Chapter 10

COMPARATIVE ANALYSIS OF ALTERNATIVE POLICY INSTRUMENTS

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1. Introduction

The choice of instruments for environmental policy implementation has had a special place in applied economics since the 1920s, when Pigou suggested the use of taxes on negative external effects and subsidies on positive external effects to correct allocative distortions. This is understandable, for at least at first glance it is a problem that appears to offer a nearly perfect target for our skills. Because of the importance here of external effects and public goods, and because the policies and thus the associated implementation strategies have had to be devised *de novo*, it has seemed an area to which economic insights, independent of other disciplines and unfettered by tedious historical baggage, could make very great contributions.

To some extent, of course, this has been true. Sophisticated theoretical work has contributed to the understanding of the characteristics of particular instruments. Empirical studies have produced estimates of the actual cost advantages to be expected from the adoption of instruments favored by economists instead of those being put in place by policy-makers. But therein lies the rub; those policy-makers have for the most part stubbornly refused to accept and act on the basis of the theory offered and the supporting empirical work. Overall, economists seem to have been perceived as gadflies, ignoring or misunderstanding the real situation and thus producing largely irrelevant criticisms of the instruments actually chosen, along with impractical, even politically dangerous, prescriptions for change. (Although see the comments on European experience below.)

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As with all such standoffs between the research and policy worlds, some truth resides with both sides. As this review will seek to show, economists have achieved some of their fundamental results by ignoring crucial features of the physical world and by abstracting from the full complexity of the economic world. These concessions to simplicty have made most of the arguments that are easiest to explain, and hence potentially easiest to sell, if not wrong at least seriously misleading. Further, and whether rightly or wrongly, economists seem to have refused to take seriously the political implications of some of their favorite prescriptions. These implications include both straightforward matters of cost distribution and more subtle problems of ethical content. At the same time, the developers and supporters of the regulatory systems currently in place in the United States and many other industrial countries have themselves been guilty of misleading arguments, and some of these will be pointed out in what follows. The core of good sense in economic criticisms of command and control regulation and in economic prescriptions for more flexible incentive systems should not permanently be obscured by the rhetorical flourishes of those who favor systems with strong and explicit moral overtones or who have narrower interests in the evolving status quo.

The structure of this chapter is designed to accomplish four specific goals as part of our broader aim of clarifying the contribution of economic analysis to the debate over the instruments of environmental policy. First, we shall describe the general situation in which environmental policy goals must be achieved. An appreciation of the complexity of this situation will provide a base from which to consider both past error and actual special cases. Second, we shall define a set of dimensions along which policy instruments may usefully be judged. These include: static efficiency, centralized information and computation requirements, enforceability, dynamic incentive effects, flexibility in the face of exogenous change, and implications for goals other than efficiency. In the process, we intend to make explicit the irreducible political content of choices among policy instruments and thus the reasons that technical arguments on the other dimensions will not be decisive in the political arena. Third, we shall briefly review both some major non-economic attempts to evade the complexity of the general case and the record of adoptions of explicitly economic prescriptions.

Finally, following this background tour, we shall return to examine more carefully some of the economic complexities associated with a variety of instruments and problems.

Section 2 will concentrate on administratively (or legislatively) set prices and taxes designed to influence behavior.

Section 3 will be devoted to instruments that complete the set of markets – that is, where commodities or rights are administratively defined and prices are set by decentralized bargains among the actors subject to the policy (owning or wishing to own the rights).

Section 4 will deal with various forms of deposit–refund systems and performance bonds, as well as liability rules.

Section 5 will take up specifications of behavior and other instruments involving direct intervention in the behavior of the actors subject to the policy.

Section 6, finally, will discuss moral suasion as a policy instrument, primarily in contexts where there are significant constraints on the set of policy alternatives.

1.1. Some definitions and background assumptions

The position adopted in this chapter is that choice of policy goal and choice of instrument or implementation system are essentially separable problems. And, for the most part, the discussion here will take as given goals or standards for ambient environmental quality (air, surface or groundwater, landscape or whatever). The conceptually preferable position, that both goal and instrument must be chosen simultaneously in a grand meta-benefit/cost analysis, is for now operationally quite hopeless. Further, we shall usually assume the existence of an agency of government charged with meeting the standards.

The essence of the agency's problem of attaining chosen ambient quality standards is that the actions of many individual and independent actors (firms, households, other government units) affect actual environmental quality. The actors will differ among themselves in production technologies and product mixes (where these words are interpreted broadly enough to include such activities as home space heating and sewage treatment plant operation). In the most general case, the environmental effect of each actor is different from that of every other actor and more than one combination of actions by all the actors will result in meeting the standard.[1] The combinations will differ both in total cost to society and in the way any particular cost is distributed across the set of actors.

These actors are all assumed to be "rational" and self-interested.[2] For the agency to succeed in attaining the ambient environmental standard it must in

[1] The most common environmental policy problems involve as "actions" discharges of pollution into part of the natural environment. The effects of each source's actions depend on the characteristics of the environment (stream flow, water temperature, and so forth for water pollutants; wind speed and direction, terrain, hours of sunlight and so forth, for air pollutants). More generally, "actions" can include such diverse matters as the construction of ugly buildings, the use of farming methods that disrupt natural terrestial ecosystems, or the placing of radioactive wastes into trenches or caverns. We usually will take "effects" to be measured relative to the ambient environmental quality standards at specified monitoring points. A more elegant treatment would involve measuring effects as damages (to human health and welfare, the ecological support systems, and so forth, but such an approach to policy implementation is not yet practically significant.

[2] Some criticisms of policy instruments that allow the actors flexibility, such as charges per unit of emissions of pollutant, appear to arise from the opposite assumption – that dischargers of pollution will act in an economically irrational way and pay a higher charge bill than would be optimal just for the pleasure of polluting. It seems possible that these critics have confused the position of a wealthy person confronted by a consumption tax with that of a firm.

some way induce at least some of the actors to take actions contrary to their narrow self-interests as defined in the pre-policy world of relative prices and constraints. The costs of these actions may involve both real resource use and transfers that are costs only to the payors, not to the larger society. The assumptions imply that when an actor, more particularly a firm, is faced with orders or charges ordained by the agency, it will respond in a way to maximize its present value in the long run. In the short run, we can usually capture all that is important by assuming the minimization of costs for given output, location, technology and factor prices, but subject to the new constraint or taking account of the new price, as on pollution discharge.[3] The difference between short and long run is the range of adjustments available. In the long run, the discharger can seek a new location, production process and pollution control technology, even entirely new products to make.

A full description of the background setting for the discussion of environmental policy instruments must include the fact that the agency cannot costlessly know

[3] Again it will be useful to tie this notion down by reference to the most common problem, pollution discharge. The cost to an actor of adjusting to orders or prices imposed by the agency will be captured in a cost-of-discharge-reduction function. This function shows the marginal (or total) cost of reducing discharge by an amount, R, below its level in the absence of any agency initiative. For many short run purposes it will be convenient to assume that this function is defined for fixed output, though more generally, output and all factor input decisions are made simultaneously with the decision about R. Thus, more generally, the two problems, one for an emission charge and one for an emission limit, may be written as follows:

Charge

$$\max_{\text{s.t.}} p'Z - q'Y - e(X-R)$$
$$0 = F(Z, Y, X-R),$$
$$Z, Y, X-R, R \geq 0.$$

Limit

$$\max_{\text{s.t.}} p'Z - q'Y$$
$$0 = F(Z, Y, X-R),$$
$$X - R \leq L,$$
$$Z, Y, X-R, R, \geq 0,$$

where Z is a vector of outputs, with prices p'; Y is a vector of inputs other than pollution discharge services, with prices q'; X, uncontrolled discharges; R, discharge reduction; e the emission charge; and L the emission limit.

In the short run, X is implicitly defined by the problem:

$$\max_{\text{s.t.}} p'Z - q'Y$$
$$0 = F(Z, Y, X),$$
$$Z, Y, X \geq 0.$$

This notion of "discharge reduction" can be expanded just as we expanded the notion of "discharge".

what each of the actors is actually doing at any particular time or on average over any period. Checking the behavior of the actors against applicable regulatory orders, or determining what is owed by way of emission charges is another resource-using problem, one we shall refer to as the monitoring problem. The subsequent matter of punishing violators of orders, or those in some way abusing the charge system, we call the enforcement problem. The monitoring problem is made especially difficult, again in the most general case, by the character of "pollution discharges". These are for the most part invisible to the unaided human senses. Furthermore, once a unit of discharge has left the discharge point it is in general not attributable to any particular discharger. Thus, measurement must occur at the point of exit (or before) if it is to occur at all.[4]

Finally, the entire problem of environmental policy implementation is embedded in changing natural (atmospheric land surface, and aquatic) and economic worlds. These changes occur on the short run, stochastic scale of wind and weather shifts as well as the secular scale of changing tastes and technology. As the world changes, with ambient quality standard held constant, the implementation problem changes. If a particular set of actions by dischargers results in meeting the ambient quality constraint under conditions A, those actions may fail to produce an acceptable result under conditions B, perhaps because sources have moved or production levels have changed in response to changing tastes or resource prices, or simply because the natural systems involved do not dilute and disperse the discharges in the same way under B as under A.

1.2. Dimensions for judging environmental policy instruments

The above description of the general situation in which environmental policies must be achieved suggests several dimensions along which potential instruments for achievement may be judged.

(1) Static efficiency. The efficient implementation system achieves the chosen goal at least resource cost. This dimension is almost always interpreted in a static sense and that will be the approach here. "Static" means, as a practical matter, that we assume an unchanging environmental goal and allow only for the first round of reaction to the implementation orders or incentives; that is, discharge reductions with fixed technology and location for each discharger.

(2) Information intensity. This dimension involves the attempt to measure, at least qualitatively, how much data and what level of predictive modeling skills must be available to the pollution control agency to use the implementation system in question. Its importance lies in the desire for efficiency coupled with our assumption of many different actors affecting the environment differently. As a

[4] This, we repeat, is the general case and may be qualified in a number of ways. See below, footnote 47.

general matter, efficiency will require that each actor be given an individually tailored order (such as a discharge reduction order) or be faced with an individually tailored price for discharges or subsidy for discharge reductions. Finding the full set of such tailor-made instruments requires either an information- and computation-intensive "model" of the situation to be regulated or a very difficult trial and error process.

(3) Ease of monitoring and enforcement. This refers to the relative difficulty of making and interpreting the measurements of discharges necessary to judge compliance, prepare bills, or audit self-reporting. These measurements are complicated not only by the features of invisibility and inherent "fugitiveness" already mentioned, but by the variability of discharges with production levels, equipment malfunctions and operator actions; by the imprecision of measurement devices and the discrete sampling techniques used in many such devices; and by the awkwardness involved in obtaining entry to a discharger's premises and setting up elaborate equipment in order to take the samples. The overall effect is to make it very expensive for the agency to use common measurement methods frequently enough to produce any reasonable probability of detecting a true violation of a time-averaged discharge standard (or to check the payment for an emission charge over a similar period).

Enforcement actions to prod violators back into line may include administrative fines, civil or criminal court proceedings and penalties, or more indirect actions, such as blacklisting. The relation between the enforcement penalties and methods and the monitoring activities (and hence the probability of detection) is important in defining the incentives for compliance with the chosen instruments, but the choice of enforcement methods may reasonably be seen as a second-order version of the choice of the instruments themselves and is therefore treated only cursorily in what follows.

(4) Flexibility in the face of economic change. Here the interest is in the ease with which the implementation system adjusts to maintain the given ambient goal when exogenous changes occur in tastes, technologies, resource use, or other features of economic activity. The fundamental distinction is between a system that adjusts through decentralized actions of the regulated dischargers – firms, households, and other government units – and one that must be adjusted through recalculation and imposition by the agency of the new discharge standards or required emission charges. The advantages of flexibility in this sense include the avoidance not only of information gathering and computations, but also of the inevitable political interference with changes in the system.[5]

(5) Dynamic incentives. This involves the actions encouraged by the instrument in the longer run. One important distinction here is between instruments that

[5] This judgement assumes that the ambient quality standard is a legitimately chosen policy goal. Tinkering with the implementation system, while aimed at changing only cost shares, may affect the society's ability to achieve the goal itself.

encourage the search for and adoption of new, environment-saving technology and those that encourage retention and operation of existing plants. Another is between instruments that distort relative factor prices, as by making capital-intensive methods artificially cheap, and those that do not. A third distinction of some interest is between instruments that provide incentives for dischargers to move and those that do not.[6]

(6) Political considerations. Several political considerations affect society's choice of policy instrument at least as much as cogent arguments about their relative merits on the above dimensions. Three are especially important. The first is distributional, the second ethical, and the third relates to broader economic stabilization concerns.

At the simplest level, it is clear that the matter of cost distribution is intimately linked to the political viability of alternative ways of meeting collective environmental goals. Because we choose a distribution of benefits when we choose the goals, and because we have no mechanisms (other than the very creakiest mechanisms) for redistributing incomes (and thus benefits), a choice of cost distribution implies a fixed pattern, of net benefits for that broad area of environmental policy.[7] If an analysis of the distribution of costs and benefits shows that a majority of voters or the members of some powerful or vocal voting block will probably incur net costs from the policy, one would certainly be tempted to predict at least a rocky road for it.[8] Note further that, from the point of view of the payor, "mere transfers", such as emission charge payments, are part of the cost of an instrument.

The ethical features of environmental policy instruments include, most prominently, the message conveyed and the extent to which the actors in the system are allowed to choose among alternative actions. One widely held view is that environmental policy should involve stigmatizing pollution, whether as a crime against nature or against other persons. [See, for example, the arguments in Kelman (1981) and those of Railton (1984).] In this view, regulatory orders backed up by criminal sanctions have the proper flavor, while charge systems that make "buying pollution" just like buying labor services, are immoral. A related matter is that of choice. While freeing up discharger choices is usually at the heart

[6] This distinction may be illustrated by the difference between an efficient emission charge system, which must be based on individual charges, tailored to location, and a uniform charge system. Under the former there necessarily are possibilities for movement to lower charge locations. (Though, as a practical matter, the anticipated savings might only rarely be large enough to justify the cost of moving.)

[7] As already discussed, in the longer run, by changing residence, job, asset portfolio, or habits, individuals can change their own net benefits from a particular policy (goal plus method of accomplishment).

[8] Because neither costs nor benefits of environmental quality improvements are easy for individuals to determine, most may find it hard to judge where their self interest stands once any option is operating. Thus, predictions may create opposition that would not otherwise exist. Indeed, the idea that opposition *ought* to exist may be itself enough to do in a plan.

of economists' arguments for the efficiency properties of economic incentive instruments, the very provision of such choice appears ethically undesirable to others. If pollution discharge is wrong *per se* (as opposed to being wrong only when done in excess of a discharge standard or as part of a fraud in the context of an emission charge system) then there should at least be no choice about how much of the wrong each actor is allowed to produce. And, indeed, there should be no confusion about "allowable" being equivalent to "acceptable". For those who hold discharge to be wrong, there is no acceptable discharge goal this side of zero, and the only acceptable dynamic incentive is one aimed at that goal. (On ethical questions in environmental policy more broadly, see Chapter 5 of this Handbook.)

Aspects of stabilization policy may also play an important role. At least, this was the case in certain European countries during the 1970s, where municipal waste water treatment installations were improved partly for reasons of environmental policy, partly to counteract recession in the building industry. [See, for example; OECD (1978).]

1.3. Avoiding the complications: Shortcuts from goals to behavior

The practical difficulties of the general case, which imply that advancing along one of the above dimensions usually means giving up something on another (or on several others), may be seen as the inspirations for attempts to construct shortcuts for society to follow. Some of the features of these shortcuts will reappear in later discussions, but for now they may be viewed simply as special cases, in which goal and instrument collapse into a single entity.

