(du/ds) - p_f(df/ds) - p_c(dc/ds)$. This expression can be simplified by inserting the derivative of the production function for commodity output (8) with respect to $s$, $(dx/ds) = x_u(du/ds) + x_f(df/ds)$, into the change in benefits to eliminate $(dx/ds)$, giving $(p_x x_u - p_u)(du/ds) + (p_x x_f - px_f - p_f)(df/ds) - p_c(dc/ds)$. Substituting in the marginal productivity conditions (9) and (10s) further simplifies the change in benefits to $b(x)_s{}^{\bar{s}} = p_c\sigma(df/d\bar{s}) - p_c(dc/d\bar{s})$. The first term on the right side is the excess $p_c\sigma$ of the marginal benefits from fuel use over the marginal resource cost of fuel, times the change in fuel resulting from a one-unit change in allowed emissions $\bar{s}$. The second term is the resource cost of emission controls $c$ resulting from a unit change in $s$. The simplifications leading to $b(x)_s{}^{\bar{s}}$ make use of the idea that in a no control equilibrium the total benefits in the production of $x$ are maximized implying marginal benefits are zero; i.e., extra resources devoted to $x$ are just worth the benefits obtained. A change in benefits when $s$ is changed occurs only if the marginal conditions are not fulfilled. The change in benefits is the difference between the marginal resource costs incurred for those inputs not being used so as to maximize benefits in the production of $x$.

The change in benefits can be simplified further because the two terms in the centered expression for $b(x)_s{}^{\bar{s}}$ just given are control cost effects. The term $-p_c(dc/d\bar{s})$ is the direct change in control costs as a result of a change in allowable emissions and would be the entire change in benefits if there were no induced change in $f$. On the other hand, if there were no change in control costs and the entire adjustment was to change $f$, adjustments in control costs would be avoided. A reduction in fuel use of one unit reduces emissions by $s_f$ making it possible to avoid reducing control inputs by $s_f/s_c$. Since $\sigma = -s_f/s_c$ the saving on control costs is $p_c\sigma$. Differentiation of the emission relation (11) with respect to $\bar{s}$ gives as a necessary condition between fuel and control input changes $1 = s_c(dc/d\bar{s}) + s_f(df/d\bar{s})$ indicating that the sum of the emissions changes due to control input and fuel adjustment must equal the total emissions change. Rearranging, the change in fuel is $df/d\bar{s} = [1 - s_c(dc/d\bar{s})]/s_f$ or the part of the emission change not met through control costs divided by change in emissions per unit of fuel change. Substituting this change in fuel into the expression for $b(x)_s{}^{\bar{s}}$ gives

$$b(x)_s{}^{\bar{s}} = -p_c/s_c. \qquad (14s)$$

The benefit resulting from a change in allowed emissions reduces the control cost saving that would be made possible by allowing a one-unit emissions change, holding fuel constant. Comparing with the previous expression for $b(x)_s{}^{\bar{s}}$ the benefit is not the actual control cost change but rather is what the control cost change would be if the entire adjustment in emissions were achieved via a change in control inputs.

The slope $b(x)_{ss}{}^{\bar{s}}$ of the marginal benefit schedule, needed to evaluate the cost of an emission regulation, can be found by differentiating (14s) with respect to $s$ to obtain

$$b(x)_{ss}^{\bar{s}} = (p_c/s_c^2)[s_{cf}(df/d\bar{s}) + s_{cc}(dc/d\bar{s})] \tag{15s}$$

where $df/d\bar{s}$ and $dc/d\bar{s}$ are given by (12s) and (13s).

### *Pollution control devices*

A second type of policy would not control emissions directly but would require producers to undertake emission control expenditures, making $\bar{c}$ exogenous. There are then no incentives to hold down fuel use. The producer model consists of (7) – (10) plus the condition that $c$ is exogenous, which is the same as the free market model except that $c$ is nonzero. The cost of this policy is simply the emission control expenditure. The effect on benefits (negative of costs) of a one-unit change in emissions achieved through altering control inputs is input price $p_c$ times the $1/s_c$ emission control inputs required to reduce emissions by one unit.

$$b(x)_s^{\bar{c}} = -p_c/s_c. \tag{14c}$$

The right hand sides of (14s) and (14c) are identical because benefit change (14s) under the $\bar{s}$ policy can be expressed as a hypothetical control cost expenditure that would be necessary. In (14c) the change in expenditure is actual.