One such case involves pure technology standards. The actors in the situation are required to adopt particular treatment (or production) technologies. Whatever discharges result from the adoption are accepted, and the ambient goal implicitly becomes whatever is achieved when all sources comply. This approach has the advantage of appearing easy to monitor (though operation is different from installation and the "easy" monitoring only applies to installation).

The technology standard may be extended to the long run and in the process appear to capture some of the ethical high ground while at the same time seeming to provide desirable incentives. This shortcut amounts to the injunction to "do your best" at all times – in particular to adopt better technology as it becomes available. This seems to force each discharger inexorably toward zero discharge. But, of course, since much technical change is endogenous to the system of incentives, and since this policy implies fresh costs for new technologies with no rewards, "do your best" seems very likely to have the effect of slowing progress toward lower discharges.

A third shortcut is to use an emission charge as a revenue-raising device for a program of environmental quality improvement based on government projects or subsidies. Here, the charge is related to some characteristic of the discharger that will be relatively insensitive to it – for example, output – with the actual unit charge usually based on a rule of thumb relation between output and discharge. In this case, the expectation is that output, and hence charge payments will remain unchanged. In such plans the revenue collected is usually intended to be used for projects such as regional sewers and treatment plants, or treatment plant subsidies to individual dischargers, designed to improve environmental quality. The facilities become, *de facto*, the policy goal, and the environmental quality they produce is accepted willy nilly. Another possibility is to treat the revenue as part of the state's general revenue. For a discussion of the reduction in excess burden from a tax system achievable under this second alternative, see Terkla (1984).

1.4. Historical perspective: Notes on chosen approaches

The residuals from human production and consumption activities have always found their way to the natural environment. And even in long-vanished ages of sparse populations and small scale production units the disposal of these residuals could create local pollution problems in the sense of significant negative externalities. There is no lack of anecdotal evidence of the seriousness of these problems, especially in large cities [Baumol and Oates (1979)]. What does appear to be lacking is evidence that prices (charges) or markets were invited to play any role in dealing with these problems. Regulatory orders backed by fines, imprisonment, or physical punishment, seem to have predominated as policy instruments, though certainly those orders could be quite sophisticated.[9]

What changed over time was the source and geographic scope of pollution problems; not the method of trying to correct them. In the nineteenth century, when rapid industrialization was producing very large air and water pollution problems all over Europe, and in the northeastern United States, it seems that slightly more modern versions of the ancient prohibitions were the medicine first prescribed [e.g. the historical sections in Johnson and Brown (1976) dealing with France, Germany, Hungary, Great Britain and Sweden]. When these manifestly failed, an effort was made to require licenses (permissions, consents, contracts) by the terms of which some limits could be placed on private and municipal dischargers [Richardson et al. (1983.)].

The first significant move away from simple prescription of particular activities in pollution control policy seems to have come very early in this century in Germany, when the first water management cooperatives or Genossenschaften

[9] For example, Parker (1976) reports that the record of the manorial court for the Chatteris Manor, including the village of Foxton (in England) contains a number of rules constraining pollution of the brook that ran through the village. Householders were prohibited from allowing ducks or geese to "frequent" the brook, from washing linen "clothes" in the brook, and from draining household wastes into the brook except after 8 at night. All rules were backed by specified fines per offense.

were authorized for river basins in the North Rhine-Westphalia state [Johnson and Brown (1976), Kneese (1964)]. Instead of attempting to forbid the inevitable waste disposal, these organizations set out to deal with it collectively, through sewer and treatment-plant construction, assessing the costs of these efforts to their members. Of even more interest to latter day economists, the method of cost assessment was (and still is) based on the waste load each member generated. Because of the units (money per unit waste) this charge-back method looks very like an effluent charge and has been described as such by many commentators. But, as we shall see below, the common arguments for the social desirability of an effluent or emission charge are based on quite different goals and system design. Therefore, however much we may admire the audacity of the Germans who broke with at least 1000 years of traditional approaches, we really must wait even longer to see a charging scheme designed with incentive rather than revenue-raising effects in mind.[10]

Implementation programs closely related to the pioneering work of the German Genossenschaften exist now in several European countries, including the Netherlands, France and Hungary [Johnson and Brown (1976)]. Sewer services charges, which are a narrower version of the same approach are widely used in the United States and the United Kingdom [Elliott (1973), Urban Systems (1979), Webb and Woodfield (1981)]. But it appears that only in the Federal Republic of Germany has a national system of charges, designed explicitly to have an incentive effect, been put in place. This system was created by the national law passed in 1976 which will take full effect in 1986 [Bower et al. (1981), Brown (1982)]. This charge is linked to permit terms and compliance therewith, but is *not* based on the costs of collective treatment works.

These European countries are exceptions, however. The United States, for example, has not adopted an emission charge system for dealing with any pollution problem (a sketch of the approaches actually adopted is found in the previous chapter). While any number of proposals for charge applications have been made, both at federal and state levels, none has survived to the stage of implementation. Examples of these failed initiatives include [Baumol and Oates (1979) and Zeckhauser (1981)]:

(1) A 1970 proposal for a national tax on lead in gasoline.

(2) A 1970 citizen's initiative in Maine that put a BOD effluent charge on the ballot as a referendum item [Freeman (1970a), (1970b)].

(3) A 1971 proposal for a national effluent charge based on biochemical oxygen demand (BOD).

[10] Arguments along this line are made by Johnson and Brown (1976) and by Bower et al. (1981). The collective decision-making process of the Genossenschaften is of some considerable interest in its own right, with the dischargers themselves dominating the boards that decide on quality levels, treatment efforts, and hence necessary charges [Klevorick and Kramer (1973)].

(4) A 1972 SO_2 Emissions Tax Proposal. [This proposal was resurrected by Senator Durenberger of Minnesota, as another alternative for dealing with the problem of acid rain. *Inside EPA* (1983).]

(5) The 1972 Vermont law establishing effluent charges for organic discharges to natural waters. (This law was never put into effect, though neither was it, to our knowledge, repealed.)

Rather, the U.S. system of pollution control developed since the Second World War, and very largely since the late 1960s, contains modern versions of the consent or permit approach.[11] At the present time, however, administrative initiatives are creating many of the features of a *marketable* permit system out of the raw material of the original legislation. These new features will be mentioned below when we discuss marketable permits generally.

It is perhaps too extreme to say that the new German national effluent charge law is the only economic incentive system for pollution control ever successfully legislated. A major exception is the so-called "bottle bill" or deposit–refund system aimed at litter pollution by drink containers. Such laws (and similar ones concerning waste lubrication oil, junked cars, etc.) have been successfully put in place in several states of the United States and many European countries and do constitute explicit attempts to influence polluting behavior through economic incentives [Bohm (1981)]. The fact remains, however, that over the long sweep of history direct regulations (prohibitions, specifications of behavior, nonmarketable permits to discharge) have been the instruments of actual choice for dealing with pollution, whether from geese in village brooks or petroleum refineries on major rivers. Unlike commodity prices and markets, which existed before economists began analyzing them, administratively set prices or legistatively created markets do not appear to have sprung up as intuitive responses to externality problems. Quite the reverse; even after sustained intellectual development of these concepts during the period from 1960, we can find few examples of their application.

Let us turn now to more careful consideration of what that development has been about and to the ongoing debate over whether these newer instruments are or are not to be preferred to one or another version of the traditional approaches.

2. Instruments in the form of prices

The use of prices as instruments of environmental policy began to receive serious, and for the most part favorable, attention from economists in the mid 1960s. The most important early work is generally acknowledged to be that of Kneese

[11] More will be said about this system below, but for a reasonably full description, see for water pollution control, Freeman (1978) and for air pollution control, Lave and Omenn (1981).

[especially Kneese (1964)].[12] The theme of this section will be to show how the extremely attractive and compelling case made by Kneese has had to be modified as inconvenient elements of reality were explicitly recognized and dealt with. Because the literature on charges is enormous, matching the broad range of specific questions that has captured the interests of economists, we shall be forced to choose only a few of many lines of argument we might trace. Our choices are based on judgements about practical importance, not necessarily on number of pages devoted to the issue in the literature. We do, however, in the footnotes refer the reader to other disputes.

After we have discussed effluent or emission charges in principle, we shall turn to the design of models for the calculation of optimal charges in realistically complex situations. We shall then be able to report some of the results obtained from those models when they are directed to questions of the relative costs of alternative implementation systems.

To this point, the section will have been couched in static terms, and even our complications of the Kneese model will have assumed away a number of further important considerations relevant to instrument choice. The remainder of the discussion will be devoted to these other matters and will parallel the list of dimensions of judgement offered in the introduction. That is, we consider enforceability, flexibility in the face of exogenous change, dynamic incentive effects, and political implications of alternative instruments.

2.1. *Arguments in the static case*

For expository convenience in this and certain other sections let us construct a very simple model of an environmental policy problem involving two dischargers of a single residual, a natural environment receiving their discharges, receptors (unspecified in number) suffering damage from the resulting environmental degradation, and two potential monitoring stations at which that degradation can be measured.[13] We shall call the dischargers 1 and 2, and the potential monitoring

[12] As a matter of intellectual history, it would be interesting to trace the development of the emission-charge idea from Pigou's statements to Kneese's influential book, with its very practical air. This is not attempted here, but we do note in passing that an even earlier RFF book contained a paper by Gulick (1958) in which the use of prices to "determine interrelationships, priorities, and comparative needs and desires" was advocated in the context of resource problems, including pollution, in the modern city.

[13] A first judgement is implicit in our choice of model structure. It is that a partial equilibrium model can be a useful tool in examining questions of instrument performance. It is not a judgement that will receive universal assent, for general equilibrium treatment allows consideration of the consumption effects of output reductions due to a tax and can thus provide important perspectives about the appropriate instrument for controlling a monopolist and about the symmetry or asymmetry of charges and subsidies. Thus, Mäler's (1974a) comprehensive and insightful treatment of issues related to instrument choice is couched in terms of a general equilibrium model. So is Fisher's (1981)

stations A and B. The following quantities and functions will be central to our purpose:

$X_1, X_2 \equiv$ raw waste loads generated per unit time at the two sources,

$R_1, R_2 \equiv$ reductions in the raw waste loads achieved at the sources, as for example by recycling,

$D_1, D_2 \equiv$ discharges from the two sources per unit time,

so that $D_i \equiv X_i - R_i$, (1)

$C_1(R_1), C_2(R_2) \equiv$ costs of pollution reduction at the two sources $\left(\text{assume } \mathrm{d}C_i/\mathrm{d}R_i > 0;\ \mathrm{d}^2C_i/\mathrm{d}R_1^2 \geq 0\right)$,

$f(D_1, D_2) \equiv$ damages suffered by receptors of the pollution $\left(\text{assume } \partial f/\partial D_i > 0 \text{ and } \partial^2 f/\partial D_i^2 \geq 0\right)$.

Sometimes we shall wish to consider ambient quality standards rather than assuming a damage function is known. For this purpose we define:

$S_A, S_B \equiv$ ambient environmental standards at the monitoring points.

The pollution control agency's problem for our simple region may be written in general terms as:

$$\min f(D_1, D_2) + C_1(R_1) + C_2(R_2) \tag{2}$$

or, by (1):

$$\min f(D_1, D_2) + C_1(X_1 - D_1) + C_2(X_2 - D_2). \tag{2a}$$

With this apparatus in hand it is possible easily to explore the "Kneese case" for charges and several of the most important qualifications to it. The classic case for emission charges depends on two assumptions: that f is linear and that the locations of the sources does not matter to their relative roles in damage production. Then the problem in (2a) becomes

$$\min a(D_1 + D_2) + C_1(X_1 - D_1) + C_2(X_2 - D_2). \tag{3}$$

The first-order conditions for an optimum are:

$$a - \mathrm{d}C_1/\mathrm{d}R_1 = 0 \quad \text{and} \quad a - \mathrm{d}C_2/\mathrm{d}R_2 = 0.$$

Thus, if the authority knows the linear damage function it can announce the optimal charge, a, without knowledge of the sources' cost functions. It is easy to see that if each firm minimizes cost, its response to this charge a will be the "proper" one, and $\mathrm{d}C_i/\mathrm{d}R_i = a$ will be true for $i = 1,2$. The emission charge

more recent and much simpler discussion. Examples of papers addressing specific issues in a general equilibrium framework include: Burrows (1981) on controlling the monopolistic polluter; Sims (1981) on the asymmetry of subsidies and charges in the short run; Meselman (1982) also on subsidies and charges; and Harford and Ogura (1983) on charges and standards.

approach therefore boasts a powerful combination of static efficiency and information economy.

The first part of this case that we shall examine is the assumption that $D_1 + D_2$ is the appropriate argument for the damage function. Consider, for example, the possibility that damages are measured at a particular point (say a riverside park) and that one source is farther upstream from the park than the other. If the residual involved is not entirely conservative, the appropriate (still linear) damage function form should be $a(\alpha_1 D_1 + \alpha_2 D_2)$ with $0 \leq \alpha_1, \alpha_2 < 1$ and $\alpha_1 \neq \alpha_2$.[14]

Then the first order conditions are:

$$a\alpha_1 - \mathrm{d}C_1/\mathrm{d}R_1 = 0 \quad \text{and} \quad a\alpha_2 - \mathrm{d}C_2/\mathrm{d}R_2 = 0. \tag{4}$$

They tell us that the optimal charges must be tailored to the location of each source (location matters) but that the authority can still announce optimal charges without knowledge of the cost functions *if* it knows both the damage function and the action of the environment on the discharges (captured in the α_i, often referred to as "transfer coefficients"). This result holds even if damages are measured at more than one point and added to get total regional damages, and if the sources affect the damage function arguments differently at each such location. Thus, if total damages are given by

$$a_{\mathrm{A}}(\alpha_{1\mathrm{A}} D_1 + \alpha_{2\mathrm{A}} D_2) + a_{\mathrm{B}}(\alpha_{1\mathrm{B}} D_1 + \alpha_{2\mathrm{B}} D_2) + \cdots a_N(\alpha_{1N} D_1 + \alpha_{2N} D_2),$$

then the optimal charge for source 1 is

$$a_{\mathrm{A}}\alpha_{1\mathrm{A}} + a_{\mathrm{B}}\alpha_{1\mathrm{B}} + \cdots + a_N \alpha_{1N}$$

The generalization to M sources is also straightforward.

The classical case for charges begins to unravel as soon as we drop the assumption of linear damage functions. Then the optimal charge is not independent of the optimal discharge levels and, in general, cost functions must be known to the agency. In the simplest such case, the damage function is non-linear in $D_1 + D_2$, and the sources have identical cost functions so that if $D_1 = D_2$, then $C_1(R_1) = C_2(R_2)$. Then it can be shown that at the optimum $D_1 = D_2$, and $\mathrm{d}C_1/\mathrm{d}R_1 = \mathrm{d}C_2/\mathrm{d}R_2$, and a single emission charge is optimal. But the optimal

[14] Notice that the form $\alpha_1 D_1 + \alpha_2 D_2$ arises whenever *either* the residual in question is nonconservative in the environment (e.g, is chemically changed or physically settles out between source and receptors) *or* where we are not in a position to measure the total contribution of a source to ambient quality by looking at a finite number of monitoring points. This latter condition differentiates the general air pollution problem from the general water pollution problem, because diffusion in the atmosphere results in the "loss" of discharged residuals.

Notice also that we are assuming a particularly simple form of the environmental model implicitly embedded in our problem. In general the effect of D_1 on the ambient quality at the damage measurement point may depend both on the level of D_1 and on the levels of all the other discharges. In this general case, things are even more difficult than we shall see them to be below for linear transfer functions.

D_1 and D_2, and hence the optimal charges depend on the cost function parameters. The agency's information requirement is immediately vastly greater.[15]

If either the cost functions are not identical or the locations of the sources matter, then the optimal charges must be source specific and depend on knowledge of the cost functions.

Any number of commentators in the early charges literature pointed out that not only was the assumption of *linear* damage functions unrealistic, but the very idea of any known and accepted damage function was more than economic knowledge could (perhaps ever) support. The point was certainly valid when made, and the reader is free to reach a conclusion on its current validity on the basis of the relevant chapters in this Handbook; our interest is in the line of analysis inspired by it (see especially Chapters 7 and 16). This is the line that takes ambient quality standards, chosen by some exogenous (probably political) process as representing the goals of environmental policy and sees charges as instruments for realizing those goals.

In this context, the agency's problem becomes:

$$\min_{\text{s.t.}} [C_1(X_1 - D_1) + C_2(X_2 - D_2)] \tag{5}$$

$$g(D_1, D_2) \leq S_A$$

for two dischargers and a standard defined at a single point. For M dischargers and N standards, the problem becomes:

$$\begin{aligned} &\min_{\text{s.t.}} \sum_i C_i(X_i - D_i) \\ &g_A(D_1, D_2, \dots, D_M) \leq S_A \\ &\vdots \\ &g_N(D_1, D_2, \dots, D_M) \leq S_N. \end{aligned} \tag{6}$$

In the simplest case, location is assumed not to matter, and only one standard is specified. Then on the basis of our other assumptions we can assume that the standard will be exactly satisfied, and a simple Lagrangian formulation suffices. The agency's problem is:

$$\min L = C_1(X_1 - D_1) + C_2(X_2 - D_2) - \lambda(D_1 + D_2 - S_A). \tag{7}$$

[15] The fact that, in the simpler case, a single emission charge applies might suggest that a trial and error process for seeking the optimum would work. The problem is that only by being able to measure costs and benefits at each trial would the agency be able to decide whether its last trial produced an improvement. Certainly measuring costs and benefits at a point does not require knowledge of the functions over their ranges, but the distinction in terms of required centralized data seems minor.

The first-order conditions are:

$$\frac{\partial L}{\partial D_1} = -\frac{\mathrm{d}C_1}{\mathrm{d}R_1} - \lambda = 0,$$

$$\frac{\partial L}{\partial D_2} = -\frac{\mathrm{d}C_2}{\mathrm{d}R_2} - \lambda = 0,$$

$$D_1 + D_2 = S_A,$$

from which we see that $\mathrm{d}C_1/\mathrm{d}R_1 = \mathrm{d}C_2/\mathrm{d}R_2 = -\lambda$. Thus, a single charge is optimal, but it can be found only on the basis of knowledge of costs or through a trial and error process. The latter is possible because after each trial the result can be observed at the monitoring point and there is no necessity for the agency actually to measure costs at all. (Of course, even though the proper charge could in principle be found via trial and error, the "errors" imply higher overall costs because of lumpy and at least partially irreversible investments. Thus, the results of proper charges set on the first try are not the same as those achieved without the knowledge necessary to that accomplishment.)

The introduction either of location differences or of a non-conservative residual complicates matters, but not fatally in principle, so long as a single standard (one monitoring point) is still all we have to worry about. The constraint in the agency's problem becomes $\alpha_1 D_1 + \alpha_2 D_2 \leq S_A$, and the first-order conditions from the Lagrangian problem are:

$$\begin{aligned}
&\frac{\partial L}{\partial D_1} = -\frac{\mathrm{d}C_1}{\mathrm{d}R_1} - \lambda\alpha_1 = 0,\\
&\frac{\partial L}{\partial D_2} = -\frac{\mathrm{d}C_2}{\mathrm{d}R_2} - \lambda\alpha_2 = 0, \qquad (8)\\
&\alpha_1 D_1 + \alpha_2 D_2 = S_A,
\end{aligned}$$

so that, for example,

$$\frac{\mathrm{d}C_2}{\mathrm{d}R_2} = \frac{\alpha_2}{\alpha_1} \cdot \frac{\mathrm{d}C_1}{\mathrm{d}R_1}.$$

This result leaves us with some hope for trial and error, because even though charges must be individually tailored, the ratio of any two optimal source-specific charges is the ratio of the sources' transfer coefficients. Thus, trial and error could proceed on the basis of a single "numeraire" charge.

Similarly, if there is more than one standard to be met, but every source affects every monitoring point exactly the same, a single charge for all sources is still optimal. The agency's monitoring problem is more difficult because it must check each point at which a standard is defined, but it can still, in principle at least, perform a simple trial-and-error exercise based on iterations on one charge.

As soon as we both introduce multiple monitoring points and allow location to matter, however, any practical possibility of trial and error disappears. Thus, in this most realistic case, an optimal effluent charge system depends on the agency having knowledge of source cost functions and calculating a set of individually tailored charges. To see why this is so, consider our simple example with a second standard (monitoring point). The agency's problem is:

$$\min L = C_1(X_1 - D_1) + C_2(X_2 - D_2) - \lambda_A(\alpha_{1A}D_1 + \alpha_{2A}D_2 - S_A)$$
$$-\lambda_B(\alpha_{1B}D_1 + \alpha_{2B}D_2 - S_B), \tag{9}$$

and from the first-order conditions:

$$\frac{dC_1/dR_1}{dC_2/dR_2} = \frac{\lambda_A\alpha_{1A} + \lambda_B\alpha_{1B}}{\lambda_A\alpha_{2A} + \lambda_B\alpha_{2B}}. \tag{10}$$

Thus, even if both constraints could be exactly satisfied so that the shadow prices, λ_j, were non-zero, those shadow prices would not be known without a full solution. And without the shadow prices as weights, the ratio of the marginal costs cannot be calculated, even when the agency knows the transfer coefficients. Thus, trial and error cannot proceed on the basis of a single numeraire related in a known and constant way to each other optimal marginal cost (charge). This difficulty is compounded when there are many sources and monitoring points, because quality at some of the latter will inevitably be better than specified by the standard when the standard is not violated at any monitoring point. The corresponding λ's are zero, but which are zero is not known in advance.[16] Thus, while an actual trial-and-error process could lead to a feasible charge set (one that produced the desired ambient quality), it will not in general produce the cost effective outcome.

2.2. Modeling of the realistic static case

These last observations carry us to the end of our discussion of the simple static case and its complication. Overall we have seen just how restrictive are the assumptions that support the classical case for charges – that static regional efficiency can be attained with no knowledge by the agency of the cost functions of individual sources. Two natural enough questions are: If calculation of individually tailored charges is usually going to be necessary, just how hard is it likely to be? And how much difference will various charging systems make? For example, if individually tailored charges are optimal, but a single region-wide charge were actually to be applied to all sources, how much additional cost would be incurred?

The answers to these questions turn out to be specific to particular regions (because specific locations and the nature of the local environments matter); and

[16] Trial and error is difficult because of the large number of "knobs" available to twist in a multi-source region. If each of only 10 sources could control to each of only three levels of discharge, there would be over 59 000 possibilities for an initial trial. That first trial might eliminate some fraction of the options as either infeasible or unnecessarily strict, but finding a feasible and even modestly efficient option might easily involve many very expensive trials.

to particular pollution problems (because the cost-of-control functions differ among residuals as does the behavior of the discharges in the environment).[17] We shall confine ourselves, however, to two examples of modeling efforts designed to mimic realistic regional environmental quality management problems, attempting thereby at least to give an indication of the variations likely to be encountered.[18] These models were chosen because they can be compared both here, where effluent charges are at issue, and in Section 3, where our attention turns to marketable permits of various kinds. After the very briefest of descriptions of the models, we shall summarize some of the lessons learned from them.

The two models we shall use for comparison were both constructed in the early 1970s when enthusiasm for such exercises, and the regional efficiency solutions to which they might lead, was sufficiently great to sustain the costs of development and computation. One, the Atkinson and Lewis model, is of major point sources of particulates in the St. Louis region [Atkinson and Lewis (1974a, 1974b)]. The other is a multimedia, multiresidual model of the Lower Delaware Valley region (referred to here for brevity as Philadelphia) [Spofford et al. (1976)]. The differences and similarities of the models are highlighted in Table 10.1; and there we can see that the biggest differences are in size and complexity. The RFF model contains many more point sources, other residuals discharged both to water and air, and interactions among residuals in treatment processes.[19] In structure, however, and in the important matter of atmospheric dispersion modeling, the two models are similar.

In Table 10.2 some key comparisons are summarized. A policy of uniform percentage reduction orders for all sources sufficient to achieve the desired standard at the worst polluted monitoring station is taken to be the benchmark for compliance costs. (This policy is close enough to that embodied in most U.S. State Implementation Plans (SIP) for air pollution control that we shall follow the studies and refer to it by this acronym.) The other two policy instruments are a regionally (or zonally) uniform emission charge and an optimal effluent charge set. The latter, of course, involves different charges at each source reflecting their different locations relative to the binding ambient quality constraints. Atkinson and Lewis look at primary and secondary particulate standards, while Spofford examines only primary standards, but has results for both particulates and SO_2.

[17] In actual cases, removal processes often display such complications as economies of scale and joint removal of two or more residuals, so that the seeking of optimal regional management solutions, including optimally tailored charges, is much more difficult than our simple example hints at. See, for example, Russell (1973) and Russell and Vaughan (1976) on industrial pollution reduction costs.

[18] See, for other examples: on organic water pollution control in the Delaware estuary, Kneese and Bower (1972) and Johnson (1967); On water pollution control in Wisconsin's Fox River, O'Neil (1980); On SO_2 control in Nashville, Tennessee, Teller (1967); On nitrogen oxide emissions control in Chicago, Seskin, Anderson and Reid (1983); On chlorofluorocarbon control, Palmer et al. (1980); On phosphorus runoff control, Jacobs and Casler (1979).

[19] The sources of cost function data also differ for the major sources. The RFF model uses specially constructed industrial LP models to derive the regional model control vectors for steel mills, petroleum refineries and power plants.

Table 10.1
Summary of model structures and data bases

Model	Basic structure	No. of sources	Residuals included	Sources of cost data	Date of cost data	Basis for air pollution dispersion model	Number of monitoring points
Atkinson & Lewis (St. Louis)	Separable LP	27 point sources (All industrial: 9 power plants, 2 pet. refineries, 4 feed & grain mills)	Particulates	IPP Model[b]	Unclear (probably 1970)	Steady-state Gaussian diffusion (Martin & Tikvart)	9
Spofford (Philadelphia)	LP[a]	183 point sources (124 industrial: 17 powerplants, 7 pet. refineries, 5 steel mills; 57 Area sources home & commercial heating)	*Air* Particulates SO_2 *Water* Biochemical Oxygen Demand (nitrogeneous and carbonaceous)	Specially constructed plant LPs[c] IPP model[b]	Roughly 1970	Steady-state Gaussian diffusion (Martin & Tikvart)	57

[a]The original version was non-linear but the results reported in Table 10.2 come from a new, linear version.
[b]The Implementation Planning Program was designed to operate on air quality control region inventories and to allow the user to specify different control options, producing an estimate of control costs for the region and predicting resulting levels of ambient quality.
[c]See Russell (1973) and Russell and Vaughan (1976) for published examples.
Sources: Scott E. Atkinson and Donald H. Lewis (1974) *A Cost Evaluation of Alternative Air Quality Control Strategies*, Report No. EPA 600/5-74-003 (USEPA, Washington Environmental Research Center, Washington).
Walter O. Spofford, Jr., Clifford S. Russell, and Robert A. Kelly (1976) *Environmental Quality Management* (Washington: Resources for the Future, Washington).
Walter O. Spofford, Jr. (forthcoming) "Properties of Alternative Source Control Policies: Case Study of the Lower Delaware Valley", unpublished manuscript in progress, Resources for the Future.

When particulate matter is the residual of concern, both models produce similar results. Compliance costs are highest for the uniform roll-back approach and lowest (of course) for the optimal charge set. A regionally uniform charge produces intermediate compliance costs in each model. Notice also that as the number of zones increases in Spofford's model, the costs for a zonally uniform charge fall toward the optimal charge result. For the primary standard (75 $\mu g/m^3$) there is even surprising agreement between the models on the relative costs under each of these instruments, though the absolute size of Spofford's costs are very much higher, reflecting a larger number of sources and worse initial quality level. The same pattern holds when Atkinson and Lewis examine the costs of meeting the secondary standard (60 $\mu g/m^3$). The cost savings achievable by

Table 10.2
Compliance costs and emission charges under different emission charge systems in two regional models

	Atkinson and Lewis (St. Louis) Particulates				Spofford Lower Delaware Valley[d]			
	Secondary Std.		Primary Std.		Particulates Primary Std.		SO_2 Primary Std.	
Policy instruments	$\$10^6$/yr	Relative to SIP	$\$10^6$/yr	Relative to SIP	$\$10^6$/yr	Relative to SIP[e]	$\$10^6$/yr	Relative to SIP[e]
Compliance costs								
SIP/uniform percent reductions[a]								
Single regional zone	\$8.3		\$2.0		\$158.0		210.5	
Three zones[b]					115.9	0.73	202.6	0.96
Eleven zones[b]					63.9	0.40	167.5	0.80
Uniform emission charge								
Single regional zone	3.8	0.46	0.3	0.15	14.2	0.09	252.0	1.20
Three zones					12.8	0.08	193.5	0.91
Eleven zones					10.4	0.06	138.5	0.66
Optimal charges[c]	1.9	0.23	0.07	0.03	7.2	0.05	118.5	0.56
Total out-of-pocket costs (including effluent charges)								
SIP/uniform percent reduction								
Single regional zone	\$8.3		\$2.0		158.0		210.5	
Three zones					115.9	0.73	202.6	0.96
Eleven zones					63.9	0.40	167.5	0.80
Uniform emission charge								
Single regional zone	6.7	0.81	1.3	0.64	54.7	0.34	504.1	2.39
Three zones					49.7	0.31	431.7	2.05
Eleven zones					32.7	0.21	295.9	1.40
Optimal charges	3.5	0.43	0.3	0.15	[Optimal charge payments not available.]			

[a] The two models have slightly different versions of this option. Atkinson and Lewis begin with industry-specific, technology-based emission standards and then reduce all discharges uniformly to reach the ambient standards. Spofford begins from the actual 1970 emission levels and reduces all discharges by uniform percentages (regionally or by zone) until the standard is met.

[b] Spofford's zones correspond to political jurisdictions: 3 states and 11 counties.

[c] The optimal charge solution corresponds to the least-cost solution when applied to both the point and area sources in the region.

[d] The Lower Delaware Valley region, which stretches from Wilmington, Del. to Trenton, N.J., includes eleven counties in Delaware, Pennsylvania, and New Jersey.

[e] The numbers in this column are calculated relative to the region-wide SIP alternatives.

going to a more efficient policy instrument are sufficiently great in both models (and for both standards in the Atkinson and Lewis work) that, even allowing for the out-of-pocket emission charge payments, it would be possible to make every discharger in the region better off through a suitable transfer arrangement.

When, however, we look at Spofford's results for SO_2 primary standards (80 $\mu g/m^3$), a very different pattern emerges. The regionally uniform emission charge produces a less efficient outcome than the uniform roll-back. Under zonally uniform charges, the more familiar pattern reasserts itself, but in all cases the call is a close one. In no case is the cost saving enough that the sum of compliance costs and emission charges is less than the compliance cost under the uniform roll-back instrument. This pattern of results happens to depend on Spofford's inclusion of home and commercial sources (the area sources) of SO_2 emissions among those subject to the charge. It can be shown, however, using a simple model like the one used in our earlier discussion of effluent charge properties, that the uniform emission charge is more likely to produce a costlier regional solution whenever sources with high marginal costs of discharge reduction have large impacts on ambient concentrations at the monitoring (standard) point. This is true in the Delaware model, where petroleum refineries with very steep marginal costs of SO_2 reduction (at the high reduction levels required) are also sited very close to the critical monitoring point. The addition of the relatively low marginal cost home and commercial heating sources far from the critical monitoring point accentuates this tendency and produces the result observed by Spofford.