The *change in marginal benefits* with respect to *emission control inputs* is the derivative of (14c) with respect to $c$, or $-p_c s_{cc}/s_c^2$. The slope being sought is the change in marginal benefits with respect to *emissions* and is this derivative divided by the associated change in emissions $ds/d\bar{c}$. Since there are no incentives to change $f$, $ds/d\bar{c}$ is obtained by differentiating (11) with respect to $c$ holding $f$ constant or $s_c$. Thus the slope of the marginal benefit schedule under the policy of controlling $c$ is

$$b(x)_{ss}^{\bar{c}} = p_c s_{cc}/s_c^3. \tag{15c}$$

### *Restricting waste producing inputs*

The simplest example of a policy operating through waste producing inputs is a direct control on an amount of a fuel. Instead of choosing fuel according to (4) or (4s), fuel $f$ becomes exogenous. The producer model then is (7) – (9) determining price of output $p_x$, output $x$ and nonfuel inputs $u$. Since no incentive is given to make emission control expenditures, $c = 0$.

Differentiate benefits $\int_0^x D(X)dX - p_u u - p_f f - p_c c$ with respect to $f$, substitute in the derivative of the production function with respect to $f$, and make use of (9) and the condition that $c = 0$ to ascertain that the change in benefits with respect to $f$ is $p_x s_f - p_f$, or the difference between marginal revenue and marginal cost from the extra unit of $f$, which is reasonable since the other inputs are either in equilibrium or are zero. Since the amount of fuel needed to reduce emissions by one unit is $1/s_f$, the effect on benefits of a unit change in emissions achieved through reducing fuel inputs is

$$b(x)_s^{\bar{f}} = (p_x x_f - p_f)/s_f. \tag{14f}$$

It was possible to express benefit change under general emission control (14s) in terms of control costs because the difference between marginal revenue and marginal cost of fuel $p_x x_f - p_f$ was equal to addition to control costs required due to adding fuel, i.e., from (10s) $p_x s_f - p_f = p_c \sigma$. The latter equality does not hold under the fuel restriction policy. As $f$ is reduced, the divergence between marginal revenue and marginal cost will grow. The value of $\sigma = -[s_f(f, 0)]/[s_c(f, 0)]$, or the control inputs that would be required to keep emissions from increasing when $f$ is changed, might be altered little if at all. Thus (14f) must remain as stated with no conversion to equivalent control cost.

To obtain the *change in marginal benefits* with respect to *fuel*, differentiate (14f) with respect to $f$ to obtain $[p_x x_{fu}(du/d\bar{f}) + p_x x_{ff} + x_f(dp_x/d\bar{f})]/s_f$. This approximation holds as long as the second term in the differentiation is zero $\{-[(p_x x_f - p_f)/s_f^2]s_{ff} = 0\}$, which is necessarily so at the free market equilibrium where $p_x x_f - p_f = 0$. The approximation remains good as long as the fuel restriction is not so severe as to raise $p_x x_f - p_f$ to a significantly large value. Another defense of the approximation is the likelihood that $s_{ff}$ will be small. The reasonable assumption that, with zero emission controls, emission will tend to be proportional to fuel input, implies $s_{ff}$ is zero, making the term in question drop out. This assumption is used in the functional form examples later.

Take the differentials of (7) – (10) letting $f$ change exogenously, and solve the linear system to obtain $du/d\bar{f} = -M_{fu}/M_{ff}$ and $dp_x/d\bar{f} = M_{fp}/M_{ff}$. Substitute these results into the change in marginal benefits resulting from a change in fuel given at the beginning of the previous paragraph, factor out $1/M_{ff}$ from the bracket, and note that the bracket then equals $M$. The change in marginal benefits from a unit change in *emissions,* achieved via an input policy such as fuel restriction, is obtained by dividing $ds/d\bar{f}$ $(= s_f)$:

$$b(x)_{ss}^{\bar{f}} = (1/s_f^2)(M/M_{ff}). \tag{15f}$$

A policy giving the producer equivalent incentives to adjust the amount of fuel would be a tax on fuel equal to $p_x x_f - p_f$, i.e., a tax making an equivalent divergence between marginal revenue and marginal resource cost of coal. From (14f) it is seen that the marginal benefits are proportional to the amount of this tax. The *change* in marginal benefits is then proportional to the change that would occur in such a tax. $M/M_{ff}$ on the right side of $b(x)_{ss}^{\bar{f}}$ is the reciprocal of the response of fuel use to a change in fuel price and is thus, in fact, equal to the change in tax that would be necessary to bring about a unit change in fuel use, which is then converted to an emissions basis by the $(1/s_f^2)$ term.

### *Restricting output*

A fourth type of policy seeks to control emissions even more indirectly, through affecting the producer's decision as to amount of $x$ produced. The simplest example is a direct restriction making $x$ exogenous. In the model of producer decision, the demand relation (7) is dropped since the regulation of $x$ prevents the producer from adjusting output to demand. The model then consists of the production function (8) and the factor demand relations (9) and (10) in which price of output $p$ is replaced by the marginal cost of output $\lambda$. The producer adjusts factors to minimize the cost of a given output but is unable to carry output to where $p = \lambda$. In the other models, where $x$ is not controlled, marginal cost equals price making it unnecessary to distinguish between $p$ and $\lambda$.

Since the derivative of benefits with respect to $x$ is $p_x - p_u(du/dx) - p_f(df/dx)$, since (3) and (4) permit the substitutions $p_u = \lambda x_u$ and $p_f = \lambda x_f$, and since the derivative of the production function (2) with respect to $x$ gives the substitution $1 = x_u(du/ds) + x_f(df/ds)$, the marginal benefit from a change in $x$ reduces to $p_x - \lambda$ which, reasonably, is the value of an extra unit of $x$ minus the cost of producing it. The *marginal benefit* with respect to *emissions,* achieved through the exogenous changes in $x$, is obtained as in the other cases by dividing by the change in emissions resulting from the change in $x$:

$$b(x)_s^{\bar{x}} = (p_x - \lambda)/s_f(df/dx). \tag{14x}$$

Using the same logic as for the fuel restriction policy, the *change in marginal benefits* with respect to *emissions* when $x$ is changed, $b(x)_{ss}^{\bar{x}}$, is $(1/s_f^2)[(dp_x/dx) - (d\lambda/dx)]$ divided by $(M_{xf}/M_{xx})^2$ which is the square of the fuel change resulting from a change in $x$ obtained from solving (8′) – (10′) with $x$ exogenous. Also from (8′) – (10′), $d\lambda/dx = -M_{xp}/M_{xx}$. From the demand relation (1), $dp_x/dx = D_{xx}$. Making these substitutions in the expression for $b(x)_{ss}^{\bar{x}}$, factoring out $1/M_{xx}$ from the bracket and noting that the bracket then equals $M$, gives as the slope of the marginal benefit schedule for the case where output is controlled

$$b(x)_{ss}^{\bar{x}} = (1/s_f^2)(M_{xx}/M_{xf})(M/M_{xf}), \tag{15x}$$

which can be interpreted as the change in tax on output required to change emissions by a unit $M/M_{xf}s_f$, divided by the change in emissions per unit change in output $s_fM_{xf}/M_{xx}$.