Thus, how seriously one takes the Spofford results depends to some extent, though by no means entirely, on how seriously one takes the idea of applying an emission charge to small dispersed sources. (Note that such application could be via a fuel sulphur-content charge, so need not depend on unrealistic assumptions about monitoring and enforcement capabilities.) Nonetheless, the fact that in particular circumstances such results *can* be observed should make us cautious about general rule ranking policy instruments. While it is true that a tailored charge set can produce large savings, it is not always true that a uniform charge can improve on a simple regulator approach – even when we confine our attention to compliance costs. When we add in potentially massive transfer payments produced by the charge we can understand why sources might be extremely reluctant to see this instrument adopted. The only general rule would seem to be that if we want to explore alternatives in real settings we ought to do so with models first and only after we have an idea of the range of useful options, propose policy changes.[20]

2.3. Other dimensions of judgement

As important as static efficiency and economy of centralized information may be in the economic literature on environmental policy instruments, we must consider

[20] However, on the legal issues surrounding actual use of models for computing optimal or other charge sets, see Case (1982).

other dimensions of judgement as well. And these dimensions can well be more important to the adoption and long-term success of an instrument than the more familiar arguments.

2.3.1. *Ease of monitoring and enforcement*

Monitoring pollution sources to ascertain that they are paying the proper emission charge is a difficult problem. But a central point, as we see it, is that the monitoring problem is no harder if an emission charge is involved than if compliance with emission standards or permit terms is the concern. Thus, criticisms of emission charges based on the claim that compliance is harder to monitor are incorrect when the alternatives are also concerned with limiting the discharge of residuals per unit time. However, in a richer model including not only the statistical nature of the monitoring problem but also the decentralization of monitoring and enforcement activities and the possibility of polluter actions to conceal true discharge levels, Linder and McBride (1984) have identified certain drawbacks to a charge system not shared by a discharge standard. These include possible encouragement for less aggressive monitoring.

2.3.2. *Flexibility in the face of exogenous change*

It is first necessary to be clear about what counts as "flexibility". We shall use that word to mean the ease with which the system maintains the desired ambient standards as the economy changes. The most important measures of "ease" are first the amount of information the agency has to have and the amount of calculation it has to do to produce the appropriate set of incentives for a new situation and, second, the extent to which adjustments involve a return to a politically sensitive decision-making process.

In the restricted situation in which charges are both efficient and independent of costs (known, linear damage functions) the case for charges remains impressive. In fact, the same charge remains optimal after the addition of a new source or the expansion or shutdown of an existing source so long as change does not shift the marginal damage function. This automatic adjustment is thus based on allowing changes in discharges and ambient quality levels while maintaining marginal damage equal to marginal cost at each source.

If the policy goal is to maintain an ambient standard at a single monitoring point, after a change the charge must be adjusted, but the convenient relationship among optimal charges based on transfer coefficients is still there to take advantage of.

In the general case, where location matters and ambient standards are the goal of environmental policy (or where damage functions are either unknown or non-linear) emission charges do not protect ambient quality unless they are adjusted by the agency as change occurs. Such adjustment requires new calculations if the charges are to be efficient. (And then, because the charges are individually tailored, each charge is a fresh chance for political action.)[21] If the actual charges used are uniform and set by trial and error, adjustment will involve the expense of error, and, in addition, static efficiency will not be achieved.

2.3.3. *Dynamic incentives*

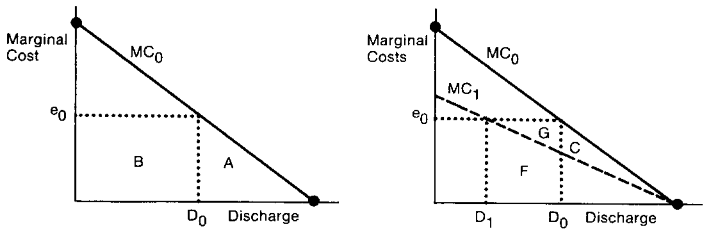
In the matter of incentive to technical change, the simple general rule may be summarized as follows. If compliance with an order is costly and if there is some choice of how to comply (what equipment or technique to use) then there will be an incentive for the source faced with the order to seek cheaper ways of complying in the long run. It is also true that for any particular source, an incentive system that puts a value on the discharge remaining after control will create a greater incentive to change than will a regulation specifying that same level of discharge.[22] We shall return to this matter when discussing the regulatory approach in Section 5.

2.3.4. *Political considerations*

Two broad questions should be dealt with here: distributional problems and ethical arguments. As for the first, it is obvious that emission charges in their pure form are bound to cost any particular source more than would a simple emission standard designed to achieve the same discharge at that source. Such evidence as that from cost models, both simple and complex [e.g. Vaughan and Russell

[21] For a discussion of the inevitability of political bargaining over emission charges, see the fine discussion by Majone (1976).

[22] This is easy to show. In the figure below, the firm's initial marginal cost-of-discharge-reduction cure is MC_0. Assume it is complying with an order to discharge no more than D_0. This could also be achieved by the agency charging a *fee* of e_0 per unit of discharge. The order costs the firm area A, the cost of control to D_0. The charge would cost it area $A + B$, the control cost *plus* the total fee paid on remaining discharges. If, as shown in the second panel, the firm can find a way to reduce its costs to MC_1, it saves C under the order system and $C + G$ under the charge.



The new discharge, D_1, under the charge system is lower as well. This result also applies to marketable permits, for the permit price corresponds to the charge even though it may not be paid out of pocket by the originally permitted source. This argument is set out more fully and formally by Wenders (1975). For a slightly different view, see Magat (1978). And for another analytical approach, see Reppetto (1979).

(1976)], suggest that out-of-pocket costs of achieving a particular emission level at a source may easily be doubled by charge payments. On the other hand, the appropriate comparison should be the regional setting for a given ambient standard and real policy instrument alternative. Thus, if the efficient set of charges is contrasted with an inefficient set of emission standards, such as that obtained by imposing uniform percent reduction requirements on all dischargers, it is, as we have seen, an open and region-specific empirical question whether or not the savings from better distribution of pollution control effort will leave none, some, or all of the sources better off under the efficient charge, even after allowing for the charge payment itself. The real political problem here may be that dischargers doubt that the efficient charge set would ever be found or applied and see that an inefficient charge has a much increased chance of just costing them more money for the same results in the short run.[23]

One response to this political problem has been the proposal to use the revenue from charges to subsidize other acts of environmental protection. Another response has been concern over whether or not those revenues should be used to compensate the sufferers of damages from the remaining pollution. Certainly the idea has political appeal and seems to provide a symmetry otherwise lacking in the charges approach. But economists appear to have agreed after some debate that this symmetry would in fact be undesirable from an efficiency point of view; that while polluters should in principle pay charges equal to the marginal social damages they cause, damaged parties should absorb those damages without compensation and not be subjected to the incentive to increase exposure to pollution to collect (additional) compensation. [See, for example, Baumol and Oates (1975), Fisher (1981) and Olsen and Zeckhauser (1970). For a discussion of some ethical aspects of the compensation issue see Chapter 5 of this Handbook.] Only slightly more palatable to economists, but politically attractive, is the alternative already mentioned of using charge receipts to finance pollution control investments, especially those of an inherently collective nature such as in-stream aeration facilities or low flow augmentation dams.

The second political question, that of ethical stance, will be mentioned only briefly. The question arises because to many people pollution is wrong, not morally neutral.[24] These people do not want to see decisions about pollution placed on a footing symmetric with the firm's decisions about purchasing "normal" inputs such as labor services or packing cases. They want pollution stigmatized by strongly worded laws with strictly defined discharge limitations and criminal penalties for violations. The polluter's ability to choose how to react to a charge, the heart of the economist's efficiency case, is also the heart of the

[23] Distributional impacts on competitive industries are analyzed under a variety of assumptions by Dewees (1983).

[24] There is no evidence about what part of the general population feels this way, but Kelman's interviews with congressional staff members and active Washington environmentalists reveal a preponderance of this view among Democratic staffers and the environmentalists [Kelman (1981)].

environmentalist's opposition (for further discussion see Chapter 5 of this Handbook).[25]

A summary of this discussion of emission charges as a policy instrument for pollution control reveals a distinctly mixed bag of features. Certainly the classical position, in which static efficiency, information economy, and automatic adjustment to exogenous change can all be obtained at once, rests on very restrictive assumptions. In the more general case, static efficiency must be purchased at the cost of both information economy and flexibility in the face of change. Beyond that, emission charges suffer in the political arena from their distributional disadvantages (potentially large transfers imposed on polluters) and their ethical "flavor", which is apparently entirely too neutral to suit those who judge pollution to be a moral rather than a technical problem of market failure. In later sections we shall see how other instruments look under the same light.

3. Incentives to complete the set of markets: Tradeable rights

In practice the commonest form of policy instrument for environmental policy is the pollution permit, the terms of which usually embody either technological specifications or discharge limitations. We have explored some of the advantages and disadvantages of replacing such specifications with administratively set prices on discharges. Another possibility is to create a situation in which prices are attached to discharges by a decentralized, market-like process. To achieve this permits must be tradable among the interested parties, and the supply of permits must be less than the potential demand at zero price.

The idea of a marketable permit system appears to have occurred first to Crocker (1966) and to have been set out more completely by Dales (1968a, 1968b). It amounts to the dual of the emission charge idea – quantities instead of prices are set administratively; prices instead of discharge totals result from the free choices of those subject to the system. Its development in the literature has roughly paralleled that of charges. Early formulations were simple and compelling but later analysis showed that introducing complications reflecting features of reality reduced that apparent attractiveness. [For an excellent recent review, see Tietenberg (1980).] Just as with the charges, alternative versions of marketable

[25] It is worth noting in passing that the early writers may have unwittingly encouraged the views that those who favor economic instruments are basically insensitive to the health of the environment. For example, Kneese in his classic 1964 work, gives as examples of policies leading, potentially at least, to more efficient regional policies, the dedication of an entire river to waste disposal (the open sewer idea) and the storage of residuals for discharge in times of high assimilative capacity. In an illustrative example he also appears to sanction pollution-caused fish kills if the costs of cleanup are not exceeded by the damages to downstream commercial fishing interests. None of these are intrinsically wrapped up with emission charges and any or all might or might not be justifiable on the basis of efficiency analysis in a particular situation. But the political realities in the United States at least have made it clear that these are unacceptable alternatives. Their appearance in a fundamentally important statement of the value of emission charges probably tainted the latter.

permits have been defined, having different static properties and different implications for information and calculation loads. In discussing these cases and their properties the idea of duality will provide a useful benchmark, though in some cases this must be interpreted broadly.

Moving beyond the simple static context to complex (but still static) regional models, we shall observe as expected that employing marketable permit systems can lead to substantial costs savings compared to regulatory methods.[26] When we expand the horizon to include other dimensions of judgement, such as flexibility in the face of exogenous change, we shall continue to find a broad notion of duality useful for putting our findings in perspective. We shall, however, find that in some respects charges and marketable permits have the same properties while in others they are different without being dual. As before, at each stage we shall pick and choose among the many issues that have interested economists but shall endeavor to provide citations where we avoid discussion.

3.1. *Simple static cases: Efficiency and information*

Strictly speaking the benefit-based arguments for charges do not have duals in the set of marketable permit systems. It is when we introduce constraints on quality that we find price- and quantity-setting systems to be dual. But it is worthwhile nonetheless to observe that some early (and even not so early) statements of the case for marketable permits introduced an assumption that was conceptually equivalent to assuming knowledge of the benefit functions. This was the assumption that environmentalists (those with tastes for a clean natural environment) could and would combine to buy and retire pollution rights, thus carrying the system toward a socially optimum level of pollution analogous to that reached by the optimal benefit-based emission charge set.[27] But this assumption is fully as unrealistic as that involving benefit functions. The problems of public goods and free riders that imply no markets in environmental services, hence no demand or benefit functions available from directly observable behavior, imply that such combinations would be very difficult to establish. Even the analogy of the environmental groups, which combine thousands of individuals into potent forces for pollution control, cannot help us here. These groups succeed through highly

[26] In modeling studies of permit systems the model is almost always asked to produce the optimal (post trade) allocation of a fixed supply of permits and not to mimic the set of trades that leads there. As we shall see, it is not necessary for the control agency to have a complete model to introduce a statically optimal permit system even in the general case when such modeling *is* necessary to find an optimal emission charge set.

[27] Emphasis should be placed on "analogous". Because the social choice process contemplated by this argument for marketable permits is completely different from that involved in voting for standards or even calculating an optimal result using costs and benefits "to whomsoever they may accrue", there is no reason to expect the same quality levels to be thrown up by the three processes – assuming for the moment that environmentally minded citizens *could* combine to purchase and retire rights.

leveraged lobbying and litigation, not competition in the market. Rough calculations strongly suggest that all the national environmental groups in combination could make only a small dent in the pollution problem of any single large urbanized region if they had to do it by purchase of rights [Oppenheimer and Russell (1983)].[28]

More realistically, marketable permit systems are seen as potential instruments for achieving chosen ambient quality goals.[29] Corresponding to the single regional emission charge, which we saw was optimal only in very special circumstances, the simplest marketable permits system involves a single regional total emission limitation and a market for emission rights equally valid at any location in the region. These permits trade at a single regional price. Such a system can produce the desired ambient quality at least cost when location of discharge does not matter. The optimum level of total discharge for the given ambient standard could in principle be found by trial and error – the largest total just allowing the standard to be met.

In other situations, specifying a regional emission total and permitting trades among all dischargers on a pound-for-pound basis is not optimal, just as the single charge is not. While market transactions would result in an allocation of the permits such that resource costs were minimized for that total, one of the following would be unavoidable.

(a) the ambient quality goal would not be met; or

(b) if the initial total were chosen so that no conceivable set of trades could result in violation of the ambient standard, then the cost of meeting the standard would certainly be higher than necessary; or

(c) even if the total were greater, so that the standard were met only after some particular predicted trades, there would in general be a cheaper way of meeting it.

Trial and error could, however, be used to find a total allocation such that after trading the ambient standard was nowhere violated.[30] The trials would involve specification of the total permits to be allocated and observation of resulting quality. The same problems of extra cost arising from irreversible investment decisions arise here as in the use of trial and error with a charge system.

At the other end of the scale is the ambient rights system where the rights specified and traded are rights to cause pollution by particular amounts (usually assumed to be steady state concentrations) at the specified monitoring points. In

[28] Another reason that rights markets are unlikely to achieve a socially efficient outcome is that interfirm pollution effects may produce nonconvexities in production sets of nonpolluters who are allowed to buy and sell permits. Multiple optima then exist and the final result will be sensitive to the amount of rights issued initially. See Crone (1983) and Tietenberg (1983).

Rose (1973) analyzed systems of permit allocation using iterative bids and responses keyed to a known non-linear damage function. This provides another link to the optimal charge literature.

[29] We postpone for now the matter of how the permits might be initially distributed. This is discussed briefly under distributional matters in Section 3.3 below.

[30] How carefully the standard is protected depends on how many monitoring points are specified. The fewer of these the higher the chance that an after-trade allocation will result in an undetected violation (a "hot spot").

the ideal ambient rights system the agency simply defines rights totals at each ambient monitoring point equal to the difference between the desired standard and the contributions of all sources not subject to the system.[31] It has been shown by Montgomery (1972) that from any original allocation of these ambient rights the least cost regional solution can be reached by decentralized trading.

This system, then, sounds very appealing. Virtually nothing need be known by the agency except what amount of ambient quality "capacity" is available to be allocated. The market does the rest, without central calculation. The problem is that the *decentralized* information problem is formidable. Each source must simultaneously decide its optimal moves in each of the several markets, because any changes in its discharge rate simultaneously affects its need for ambient rights in every market.[32] If each source can be assumed to be a price taker in every market, the system looks like a set of competitive factor markets and we can invoke familiar market stability theorems to reassure ourselves that the optimal trading *could* go on. With only a few large buyers and sellers in each market, however, the practical chances for optimal decentralized results fall substantially. Thus, the centralized information intensity of the optimal charge system has its "dual" in the decentralized information problem of multiple markets in ambient quality permits.

Compromises between the extremes have been proposed. In zoned emission permit systems [e.g. Tietenberg (1974), Atkinson and Tietenberg (1982)] the region is divided into subregions, emission permit subtotals are allocated to subregions, and within subregions to sources. A source can trade pound-for-pound within its subregion and not at all outside it. If the initial allocation does not violate the standard, the zoned system raises the chances that no allowable set of subsequent trades will do so.

The zoned system raises in a more insistent way a problem we have so far ignored: market thinness. Tradable permit systems depend for their desirable properties on trades taking place and on these trades being sufficiently frequent to

[31] Such sources are usually termed "background", meaning such contributions as those blowing (or flowing) in from other regions or those from natural, uncontrollable sources in the region. More completely, however, the allocated permit totals can only equal whatever is left at each station when all sources not required to hold permits are operating in accordance with assumed regulatory requirements. Thus, the contribution of home heating discharge to regional SO_2 and particulates could be estimated using assumptions about fuel quality requirements.

[32] For a given initial allocation of ambient quality rights, q_{ij}^0 to each source i at each point j, each source must solve the problem:

$$\min_{\text{s.t.}} C_i\left(X_i - D_i^1\right) + \sum p_{q_j}\left(\Delta q_{ij}\right)$$

$$\alpha_{ij} D_i^1 \leq q_{ij}^0 + \Delta q_{ij}, \quad \text{for all } j,$$

where $C(\cdot)$, D, X, and α are as defined in Section 2 and superscripts denote before and after trading situations, q_{ij} is the initial allocation of ambient quality permits at point j to source i, Δq_{ij} is the change through trade in the number of permits held at j by i, where purchases are plus and sales are minus and p_{q_j} is the price of permits at point j.

establish a market clearing price (regional emission permits) or a number of market clearing prices (ambient permits, zoned emission permits). If there are only a few sources in each market there may be no transactions for many periods because of capital commitments in particular production or discharge control process. Or transactions may be distorted by monopolistic or monopsonistic (duopolistic or duopsonistic) behavior by a dominant source or sources. These problems are major concerns of Hahn and of Cass et al., who have worked on designing an SO_2 discharge permit system for the South Coast Air Basin in California [Hahn (1980), Cass et al. (1980) and Hahn and Noll (1981)]. See also Russell (1981) for some preliminary evidence on numbers of sources and the supply and demand for permits. Several workers in this field, for example Tietenberg (1974) and David et al. (1980) have advocated periodic expiration of rights, making them like leases rather than freehold properties, with the idea that when some or all of a source's permits expired it would be forced into the market to obtain replacements.[33]

Another compromise emission permit system depends on "trading ratios" related to the source-specific transfer coefficients. If it is possible to identify a hot spot in advance, the coefficients relating all source discharges to that point can be used. Then, if source i sells to source j e_i units of emission permits, source j can use (discharge) α_{ik}/α_{jk} (e_i) units where k is the designator of the potential hot spot.[34]

3.2. *Evidence from regional models*

It will be useful here, as it was in our discussion of emission charges, to introduce some evidence from realistic regional models. In order to maintain comparability

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If the p_{qj} were exogenously given, this calculation would be straightforward for any source. But for the decentralized system the p_{q_j} are only implicitly defined by the market-clearing relations:

$$\sum_i \Delta q_{ij} = 0, \quad \text{for all } j,$$

and

$$\sum_i \sum_j p_{q_j} \Delta q_{ij} = 0.$$

[33] This strategy is also liked by some writers to the maintenance of agency "flexibility" – the ability to retire permits without the cost or fuss of litigation over the taking of property. See the discussion below under flexibility.

[34] A complete system of implicit trading ratios constraining trades has been suggested by Krupnick and Oates (1981) and Krupnick, Oates and Van de Verg (1983) who refer to it as an "offset system". This scheme protects ambient quality at all monitoring points (points for which transfer coefficients are available). In fact, however, the constraints faced by each source in deciding how to trade seem to be equivalent to those involved in the ambient permit system when source-specific constraints are combined with the zero net creation of permits at each monitoring point. For a system aimed at maintaining the status quo quality if that is better than the standard, see McGartland and Oates (1983).

Table 10.3
Compliance costs under different marketable permit systems in two regional models

	Atkinson and Tietenberg (St. Louis) Particulates				Spofford (Philadelphia) Particulates Primary Std		SO_2 Primary Std	
	2 $\mu g/m^{3d}$ $\$10^6$/yr	Rel to SIP	10 $\mu g/m^{3d}$ $\$10^6$/yr	Rel to SIP	$\$10^6$/yr	Rel to SIP	$\$10^6$/yr	Rel to SIP
SIP/uniform percent reduction[a]	$9.8		$6.2		$158.0		$210.5	
Emission permits[b]								
Single zone	8.0	0.82	1.5	0.24	16.0	0.10	199.4	0.95
Three zones[a]	6.9	0.70	1.5	0.24	16.1	0.10	204.6	0.97
Six zones	8.6	0.88	1.8	0.29				
Eleven zones					23.3	0.15	215.2	1.02
Ambient permits								
Single Market	3.5	0.36	0.6	0.10	–		–	
Multiple Markets	3.1	0.32	0.5	0.08	9.7	0.06	177.1	0.84

[a] The Atkinson and Tietenberg SIP strategy involved first assigning to each of 27 sources an emission level based on application of control strategies represented in SIP guideline documents. To produce the level of ambient pollution at the worst receptor point shown in the table, further necessary reductions were made on an equal-percentage-reduction basis. Spofford's version of this policy involves equal percentage reductions at all sources from a base of 1970 inventory emissions.

[b] For the emission permit and ambient permit systems, Spofford imposes fuel quality regulations on home and commercial heating activities. These activities do *not* participate in the permit markets.

[c] Atkinson and Tietenberg report on two slightly different versions of a three-zone permit system. The costs reported here are a rough average of those reported in their article (Figure 4) for the two versions.

[d] Contribution to annual average concentration of suspended particulates at receptor location with worst quality of the 27 point sources modeled. Nothing is said about what value of this indicator might correspond to the primary air quality standard of 75 $\mu g/m^3$. Results are given for levels of this indicator from roughly 2 to 12 $\mu g/m^3$.

Sources: Scott E. Atkinson and T.H. Tietenberg (1982) "The Empirical Properties of Two Classes of Designs for Transferable Discharge Permit Markets", *Journal of Environmental Economics and Management* 9, no. 2, 101–121.

Walter O. Spofford Jr. (1983) "Properties of Alternative Source Control Policies: Case Study of the Lower Delaware Valley", Resources for the Future, unpublished report.

across instruments, we shall again concentrate on two such modeling projects: Atkinson and Tietenberg's work on particulate control in the St. Louis region;[35] and Spofford's analysis of particulate and SO_2 control in the Lower Delaware Valley Region (Philadelphia).[36]

Some control-cost results from these two modeling exercises are summarized in Table 10.3. These must be interpreted with caution, because the ambient stan-

[35] The Atkinson/Tietenberg work is based on the same model as that of Atkinson and Lewis (1974), used in the emission charge section.

[36] Again, this by no means exhausts the possibilities. Other studies providing modeling evidence include: deLucia (1974) on BOD discharge permits for the Mohawk River in New York; Cass et al. (1980) and Hahn and Noll (1981) on SO_2 discharge permits in the South Coast Air Basin in California; Eheart (1980) on BOD discharge permits for the Willamette River in Oregon; David et al. (1980) on phosphorus discharge permits for Lake Michigan; O'Neil et al. (1981) on BOD discharge permits for the Fox River in Wisconsin.

dards imposed on the models were not the same. Atkinson and Tietenberg (A&T) report the contribution of 27 major point sources to quality degradation at the receptor location with the worst air quality. Spofford imposes the primary air quality standards at each of 57 monitoring points in the region.

Subject to this caveat, however, the pattern of results is of some interest. Most obviously, the A&T results for the less strict "standard" (10 $\mu g/m^3$) look very like Spofford's results for the primary particulate standard. Either type of permit represents a very large improvement over the SIP/uniform percentage reduction policy, with the ambient permit system costing 10 percent or less of the strictly regulatory approach (in terms of real compliance costs only).

For both the stricter particulate standard in A&T's work and in the SO_2 example from Spofford, however, the relative cost differences change. In the former these drop off less dramatically. In the latter, the emission permit systems represent either no cost improvement or only the tiniest of improvements, and even the ambient permits are well over 80 percent as expensive as the SIP policy.

Thus, again it appears that the rankings of policy instruments, even in static efficiency terms, will in general depend on the residual in question, the strictness of the ambient standard being contemplated, and the characteristics of the regional economy and environment. We cannot even be certain that the theoretically best ambient permit system will be the lowest cost alternative because of the important of such small sources as home heating, for which permit requirements and trading seem completely out of the question.

3.3. *Other dimensions of judgement*

Marketable permit systems display both similarities to and differences from emission charges when judged on such dimensions as ease of monitoring and enforcement, flexibility in the face of exogenous change, dynamic incentives, and political attributes. We consider these in turn in this section.

3.3.1. *Monitoring and enforcement*

Monitoring an emission permit, marketable or not, defined in terms of allowable emissions per unit time, is the same problem as monitoring for emission charge billing. When permits are marketable, the problem may be compounded by the necessity of being current with completed trades. And this extra difficulty might be exploited by dischargers trading in the short run to stay one jump ahead of agency monitoring teams.[37] Problems are compounded if trades are allowed between conventional sources such as stacks and hard-to-monitor sources such as dirt roads and refuse piles.

[37] This strategy could be foiled by requiring long minimum holding periods, but this would have to be backed up by a complete, real-time inventory of all permits. David et al. (1980) propose that all trades take place only at quarterly auctions as another strategy to assist in monitoring for compliance.

One extra fillip accompanies an ambient permit system, however. The current state of technology does not in general allow us to differentiate the contributions of specific dischargers to concentrations of pollutants observed at an ambient monitoring point.[38] This means that monitoring for compliance in this case must also involve monitoring of discharges. That is, a portfolio of ambient permits must be translated into an effective discharge permit by use of an agreed-on regional environmental model.

3.3.2. Flexibility in the face of change

This is a dimension on which a marketable permit system seems to have a distinct advantage. Once established, and assuming necessary monitoring and enforcement effort, a permit system maintains either discharge totals (regional or zonal) or ambient quality standards without constant intervention and recalculation by the agency. If the demand for permits shifts because of regional growth or decay, this will be reflected in the market prices of permits. Permit reallocation takes place as firms find it in their interest to reduce discharges and sell permits to new entrants or expanding resident firms.

With reallocation through trading of emission permits goes the continued danger of new hot spots.[39] This danger, plus the thought that the initial allocation might be regretted for other reasons, has inspired several analysts to push for a different kind of flexibility – bureaucratic ability to reduce the total of emission permits outstanding without compensation [e.g. Tietenberg (1974), deLucia (1974)]. This flexibility would be obtained by automatic and periodic expiration of rights (e.g. one-fifth might expire every year). There would be no obligation to reissue the same number that expired, and in some systems, all new rights would be auctioned. This particular form of flexibility seems to threaten the real long run advantages of permit systems, however, for decisions to buy and sell permits would become shorter run matters if expropriation after only 5 years were a real possibility.

3.3.3. Dynamic incentives

In principle, the incentives for technical change provided by permits correspond to those produced by charges. In either case, reducing discharges produces a

[38] But see footnote 47 on inferring discharges from the multiple sources affecting multiple monitoring points on the basis of knowledge of the elements in each discharge stream.

[39] Notice that by a suitably conservative choice of initial allocation the agency could avoid all possibility of hot spots no matter what the pattern of trading. This would in general imply a very severe restriction on total permits and thus a high control cost. One place for modeling, then, as in our empirical section, is to identify the efficient post-trade pattern of discharges so that the initial total allocation can be such as to produce the desired ambient standard under that spatial pattern of discharge. Thus, costly information again can substitute for costly discharge control.

monetary gain to the source. However, it may be difficult to sell any substantial number of permits, especially if the market is thin; hence, a (full) monetary gain may not be captured as easily under the permit system as under the charge system. Moreover, for very strict initial allocations of emission permits designed to avoid hot spots under any possible set of trades, the permit price will be higher than that implied, for example, by an ambient quality permit system. Thus, static inefficiency can produce larger long-run incentives to reduce discharges.

3.3.4. *Political considerations*

The distribution of costs under a marketable permit system depends on both the number of permits originally allocated and on how the allocation is done. Auction systems are conceivable [e.g. Rose (1974), Repetto (1979) and, for a "Vickrey Auction", Collinge and Bailey (1983)] and produce a result similar to emission charges, with all sources being out of pocket for both control costs and permit costs. More likely would seem to be free initial allocation, either in proportion to original, uncontrolled emissions or to a projected equilibrium allocation. In either case, the value of the issued permits is a windfall to the existing sources. This may purchase their acceptance of such a system, where they seemed likely (though not certain) to oppose an emission charge. The other side to this coin is the opposition such a windfall is likely to create among environmentalists – and, indeed, others.

The other political consideration we have mentioned is the extent to which the instrument stigmatizes polluting activity and appears to give the polluter no choice but to clean up. On this scale, the marketable permit looks modestly preferable to the charge. The chance to pay and pollute without committing a "violation" is limited by the total number of available permits, not merely by the arguments of economists who assume rational cost-minimizing behavior. While permit violations are possible the very use of the word "violations" indicates that such behavior is considered wrong and presumably subject to punishment.

3.4. *A real-world approximation*

More significant than intricate efficiency arguments, modeling exercises, and speculation about political considerations is the fact that an approximation to a marketable emission permit system is now in place for air pollution control in the United States. This system has been developed out of a combination of necessity and imagination by the USEPA and certain of its contractors. It has three major components:

(1) *Offsets* – arrangements that allow a new or expanding source to buy into an area by paying for the reduction of emissions at other sources. The reductions must more than balance the new source's emissions, and the new source must

meet applicable regulatory requirements such as new source performance standards. [See, for example, Liroff (1980).]

(2) *Bubbles* – originally designed to let a single complex plant balance its pollution control effort among its several stacks in such a way as to reduce its costs while simultaneously reducing its emissions. The idea is basically to relax specific regulatory requirements at one or another high cost process in return for extra effort at a place where extra removal comes cheap. The idea has subsequently been expanded to allow multiplant bubbles which amount to permit trades among existing sources (e.g. Brady and Morrison, (1982)].

(3) *Emission Reduction Credit Banking* – This feature allows sources that have opportunities to reduce emissions but no current markets in which to sell the freed up "permits" to bank them for later use or sale [Brady and Morrison (1982)]. The system represented by these related features is complicated and constrained by the apparatus of direct regulation that has been retained. This apparatus limits the extent of control effort relaxations a source can buy, limits the circumstances in which trades can take place (both in terms of source compliance with regulatory requirements and of regional compliance with ambient quality standards) and introduces separate and to some extent inevitably *ad hoc* approval procedures for each desired trade. On the other hand, the regulatory apparatus introduces possibilities for unwanted outcomes. For example, existing permits under State Implementation Plans may allow sources far more discharges than they are using or indeed have ever used. These excess emissions are apparently available for trade and the results have been damned as "paper offsets" when used [Liroff (1980)].

An analysis of the actual cost and discharge results of operation of this system must wait on more experience. What seems likely at this point is that many proposed and approved trades have involved notional or paper offsets or their equivalent in bubbles – as when dirt roads are to be oiled to cut ground level dust in exchange for relaxation in high-level particulate emission requirements. On the other hand, the mere existence of the system and experience with its operation can give us confidence to go on into better structures.