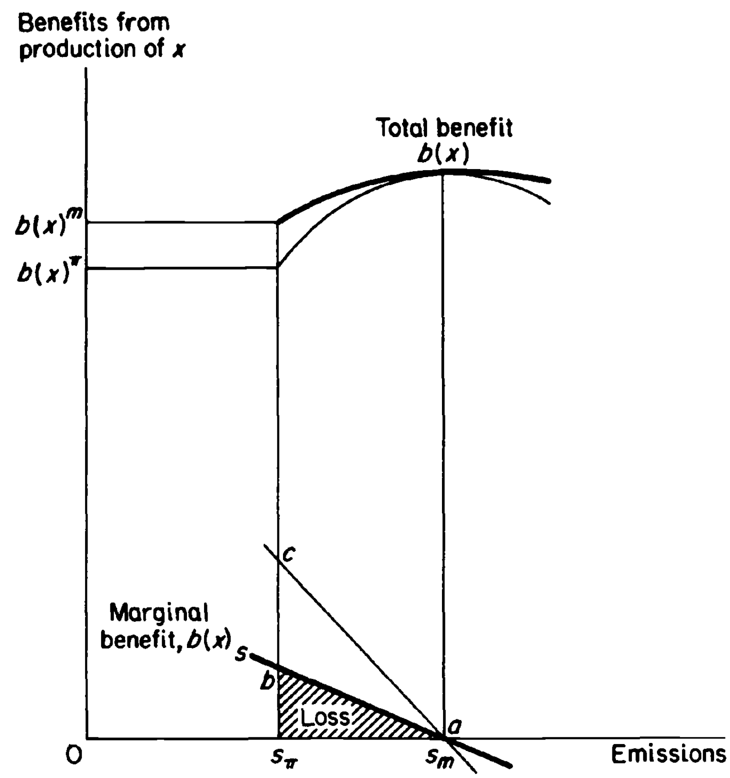
### *Comparison of the four policies*

The curved lines in figure 3 are total benefits from $x$ and are a maximum $b(x)^m$ at the free market level of emissions $s_m$. Benefits from $x$ are reduced as one moves to lower emissions. The dark curved line shows the benefits from $x$ under one particular policy. The dark straight line is the marginal benefit curve from $x$ for this policy. The light lines in Figure 3 pertain to an alternative policy. The cost of a policy is the difference between free market benefits and benefits under the policy, or $b(x)^m - b(x)^\pi$ where $\pi$ denotes the policy. This cost is the shaded area under the marginal benefit schedule in figure 3.

The cost is the sum of marginal benefits in going from free market emissions $s_m$ to emissions $s_\pi$ under the policy, or $[b(x)^m - b(x)^\pi] = -[\int_{s_\pi}^{s_m} b(x)_s^\pi ds]$ $b(x)_s^\pi ds$. Marginal benefit at the free market solution is marginal benefit at any lower level of emissions $s$ plus the sum of changes in marginal benefits going from the lower level up to the free market level, or solving for the marginal benefits at the lower level $b(x)_s^\pi = b(x)_s^m - \int_s^{s_m} b(x)_{ss}^\pi\, dS$. Substituting this result into the expression for cost and assuming free market marginal benefits from pollution are zero, the cost of any policy $\pi$ is:

$$b(x)^m - b(x)^\pi = \int_{s_\pi}^{s_m} \left[\int_s^{s_m} b(x)_{ss}^\pi\, dS\right] ds. \tag{16}$$

*Figure 3*




If the marginal benefit schedule can be approximated as linear, $b(x)_{ss}{}^{\pi}$ is constant giving as the cost of any policy $\pi$:

$$b(x)^m - b(x)^\pi = b(x)_{ss}{}^{\pi}(s_m - s_\pi)^2/2, \tag{16L}$$

verifiable as the shaded area by inspection. For a given emission reduction, the costs of the policies are thus proportional to the slopes $b(x)_{ss}{}^{\pi}$ of the marginal benefits schedules (15s), (15c), (15f) and (15x).

A possible functional form for the emission relation (11) is $s = mfe^{-kp_c c/p_f f}$, where $m$ is emissions per pound of fuel if there are no control inputs and $k$ is the percentage reduction in emissions per pound of fuel resulting from an extra dollar of expenditures on control inputs relative to fuel. Evaluating (15s) with this functional form, inserting the result into (16L) and dividing by $p_f f$ reveals that the estimated cost, relative to fuel expenditures, of a policy of regulating emissions is

$$[1/(-k + \eta_f)](a^2/2), \tag{16s}$$

where $a$ is the reduction in emissions as a per cent of total emissions and $\eta_f$ is the own price elasticity of demand for fuel. Similarly, the cost relative to fuel expenditures of a policy of requiring emission control inputs is