4. Other incentive systems

4.1. Deposit–refund systems and performance bonds

As we have seen, remedies such as charges or marketable permits require that discharges be monitored. This may not be feasible in practice, i.e. when the sources of environmental degradation are numerous and/or mobile. Moreover, a system of charges or marketable permits provides incentives for concealing the volume of discharges, which may jeopardize reliable monitoring. For these reasons, such systems are not likely to work in many cases, such as releases of freons from automobile air conditioners; improper disposal of mercury batteries or waste lubrication oil and other hazardous material by individuals; or littering, be it beer cans or abandoned cars.

Similarly, the establishment of property rights through appropriate liability assignments (discussed below in this section) runs into many problems that limit its use. For example, proof of guilt is required and often difficult to establish even in cases where proof of innocence would be easy, had it been required or had there been incentives to provide such evidence. Furthermore, the exact implication of liability may be unclear, in particular concerning the size of indemnities for a given type of violation, which makes the deterrent role of this instrument unclear. In addition, if the probable size of indemnities exceeds the net worth of the violator, the incentive effect on behavior as well as compensation to the injured party (when relevant) may be insufficient.

General deposit–refund systems may be a better instrument in such situations. These systems imply that the potential injurer is subjected to a tax (deposit) in the amount of the potential damage and receives a subsidy (refund), equally large in terms of present value, if certain conditions are met, e.g. proof that a product is returned to a specified place or that a specified type of damage has not occurred. Thus, such systems introduce a price for the right to inflict detrimental effects on the environment and a (negative) price if this right is not used. As a special case, the government may not be involved at all, instead the separate tax payment is set to zero and the subsidy payment is required to be made by a non-government party engaged in damage-related transactions with the potential injurer (for example, sellers of beverages in certain types of containers). This party would typically respond by increasing prices for such transactions and by introducing a "deposit" as part of the new price. The resulting arrangement amounts to a so-called "mandatory deposit" where the sole requirement is that a refund be made (e.g. mandatory deposits on beverage containers). As another special case, the potential injurer may be allowed to transfer the liability to pay the *net* tax/deposit, i.e. when the conditions for a subsidy/refund are not met, to a trusted third party such as a bank or an insurance company. This amounts to a "performance bond" for which the potential injurer will have to pay some interest or a premium [Bohm (1981)].

Deposit–refund systems may perform better than alternative instruments in that (a) they also work when the act of environmental degradation is not directly observable or when the potential injurers are numerous and/or mobile, (b) they simplify the proof of compliance in some cases, (c) they specify the (maximum) economic consequences of noncompliance, (d) actual or expected damages are covered by actual payments, at least in principle, and (e) in certain applications they may stimulate people other than those directly involved to reduce the effects on the environment (such as scavengers in the case of refunds on littered items).

In addition, as compared to alternative economic incentive systems such as pure charges or pure subsidies, deposit–refund systems have properties that would make them more attractive from the politician's point of view. Charges have sometimes been avoided because of fears that low-income people would be found to be hit relatively hard by such measures. In contrast, taxes/deposits are balanced by the right to subsidies/refunds which would leave nominal income unaffected. Indeed, the refund incentives may be particularly strong for low-income people and allow them to make income gains on balance. Subsidies have to be financed by government revenue and are disliked for that reason. In contrast, the specific taxes/deposits cover the subsidies/refund in a deposit–refund system.

Thus, deposit–refund systems – when applicable – can be said to provide the advantages of an economic incentive system, while avoiding some of the political disadvantages of the "traditional" forms of such incentive schemes. The applicability of such systems requires that it is technically feasible and not prohibitively expensive to establish proof of absence of pollution from the potential polluter.

4.1.1. Forms of deposit–refund systems

4.1.1.1. Adjusting market-generated systems.. The fact that deposit–refund systems (or refund offers) are found in the market economy indicates that there exists empirical experience with such systems. The reasons for the emergence of market-generated systems are diverse, ranging from a reuse value (e.g. old tires, containers) or a recycling value (e.g. lead batteries) to price differentiation or the speeding up of replacement purchases by refund offers to old customers. Thus, the rationale for voluntary refund offers may be that the reuse or recycling value (V) is positive or simply that a refund prospect (R) stimulates demand; in the latter case, V may be negative.

As some monopoly element is likely to be present when an individual firm makes a refund offer, we may write the profit function as

$$\Pi = p(Q, R)Q - c(Q) + r(R)(V - R)Q,$$

where Q is output;[40] $p(Q, R)$ the inverted demand function; $c(Q)$ the cost function; and $r(R)$ the return rate. The return rate will be determined by the individual consumer's (i) disposal options, where c_{d}^{i} is the total unit disposal cost without a return alternative and c_{R}^{i} the corresponding cost of returning the product. Consumers whose $c_{\mathrm{R}}^{i} - c_{\mathrm{d}}^{i}$ falls short of R will be assumed to choose the return alternative.

[40] To fix ideas, the output may be considered as a quantity of bottled beverages or mercury batteries. Later when we deal with government-initiated deposit–refund systems, a better illustration may be provided by the production of freons; here, freons (chlorofluorocarbons) could be returned for a refund (instead of being released into the atmosphere) when cooling equipment is being serviced or scrapped.

The introduction of a refund offer can normally be expected to raise prices. However, for a given demand effect $\partial p/\partial R > 0$, and a given effect on the return rate, $\mathrm{d}r/\mathrm{d}R > 0$, a sufficiently high V value will cause equilibrium prices to fall [see Bohm (1981, ch. 2)]. Regardless of this outcome it is up to the firm to announce that part of the price now is a "deposit" $D = R$.

If non-returns typically create negative external effects in the amount of E (expected environmental hazards or extra waste treatment costs), the firm's optimal $R(R_F)$ may not give rise to a socially optimal return rate. Assuming for simplicity that V also equals the social reuse value and that second best complications from the monopolistic behavior of the firm can be disregarded, the socially optimal $R(R_S)$ would equal either $E + V$ (where $V \gtreqless 0$), if the return rate is less than 100 percent, or the lowest level $R < E + V$ at which a 100 percent return rate is attained. Thus, if $R_F < R_S$, an adjustment of the market refund rate may be called for. A "mandatory deposit" in the amount of R_S may, however, create problems, as the firm would lose when refunds are set at a level other than R_F. Hence, the firm might want to obstruct the system by making returns from consumers more cumbersome (increasing c^i_R). If so, either measures specifying the obligations of the firm would have to be taken or the government would have to become financially involved in the administration of the system. The latter alternative could be designed as a full-fledged deposit–refund system with a tax imposed on output in the amount of $D = R_G = R_S - V = E$ and a subsidy per unit returned in the amount of R_G.

4.1.1.2. Government-initiated systems. If no market-generated return system is in operation, but the disposal of used products gives rise to negative marginal external effects (E), which would be avoided if the used products were returned, a deposit–refund system of the type just mentioned could be introduced by the government. Assuming that the industry is competitive, $V \gtreqless 0$ would be the market price, equal to the firms' value, of a returned product, whereas the socially optimal payment (R_S) for a returned unit equals (at most) $E + V$. If so, a tax/subsidy in the amount of $D = R_G = R$ would be appropriate.

As consumers whose disposal cost difference $c^i_R - c^i_d$ exceeds the total payment, $R_S = R_G + V$ will continue to use the traditional disposal option, then, at the margin, social costs of traditional disposal equal social costs incurred by the return alternative, i.e. $E + V = c^i_R - c^i_d = R_S$. In the general case, the shift to this optimum will give rise to various distribution effects, where the losers will be (a) the producers, as producer price net of D is likely to go down (by $\Delta p < 0$) and (b) those consumers whose $c^i_R - c^i_d > R_S + \Delta p$. The winners include the remaining consumers, scavengers (who now may pick up discarded units for a refund), and taxpayers, to the extent that total deposits exceed total refunds.

So far we have discussed deposit–refund systems for *consumers* of products which may create environmental effects when disposed of (mercury and cadmium batteries, beverage containers, tires, junked cars, used "white goods", lubrication oil, freons in air conditioners and refrigerators, etc.). Similarly, deposit–refund systems may be designed for *producers* to check hazardous emissions of chemicals into the air and waterways or dumping of toxic wastes, in particular when proper treatment of such releases or wastes is expensive and improper disposal is easy to conceal. If the potential emissions or wastes are related to certain inputs in a straightforward fashion (such as potential sulphur emissions to the input of high-sulphur fuel oil), a tax/deposit could be levied on these inputs and a subsidy/refund paid for the quantity of chemicals (e.g. sulphur) or toxic material transferred to a specified type of processing firms. Here, as well as for other deposit–refund systems, a precondition for a well-functioning application is that there are sufficient safeguards against abuses, e.g. that ordinary sulphur cannot be bought and passed off as sulphur extracted from stack gases.

4.1.1.3. Performance bonds. Producer-oriented deposit–refund systems can be used to control other kinds of detrimental effects on the environment than those explicitly discussed so far [Solow (1971)]. First, restoration of production sites after shut-down may be required to avoid unwarranted permanent eyesores or accident risks (strip mining sites, junk yards, etc.) Second, we have the vast problem of safeguarding against *a priori* unknown environmental effects of new products, in particular new chemicals, or new production processes. When applying the principle of deposit–refund to such cases, the producer could be required to pay a deposit, determined by a court estimate of the likely maximum restoration costs or the maximum damages (in general or specific respects), to be refunded if certain conditions are met. In this way, society is protected against incomplete restoration because of intentional or unintentional bankruptcies. Moreover, the firm's operation will now be planned with respect to future restoration costs as well. And in the case of potential risks of innovation, this creates an alternative to awaiting the results of a test administered or supervised by the government. In this way, the introduction of the new products or processes would not be delayed. This alternative may be attractive to the innovating firm because the firm may have gathered information – and now definitely has an incentive to gather such information from the beginning of its R&D activities – implying that no harm is likely to result. Therefore, the firm may be willing to market the product or start using the new process with the specified financial responsibility, and both the firm and its customers may be better off [Bohm (1981, ch. 4)].

Although we take it for granted that the government will not trust a firm to meet its obligations without a financial commitment, in either of our two cases, it is conceivable that other parties which *are* trusted by the government would like to assume the financial responsibility involved. Thus, for example, the firm may convince a bank or an insurance company that the new product is safe. Or the firm may reveal its product secrets only to such a party but not to a public authority. If so, banks or insurance companies may assume the liability at a price. Thus, by using the risk-shifting mechanisms of the credit or insurance markets, the deposit–refund system can be transformed from a cash deposit version to a performance bond version, or firms could be allowed to choose either of these two versions. In other words, whenever the environmental effects potentially attributable to an individual decision-maker, and hence that individual's deposit, become sufficiently large and the transaction costs of the credit or insurance markets are no longer prohibitive, deposit–refund systems are likely to take the form of a performance bond system.

4.2. Liability

Another possibility for providing incentives to polluters or potential polluters is to make them liable for the actual damage they cause but without demanding a deposit or performance bond. To some extent, of course, they have been liable right along, at least in common-law countries; and remain liable even after the enactment of regulatory legislation aimed at pollution control. This liability arises under the common law of private and public nuisance and is enforceable through the courts; by damaged parties in the former case and, for the most part, by governments in the latter case [Boger (1975)]. The "natural" occurrence of this instrument and thus its apparent independence of government regulatory activity have made it attractive to those who favor minimal government interferences with the functioning of the market system.

The theoretical literature dealing with liability as a policy instrument for the most part descends from the important and challenging theorem of Coase on the irrelevance of property rights to efficiency outcomes in environmental conflicts [Coase (1960)]. This line of descent is hardly surprising, since the right to enjoy property free from external interference and the entitlement to liability for interferences that do occur are closely related though distinct possibilities for dealing with conflicts over the use of property generally [Calabresi and Melamed (1972), Bromley (1978)].

This literature is interested in the efficiency properties of these alternative principles and in their comparison with explicit government intervention of the classical Pigouvian sort (e.g. Brown (1973), Polinsky (1979)]. For the most part it confines itself to the case of small numbers of both polluters and damaged parties, though alternative assumptions about the availability of information and the behavior of those parties in bargaining (cooperative or not) are explored. In addition, the costs of enforcement through the courts and the problems of proof of damage for liability purposes are generally ignored. In these circumstances property and liability approaches have been shown to be roughly equivalent in efficiency properties and in terms of protection of the entitlement at issue and both have been shown superior to Pigouvian taxes when behavior is not cooperative [Polinsky (1979)].

Unfortunately, the restriction to small numbers, which frees one from the internal information and decision problems that would be faced by, say, a river basin's population if it tried to act collectively to stop a polluter of the river, and the ignoring of real costs of enforcement make these results of limited interest. Moreover, writers on liability seem, rather surprisingly, to have ignored the problem of incentives for damage-seeking behavior created by the liability payments to damaged parties. As we noted above in Section 2, in discussing the possibility of using emission-charge revenues to indemify pollutees, the conclusion of writers in that literature has been that such a policy would be incorrect. The largest difference between this conventional charge-payment idea and a legislated liability system would be that arising from uncertainty about whether damage would occur or not and whether if occurring they would be compensated. It would still seem to be the case that the more successful a liability system were in guaranteeing compensation, the stronger the incentives it would provide to potential damaged parties. Finally, the problems raised by the uncertainty itself have been disregarded in this literature's comparisons of liability rules and Pigouvian taxes. If discovery and ultimate proof of responsibility are uncertain, the polluter must face a potential payment adjusted to provide the socially correct signal, given that no payment at all may be required even if an incident occurs and damages result. [For a discussion of a related situation, see Shavell (1982).]

Somewhat more to the point is work such as that of Wittman (1977), focusing on the role of monitoring costs in choosing between prior regulation and expost liability for attacking public problems. This points the way to some interesting considerations relevant to choosing liability as a policy instrument. It also emphasizes the close relation between a system of expost liability and some of the deposit–refund arrangements just discussed.

Thus, a liability system, despite its drawbacks, may be a desirable way to approach problems for which information, in any of several senses, is scarce and expensive. For example, take a case in which the prospective damages of some contemplated action (introduction of a new drug or construction and use of a hazardous waste dump site) cannot be estimated even in a meaningful probabilistic sense. This might be true if the experts' prior subjective probability density function were uniform over a very wide range from zero to some catastrophic loss. This provides a weak basis for choosing a particular set of regulations (deciding on a drug ban or on a landfill design requirement) or for setting a fee (for wastes dumped at the site). In such circumstances a designation of strict liability could be appealing. The liability payment might be guaranteed by a performance bond or insurance policy as described in the preceding subsection. It would provide incentives for the active party to engage in information gathering and to take some actions aimed at prevention, at least those where the costs are small and the information or prevention effects are likely to be substantial.

If monitoring of actions to avoid causing damage (e.g. discharge reduction or spill prevention activities) is expensive or technically very difficult, but the sources of actual discharges or spills could be identified expost, a liability rule might usefully substitute for a regulatory rule. If monitoring the quantity of discharge, as opposed to the mere existence of some discharge, is expensive, or if the problem is with spills seen as stochastic discharges, so that fixed fees per unit discharge are difficult to apply, liability may again hold promise.

Notice that even in these rather special situations, the choice of a liability approach is not without serious disadvantages, however. Unless some special process of enforcement were set up, damaged parties would still: suffer real and possibly very serious damage; have to hire lawyers and go to court to claim their entitlements; and have to prove the connection between their damages and the act of the active party. The first of these three requirements must be seen as a political strike against the liability instrument. Policies that appear to prevent damages are surely easier to sell to an electorate than policies that depend for their working on a more or less ironclad guarantee that damage will be compensated by the polluter.

The third requirement, that a connection between action and damage be proved, also looms large as a potential difficulty. If the drug we originally hypothesized could only have one type of ill-effect or if the landfill were completely isolated from other sources of ground-water pollution, the position of the damaged party would be most clear cut. But, if the drug might produce long-delayed symptoms that could also be attributable to naturally occurring disease, or if the landfill site is surrounded by other industrial and commercial establishments (and perhaps even old landfills) proof of the cause–effect relation may be very difficult or even impossible. Special standards of proof (one or another version of a "rebuttable presumption" of causality) may be established to get around this obstacle in particular circumstances, but this course is circumscribed. If every case of X arising within T years among residents of area Y is by fiat to be attributed to our landfill, we must be quite sure that X arises only rarely without *any* obvious cause. Furthermore, we cannot thereafter similarly attribute X to another competing cause should we wish to use the liability instrument in other contexts near Y. This limitation has its most obvious meaning where some ubiguitous cause, such as sulfate air pollution of a metropolitan area is to be attacked by a compensation scheme amounting to the imposition at joint liability on the polluters of the region.

Where these difficulties of proof can be overcome, and where the political objection to a damage-accepting policy can also be overcome, liability as an instrument of policy does offer some dynamic advantages. It is self-adjusting in the face of exogenous change. For example, as technology changes, the polluters can adjust their actions to reflect the new balance between avoidance cost and expected damages. And it provides a continuing incentive to seek new technologies reducing expected damages.

An actual strict liability system, where damages are hard to estimate and preventive action hard to monitor, has been established in the United States Outer Continental shelf oil tract leasing program. Liability for damage from spills attaches to lease owners, and some information is available on the likelihood that a spill in a particular block would affect either fishing grounds or beaches. For a brief description and some preliminary evidence that the value of leases reflects an estimate of potential liability costs (or the costs of their avoidance) see Opaluch and Grigalunas (1983).

For a brief discussion of the problems of liability law in the context of damages from toxic substances, and for suggestions on moving away from that law toward "no-fault" victim compensation funds, see Trauberman (1983). It would appear that the desire to make compensation easy to obtain conflicts with the desire to impose incentives for improving disposal systems on individual generators of hazards. An attempt to make the two goals more compatible is the proposal to fund the U.S. Superfund (for the restoration of hazardous waste disposal sites and other compensation-type activities) from a tax on hazardous waste disposal rather than chemical feedstock use [AWPR (1983)].

5. Regulation

By regulation we mean essentially "a directive to *individual* decision-makers requiring them to set one or more output or input quantities at some specified levels or prohibiting them from exceeding (or falling short of) some specified levels" [Baumol and Oates (1975)]. As pointed out earlier, regulation has been the form of environmental policy preferred by politicians throughout the industrial world. We begin by presenting what appears to be the main arguments for this choice (Section 5.1). The different forms of regulation and their efficiency effects are then discussed (Sections 5.2 and 5.3). Finally, we analyze how the drawbacks of regulations in some applications can be mitigated or eliminated by certain modifications and, in particular, by introducing some complementary element of economic incentive systems (Section 5.4).

5.1. Why politicians prefer regulation

As we shall see in the next section, in some situations regulation emerges as an efficient instrument of environmental policy. However, efficiency aspects alone do not explain why governments in most countries have relied mainly on regulatory instruments in this field. It is hardly an easy undertaking to pin down what these other considerations have been. Different reasons seem to have been invoked in different nations and in different policy situations as well as at different points in time. In addition, important reasons may not have been explicitly invoked at all, implying that their identification becomes guesswork and possibly subject to tendentious interpretations. An attempt to identify the factors which influence the choice of policy instruments is nevertheless central to a discussion of environmental policy alternatives in the real world. This statement is partly due to the fact that not all – perhaps not even one – of these factors are irrelevant from the point of view of the complete set of policy goals and the policy constraints existing in a democratic political environment.

In passing, it should be pointed out that the dominance of direct command and control instruments can be observed not only in environmental but also in other policy areas such as occupational health and safety, consumer protection and transportation. It appears that taxes and charges have rarely been introduced as instruments to control specific activities; and even more rarely have they been designed to achieve a specified goal with respect to such activities. The principal long-term function of these "economic incentive" systems has been to withdraw purchasing power from consumers and firms in order to finance the activities of the public sector. (The economic incentive system of subsidies, on the other hand, seems to have been used as an intentional control instrument, although the transfer of purchasing power may have been an important complementary reason for such a policy.) But this principal function of taxes and charges only increases economists' doubts about why governments "avoid" the use of charges in environmental policy when these charges, in contrast to those instruments now in force would provide revenue to the government without, in principle, any deadweight loss or excess burden.

We now try to identify some of the main reasons why politicians have taken the regulatory approach to environmental policy; most of the reasons that originate in the technical characteristics of environmental problems are left to the next subsection.

(1) In many countries, economists play a minor part, if any, in the administrative groundwork of environmental policy. If the administrators have a background in science, technology or law, the economic aspects will not always be taken into account for obvious reasons. And especially when members of the legal profession dominate the higher echelons of the executive agencies, instruments of the law and of traditional law enforcement are more likely to have the upper hand.

(2) Still, economists have made their voices heard and have confronted politicians with efficiency arguments in favor of economic incentive instruments. One reason why the impact has been small seems to be that these arguments are "sophisticated" and rely on an understanding of the market mechanism and of

the "indirect" effects of prices. In contrast, bans and other forms of regulation are often geared precisely to the activity which is to be controlled. Even when politicians grasp the implications of the alternative policy solutions, they may feel that their constituents would not and that they have to settle for the policy which will receive broad support from the general public.

(3) Financial considerations can prompt the government to prefer regulation to economic incentive instruments. This is obvious for the case of subsidies, but it may concern the case of charges as well. The fact that the effects of effluent charges on ambient quality are uncertain means that government revenue from such charges is also uncertain. This is a drawback from the point of view of budget administration.

(4) Taking other specific policy goals into account can favor the regulatory alternative. Thus, charges will add to inflation, whereas regulation may not do so to the same extent.

(5) Charges can have clearcut distribution effects, which the government may be hesitant to accept. This is so, for example, when low-income groups are in a position where reductions in real income are judged to be unacceptable and when a charge system would hit the consumption of this group. The fact that the distribution effects of regulation are less conspicuous of course does not mean that they are unimportant or even that they are less objectionable than those of a corresponding system of charges.[41] As the time profile of the price effects of charges and regulation, respectively, may be quite different – say, higher prices in the short run with charges than with direct control measures, and vice versa in the long run – an adequate consideration of the distribution aspects becomes quite difficult. But from a political point of view, the short-term distribution effects may be judged to be the most important ones, and here regulation is likely to perform better.

(6) Moving into the sphere of environmental policy proper, it is important to note that, if successful, regulation of discharges or the production processes of polluters will, in general, result in a more certain effect on ambient quality than charges levied on pollutants. As we saw in Section 2, unless the cost function for the reduction of discharges is known, directly or after a trial-and-error process, the effect of a given effluent charge is uncertain. We return to this important aspect in what follows.

(7) Regulation, even when it is less direct that we just suggested, has the aura of being a "no-nonsense" instrument, adequate for the control of serious environmental problems. In contrast, charges have often been viewed as an imperfect obstacle to continued environmental degradation and even as a "license to pollute".

[41] See, for example, White (1982, pp. 88, 89), where he estimates that both costs and benefits of the regulation of automobile exhaust emissions in the United States have been regressive. See also Pearce (1983).

(8) It may have come as a surprise to those who hold the view that charges provide a "license to pollute" that the polluting firms and their trade associations seem to prefer regulation to charges. This, in itself, may be enough to make the government choose a regulatory approach. There are at least three reasons for polluters to take a position in favor of regulation:

(a) If charges were to be set at levels which would produce the same reduction of discharges as a regulation in the long run as well as in the short run, it is of course worse for the polluters if they, in addition, would have to pay fees.[42]

(b) As regulation in general can be said to be more uncompromising for the polluters than charges, government is more inclined to listen to the views of the polluters or their representatives before any action is taken. In this process, the polluters may expect to have some influence on the design and stringency of the regulation.[43]

(c) In certain countries, the legal process of introducing new regulations implies drawnout negotiations and provides ample opportunity for appeals. In this way, government intervention may be delayed for a considerable period of time to the benefit of the polluters.

5.2. Forms of regulation: Static efficiency and information

In what follows we take the polluter to be a producer. (This terminology is formally adequate even for a polluting household which obviously not only consumes but also produces effects on others.) The main reason for this choice is a practical one; more often than charges or subsidies, regulation has been and, on administrative grounds, must be aimed at firms.

If a set of effluent charges can be determined so that given ambient standards are met, it is obvious that the same result can be achieved by regulating individual sources of pollution, provided the necessary information is available.[44] Thus, if such charges would make producer A reduce his discharges by 90 percent next year and producer B by 1 percent (due to higher removal costs), effluent standards for the two sources could be so specified.[45] If it were known that the charges would lead to the introduction of a new abatement technology in firm A five years from now and in firm B two years from now, design standards for the two firms

[42] On this point, and the possibility that cost savings would more than make up for the added out of the pocket cost of charge payments, see the discussion of Spofford's study above.

[43] For a discussion of the influence of business on regulation in the United States see Quirk (1981). For a different view, see Linder and McBride (1984).

[44] For a simple presentation see, for example, Tresch (1981, pp. 164–168).

[45] However, the *optimal* volume of pollution will, in general, vary with the policy instrument used. See Harford and Ogura (1983).

could be so specified. What may differ between the two alternative policies are the costs of administration, monitoring and enforcement. Once we observe that the necessary information is not *freely* available, however, an even more important difference between the two policies is seen to be the information cost or the availability of the necessary information at any cost. If the necessary information is not attainable, the two alternatives are no longer comparable on a cost-effectiveness basis; policy benefits as well as compliance costs may differ as well. To complicate matters further, given the information constraint, these differences cannot be known in complete detail.

This sets the stage for evaluating the static efficiency of regulation. What are its benefits, compliance costs, information costs, and administrative, monitoring and enforcement costs? Space does not allow us to cover all these aspects, nor does the literature or at least our knowledge of it. Instead we observe the different principal forms of environmental regulation, essentially in the order of decreasing degrees of freedom for the regulated parties, commenting on that appears to be the characteristic differences in the dimensions just referred to.

5.2.1. Forcing the polluter and the pollutee to negotiate

This regulatory approach obviously requires that the parties involved be either few in number or organized in such a way that they emerge as only a few negotiating parties. The two-party case is probably the only one pertinent for this kind of mild regulation. At the one extreme, negotiations would develop similar to those within a merger and lead to an efficient solution. Such an outcome would imply that both parties have free access to relevant information about one another. This outcome is likely only for parties engaged in activities about which there is common knowledge. At the other extreme, information and bargaining strength are unevenly distributed between the parties so that the outcome may be far from a first-best optimum, say close to status quo but with significant negotiating costs being incurred.

Thus, legislation that forces a polluter and a pollutee to negotiate a settlement can be an efficient policy under certain conditions. These conditions would include, in addition to complete information about relevant costs on both sides, sufficiently small monitoring costs, small compliance costs for the polluter, and the threat of alternative measures if a settlement satisfactory to the pollutee is not reached. One important case where this kind of regulation is not likely to be an efficient policy should be mentioned. If the information as specified is far from complete, while the authorities can extract the necessary information at low costs, other solutions, such as a more interventionist form of regulation, may be preferable.

5.2.2. *Performance standards*

A form of regulation that provides the polluter with maximum freedom of compliance is the establishment of effluent standards for pollutants. Assuming that monitoring does not cause any significant problems and that information about compliance costs is available to the regulator at low costs, this kind of performance standard is likely to qualify as an efficient instrument. It should be noted, however, that the determination of *optimal* effluent standards requires at least as much information as the determination of optimal effluent charges [Mäler (1974b)].

Even when little is known about compliance costs, effluent standards may be more efficient than alternative instruments such as effluent charges.[46] One reason is that the costs of the trial-and-error process of adjusting charges to meet the given standard may be high (see Section 2). Another reason arises when, in a given air- or watershed, there are several polluters, whose discharges have different "transfer coefficients" (the α_i in Section 2.1). As the optimal charges must be source specific in this case, effluent standards would perform at least equally well. Temporary fluctuations in the assimilative capacity of the environment, giving rise to occasional environmental crises, would call for either "unrealistically" frequent changes in charge levels or more constant and occasionally too high charge levels. In such cases, a flexible effluent standard has been suggested as a feasible and more efficient solution [Baumol and Oates (1975, ch. 11), Baumol and Oates (1979, ch. 20) and Howe and Lee (1983)].

Another instance when performance standards can be an efficient instrument has been discussed above in the context of marketable permits (Section 3). In the simplest version of such a system, pollutants released from all members of a given set of sources are taken to have the same environmental impact. Although the initial distribution of pollution rights is specified according to source, the transferability of these rights makes the regulation area-specific instead of source-specific.

Turning to applications of performance standards where inefficiency is likely to result even in the short run, we should note at least the following three cases. First, we have the traditional showcase of inefficient standards, where different polluters with the same environmental effect per unit of pollutant discharged have different marginal removal costs at the individual standards assigned to them (e.g. a 50 percent reduction of discharges for all polluters). Here, a given reduction of pollution is achieved at a higher total costs than would be the case for a uniform charge per pollutant which would equalize marginal removal costs for all polluters.

[46] For a pathbreaking analysis of charges vs. standards in the presence of uncertainty about, *inter alia*, compliance costs, see Weitzman (1974). See also the survey article by Yohe (1977). And for a recent extension to more general functional forms and error structures, see Watson and Ridker (1984).

Second, effluent standards are often differentiated between old and new sources of pollution. For example, a producer who operates an existing plant is exempted from pollution control, whereas new or remodeled plants are subjected to emission limits. This application of performance standards provides an incentive to keep old plants for a longer period of time than would be the case, for example, under a system of effluent charges. Obviously, this is a special case of the problem discussed in the preceding paragraph, with effluent standards allocated to polluters regardless of marginal removal costs.

Finally, when monitoring the discharges of pollutants is costly, neither effluent charges nor effluent standards may be the optimal policy choice. The special difficulty of the monitoring problem should be elaborated at this point. This difficulty may be ascribed to five features and applies, as already observed, to charges and marketable permits as well as performance standards.

(1) All emissions are fugitive in the sense that once outside the source's stack or wastewater pipe they are lost to measurement. They leave no trail unless some human agency intervenes.[47] Thus, we cannot monitor at our leisure if we really wish to know what is and has been going on.

(2) Discharges very randomly because of random equipment breakdowns, shifts in product mix or input quality, and changes in production levels at the source. These variations, it must be stressed, are separate from any intention the discharger might have to cheat; even the best corporate citizen can suffer a breakdown of a precipitator in vastly increased emissions. This randomness has itself two implications. First, we cannot usefully think of emission standards as simple fixed numbers. The appropriate orders for a region must take into account source variations and the probability of ambient standard violations. In addition, the orders must recognize in one way or another that in adjusting to the order (or to an economic incentive) the source must balance probability of violation against cost of controlling or narrowing its range of variation.[48] Second, the rules for

[47] This statement must be qualified in two ways. Remote monitoring equipment makes it possible to measure concentrations of certain residuals in a stack plume, though these methods are neither simple nor precise. [See Williamson (1981)]. Somewhat more tenuous is the technique of using ambient quality levels and discharge composition to infer discharges, though it might in some cases provide a defensible check on self monitored data. See Courtney, Frank and Powell (1981) and Gordon, (1980). More generally, some residuals are disposed of in "packages" - for example, drums of hazardous pollutants.

[48] At its simplest this means that if the agency orders a source to hold its dischargers below D at all times, the source must actually aim at a target or mean discharge value far enough below D that random occurrences of excess emissions will be so infrequent as to be ignored. How far below D the target emission must be depends on the width of possible swings in discharge, the costs of control, and the penalties for detected violations. If the regulatory agency wants to see the source emit D on average, it must redefine a violation. For example, if it knew the distribution of actual discharges around the source's target, it might define a violation as any discharge greater than $D + K$. K would reflect how closely the source could control its emissions and would be matched to an appropriate penalty reflecting the costs of this control and the acceptable probability of really high emissions (greater than $D + K$).

identifying violations must be consistent with the statement of the discharge limitation orders [e.g. Beavis and Walker (1983)].