$$(1/-k)(a^2/2). \tag{16c}$$

The cost relative to fuel expenditures of restricting fuel inputs is

$$(1/\eta_f)(a^2/2). \tag{16f}$$

Finally, the cost relative to fuel expenditures of restricting the firm's output is

$$(1/-v\eta_x)(a^2/2), \tag{16x}$$

where $v$ is the change in value of fuel inputs per unit of output accompanying a change in output and $\eta_x$ is the elasticity of demand for fuel with respect to the price of output.

With regard to the last policy, the elasticity $\eta_x$ in (16x) is a firm scale effect. The only reason that fuel use is affected by the price of $x$ is that there is a product output response which changes all inputs. On the other hand, the elasticity $\eta_f$ in (16f) contains both a scale effect and a substitution effect. In addition to giving incentive to change the scale of output, a change in the price of fuel gives incentives to substitute between fuel and other inputs, indicating that the cost of a fuel restriction policy relative to fuel expenditures is less than that of restricting the firm's output.

Comparing (16f) and (16c) indicates that whether a fuel restriction policy is cheaper than requiring emission control inputs depends on whether $\eta_f$ is less than $k$. Since the formulas express costs as a per cent of fuel expenditures, the cost comparison also depends on the absolute level of fuel expenditures. The least costly of the four policies relative to fuel expenditures is emission regulation, which (16s) reveals to be a combination of the fuel restriction and control input policies. If the latter two policies happen to be equally costly, the emissions regulation will be half the cost of either of them.

As a further application, if the firm faces a perfectly elastic product demand and has a CES production function for output, evaluation of $M$ and its cofactors gives $\eta_f = [1 - \epsilon + (\epsilon - \gamma)/(1 + p_f f/p_u u)]/(1 - \epsilon)(\gamma - 1)$ and $\eta_x = 1/(1 - \gamma)$ where the elasticity of substitution is $1/(1 - \epsilon)$ and the scale parameter $\gamma$ is the percentage change in output that would result from a simultaneous 1 per cent increase in inputs $u$ and $f$. For a short run situation, suppose the elasticity of substitution is zero ($\epsilon = \infty$). Suppose that expenditures on fuel and other variable inputs are each a third of the value of output, the total of the shares being substantially less than one due to short run fixity of many inputs. Assuming the shares add to the elasticity of output with respect to the inputs implying $\gamma = 2/3$, the costs of a fuel restriction and an output restriction policy are identical because of the zero elasticity of substitution assumption and are $a^2/3$ of fuel expenditures. For a long run situation, suppose that the elasticity of substitution is one ($\epsilon = 0$), fuel is a third the value of output, and other inputs are one-half the value of output with $\gamma = 5/6$. As a per cent of fuel expenditures, the costs of a fuel restriction policy are then $a^2/6$ and the costs of output restriction policy are $5a^2/24$. If $k$ is 5, costs relative to fuel expenditures of a policy of requiring emission control inputs are $a^2/10$. The costs of emission regulation relative to fuel costs are $a^2/13$ in the short run and 1/8 in the long run. These examples illustrating how factor substitution and scale effects determine policy costs are consistent with the idea that costs rise with increasing rapidity as emission reduction approaches 100 per cent, in view of the $a^2$ term.

### *Relevance*

This section has dealt with production theory for a firm under restrictions, in contrast to previous studies in which information about specific control devices and fuels has been used to estimate dollar costs at a point assuming no substitutions, for example the two studies done by the U.S. Environmental Protection Agency in 1963 and 1970. In future work it would be most useful to draw on details in engineering and physical science studies to estimate emission and output production functions, thus obtaining refined measures of substitution and scale effects.

Each policy type has many examples, all in need of the analysis contained in this section. The proposed tax on sulfur dioxide emissions is an example of the least costly of the four policy types. The major approach to air and water pollution followed in practice is of the same general type in that it deals with emissions. In the emission relation (11), there is a positive relation between polluting inputs and emissions. The