(3) Some pollutants are measured using "batch" or discrete sampling techniques.[49] This means that the choice of discharge limitation order and the source's optimal reaction to it should both be complicated by the choice of sampling regime (how often to sample and how many individual samples to draw at a time).[50]

(4) Monitoring instruments are inevitably imprecise – that is, they measure with some error. This further complicates the task of defining and finding real violations.

(5) All the above features of the monitoring problem take on a different cast when we drop the implicit assumption that sources *will* try to obey their discharge limitation orders. Cheating will be worthwhile if the probability of detection and the penalty for a detected violation do not together provide a strong enough incentive. Where intermittent agency monitoring visits are involved, we further have to reckon with legal problems of access to sample, whether (and how much) advance notice is required, and how hard it is for the source to adjust discharges up and down – to avoid being caught cheating. Given these monitoring problems, regulatory orders other than simple discharge limits may be preferable.

5.2.3. Regulating decision variables correlated with emissions

If certain inputs or outputs are perfectly correlated with the volume of pollutants discharged and less costly for the government to monitor, indirect control is more efficient than direct control. This may be true even when correlation is less than perfect, but the advantages of indirect regulation may be limited to the short term and may not even hold for the period during which the firm's basic production process remains unchanged. The correlation between emissions of pollutants and the variable monitored may be based on the inspection of a plant or a piece of equipment when new (see, for example, standards for noise and exhaust emissions from new vehicles) or when carefully maintained with respect to releases of pollutants. This performance may not be representative at later stages of operation or when it is no longer worthwhile for the firm to undertake maintenance. Thus, if the government is forced to rely on information provided by the polluters, the reduction in monitoring costs from making control indirect may be outweighed by the imperfections of such information.

[49] It appears that continuous sampling methods with automatic recording are being developed for more and more pollution types, so this difficulty may tend to disappear as time goes on [APCA (1981)].

[50] Sampling size and frequency, given the source's distribution of discharges and the characteristics of the tests performed, define the probabilities of missing the violations and of finding false violations [Vaughan and Russell (1983)].

5.2.4. Design standards

When direct as well as indirect monitoring of releases of pollutants is unreliable, expensive or technically infeasible, requirements that producers use a specific technology become an obvious candidate for optimal policy. Such a policy has been used in practice in a large number of cases. For example, it is often difficult to monitor the source of air or noise pollution. Measuring emissions of BOD in waste water has proved expensive. In such cases, producers can be required to use particular production processes or input qualities (e.g. low-sulphur fuel). Alternatively, they can be required to install a specific kind of abatement or purification process or be forced to reprocess certain kinds of wastes. Or they can be required to transfer certain wastes to publicly owned purification plants, without (as it often happens) being charged the full costs of waste treatment.[51] As a less specific kind of design standard (at the time of the regulatory decision), dischargers may be required to apply the "best practicable technology" (BPT) or "best available technology" (BAT) at some given future date.

Design standards can be efficient policy not only for reasons of low monitoring costs. They also provide a way to save information costs among polluters. When there is no doubt about the most efficient solution to meeting a certain performance standard, a design standard is the obvious policy choice [Crandall (1979)].

But, when there are doubts about the most efficient approach to meeting a performance standard, the requirement that a specific technology be used is likely to cause misallocation of resources. For all firms in an industry, a series of small adjustments of the existing production processes or simply reduced output may turn out to be less costly alternatives to the required production process or abatement technology. More often perhaps, different firms in an industry have different least-cost solutions to the reduction of discharges accomplished by a certain design standard [see, for example, OECD (1982a, 1982b)].

Many of the political aspects discussed in the preceding subsection may explain why politicians often prefer the design standard solution. Installation of purification equipment is the "natural" policy if you want wastes to contain a smaller volume of pollutants; moreover, it may appear as an effective instrument if you want to satisfy the environmentally conscious general public, etc. Above all, perhaps, design standards are believed to contribute to protection of the environment with a high degree of certainty. However, there is evidence that the security provided by design standards in environmental policy is false or exaggerated in a number of cases. Thus, as touched upon earlier, the amount of actual discharges for which the required process was designed may be exceeded dramatically [see,

[51] This may be seen as a combination of a design standard and a subsidy. It is a subsidy in the sense that all costs of the regulation are not borne by the regulated party. Combinations of this kind have been quite popular with policy-makers, involving either lump-sum subsidies or subsidizing a part (or percentage) of the costs incurred, e.g. a percentage of the installation costs for the equipment required.

for example, Mäler (1974b)]. And equipment which meets certain standards when leaving the producer may be tampered with by the user; although peripheral to the case of design standards, the difference between emission levels for new cars and actual in-use emissions is a good illustration [White (1982)]. There are also indications that stricter standards for new equipment are circumvented by increasingly frequent modification of the equipment when in use.[52]

In many cases where design standards have not proved effective in practice, the problem has not so much been the standards themselves as the way they are enforced or checked. Thus, inspection of plants or equipment when in use can improve the results of design standards. However, the advantages in terms of low administrative costs that this kind of regulation was credited with may be lost in the process.

5.2.5. Bans on products or processes

Outright bans may appear to be the strictest form of regulation. Banning the production (or use) of a product which has no close substitutes is a case that supports this view. But close substitutes are often available at low extra costs (as is illustrated, for example, by the appearance of other propellants for aerosol sprays when chlorofluorocarbons were banned in certain countries). And this may be true when bans are imposed on certain inputs, such as high-sulphur fuel in certain areas. Moreover, when bans take the form of zoning or curfews, compliance costs may be small, because alternatives remain open to the regulated party. This is so in particular when bans are announced well in advance. In this perspective, design standards rather than bans represent the most severe type of regulatory constraint.

It follows from what we just said that bans on products or processes may be an efficient policy instrument when there are close substitutes at low additional costs. Moreover, bans – and even more, design standards – may make economies of scale in the production of the substitutes (the required or nonbanned equipment) materialize faster than through the market mechanism by itself. In fact, nonconvexities in production may prevent the market mechanism from ever reaching a point which is less harmful for the environment, and, at the same time, less costly; in such a case, regulation may be the obvious way to eliminate, as it were, the two market failures.

A similar case of non-convexities appears when the pollution problem is only latent, but still the source of inefficient resource allocation. This is the case, for example, where an existing plant pollutes the environment so that certain other activities sensitive to the pollution have never been established in the vicinity, although the social surplus would be higher if they were than if the existing firm

[52] See Broder (1982, ch. 5) for the case of noise emissions from motorcycles.

were kept there.[53] Charges are not likely to work in this situation, especially not if they should reflect the value of the latent externalities; an arrangement along such lines might incite blackmail or at least create insurmountable information problems. A ban on pollution, e.g. in the form of zoning, is perhaps the obvious choice of policy in this "no-pollutee case", given that the optimum form of land use has been identified.

The traditional case for bans is, of course, when environmental standards call for the elimination of a certain kind of discharges, such as highly toxic substances. In addition, even though zero pollution from a particular type of activity is not called for, a ban may be chosen for administrative reasons, e.g. because it is immediately apparent when the ban has been broken.

5.2.6. Collective facilities – a digression

Government investments in facilities for environmental protection (sewers, waste treatment plants, walls for protection against motorway noise, etc) or government restoration activities (cleaning up, reforestation, reaeration of lakes, etc.) bear some resemblance to the regulatory solution and may be discussed at this point.

The analytical background for government protection and restoration activities can be briefly outlined as follows. If costs of protection/restoration fall short of the value of the corresponding environmental damages, there is a case for protection/restoration. Furthermore, if collective protection/restoration activities are less costly than environmental protection administered by the polluters individually, the collective alternative is favored. To implement this kind of policy, it may be sufficient for the government to ban certain kinds of discharges into the environment, provided that this ban actually institutes voluntary actions leading to the emergence of the optimal, collective arrangement. An illustrative example here could be the emergence of privately owned refuse collection activities as a consequence of such a ban.

Privately owned facilities of this kind may not materialize for reasons of administrative complexity or when the protection involved is a pure public good, instigating free-rider behavior among individual members of the common-interest group. Or organization costs may simply be believed to be too high, e.g. due to fears that several competing units may be established (at least temporarily) for a private-good kind of activity subjected to large economies of scale. Or a privately owned natural monopoly, once established, may charge monopoly prices. For

[53] For example, the existing firm A runs at a profit of $1 million per year. The "other activities", if firm A were absent, would run at an aggregate profit of $2 million per year. However, when A is present, they would not be able to make a profit due to pollution from A. Moreover, costs of organizing these other activities or lack of available funds bar the formation of an interest group which could buy firm A and shut it down. Or, there may be space for a new firm A′ to locate in the area once firm A is shut down. Hence, for several reasons, the market cannot make the optimum allocation materialize.

reasons such as these, the government may prefer to give the protection activities a public-utility status with the government control accompanying such a status; or the activities may be operated directly by the government. To implement such a choice, the government may want to complement the ban with, or have it replaced by, a design standard requiring the polluters to be connected to a central waste treatment plant.[54]

In other instances of government provision of collective facilities, no act of regulation may be involved. This is the case, for example, with most forms of restoration campaigns as well as with all improvements of existing waste treatment plants. To evaluate whether such activities are worthwhile it is only required that they meet the relevant cost–benefit criterion.

5.3. *Regulation and dynamic efficiency*

In the preceding section, our primary objective was to describe the principal forms of regulation and their static or short-term efficiency characteristics. In this section, we discuss regulatory instruments with respect to efficiency over time. Economists' evaluations of environmental regulation have to a large extent concentrated on this aspect. Here we discuss the following three issues: adaption to changes in exogenous variables, incentives to develop new forms of pollution-abatement technology, and effects on market structure and competition.

5.3.1. *Environmental regulation in the presence of exogenous changes*

Efficiency over time requires, in principle, that policy be adapted to exogenous changes in environmental costs as well as compliance or removal costs, subject to administrative and other specific costs associated with policy change. As mentioned above these costs of policy change may be lower for regulatory instruments than for economic incentive systems in the context of short-term fluctuations in the assimilative capacity of the environment. This might extend to the case of exogenous long-term fluctuations as well. In practice, however, regulation may not be administered with sufficient flexibility to take advantage of this potential. This is likely to be true at least for certain forms of regulation such as design standards, for which the regulatory process may be very slow.

If it turns out that regulation and economic incentive systems in fact tend to be equally inflexible over time, we may investigate the relative merits of the two policy approaches when confronted by exogenous changes. Assume a situation where a system of effluent charges and a system of effluent standards would be

[54] For a discussion of the choice between pollution charges leading to individually administered protection and forcing or simply allowing polluters to connect to centralized waste treatment activities, see Bohm (1972).

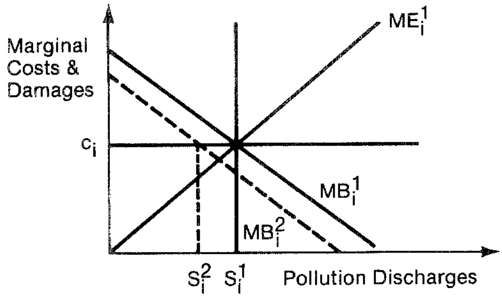

Figure 10.1. Charges, standards, and change in the marginal costs of reducing pollution discharges.

equally efficient and successful in guiding the economy to a short-term optimum position. See the standard S_i^1 for firm i and the uniform charge per unit of pollutant c_i in Figure 10.1. Emissions are brought to the point where the initial marginal benefit curve of the polluter (MB_i^1), also the reverse of the marginal compliance cost curve, intersects the curve for the marginal environmental effects (ME_i^1). Now, we should note first of all that neither the inflexible charge system (with unit charges independent of emission volume) nor the inflexible standard would allow any adjustments when the estimated marginal environmental effects (ME_i) change. On the other hand, when external factors influence the marginal compliance costs (MB_i), some adjustment will automatically take place in the charges case. (MB_i^2 produces discharges equal to S_i^2 when a charge is applied.) But as long as the charges do not change to perfectly reflect the marginal environmental effects, these adjustments may not be preferable to the absence of adjustments in the standard case. It is clearly seen that the outcome will depend on the extent to which both the marginal environmental effects and the marginal compliance costs change in the relevant interval. The charge system will tend to perform better than the standard if the marginal environmental effects are sufficiently close to being constant around the initial optimum point. Conversely, if these effects rise at a sufficiently high rate at this point, the inflexible standard will be the least imperfect instrument of the two [see Weitzman (1974) and Yohe (1977)].

5.3.2. Endogenous adjustments of compliance costs

When subjected to a given policy, the polluter has, in principle, a larger number of adjustment options open to him, the longer the adjustment period. Furthermore, if there are incentives for the polluter to develop new forms of adjustments – something which may be influenced by policy design – additional options may emerge over time. For these two reasons, compliance costs of a given policy will in general be lower in the long run than in the short run. This also means that compliance costs at the time the policy takes effect will be lower if advance notice of the policy is given [see Kneese and Schultze (1975, pp. 79–80)]. And this may be increasingly important the fewer the options allowed by the policy. For example, compliance with a design standard introduced on short notice may be very costly to the firm; say if the plant has just been remodeled. In contrast, an effluent standard – and even more an effluent charge – may allow the firm to make a much less costly temporary adjustment and introduce the technology implied by the design standard at a later stage, assuming this standard is the most efficient form of long-term adjustment.

Incentives to develop new options diminish the smaller the scope of adjustment allowed by the policy, *ceteris paribus*. Thus, with effluent charges, a maximum number of compliance alternatives are acceptable and hence, technological R&D may be pursued in any direction. At the other extreme, a design standard leaves no room for innovation. Or this is so at least if policy cannot easily be redesigned should new and superior ways to meet a given ambient standard happen to be developed. The important aspect from the incentive point of view is, of course, to what extent the firm believes it to be possible to influence policy by developing new and more efficient technology.

Moreover, once the polluter has adjusted to the new piece of regulation, there is no longer any incentive for him to attempt reaching a lower level of pollution than that implied by the regulation (be it a performance or a design standard), even when such a reduction would be valuable to society. Charges, on the other hand, provide such an incentive although its size may be nonoptimal (e.g. too large in the situation portrayed in footnote 22). Certain forms of regulation may even actually discourage the development or introduction of innovations. Thus, establishing shifting BAT standards for an industry creates perverse incentives for innovation.[55]

Although no real-world policy instrument can be expected to send correct signals to guide the long-term adjustment of pollution abatement and the development of new abatement technology, regulation and especially design standards are likely to perform much worse than economic incentive systems in these

[55] For example, in the U.S. Clean Water Act explicitly, and at least in the rhetoric surrounding the Clean Air Act, improvements in technology are supposed to trigger tightening of the standards [Clean Water Act, Section 302d in Government Institute (1980)]. This reduces the incentive to seek cost-reducing technical improvements in production process or treatment equipment, and under some circumstances may eliminate the incentive altogether. A very simple way of looking at this process uses the figure in footnote 22. When technology is improved, and marginal cost falls to MC_1, the ratchetting-down requirement implies a new lower discharge standard. Let us say that the rule for choosing this level is to maintain equal marginal costs (e_0) before and after. Then, after technical change, the standard would be D_1, and the net savings to the firm would be C-F. In this figure, area F will always be greater than area C, so there is a *disincentive* to innovate. More generally, the existence of the additional cost, F, will at least reduce the positive incentive to innovate.

respects. Thus, a policy that relies on regulatory intervention tends to make the long-term costs of attaining a given ambient quality unnecessarily high. This does not mean, however, that environmental regulation must lead to a reduction in productivity as commonly measured. In fact, there are some indications that increasingly stringent effluent standards have operated as a challenge to industry and spurred an innovation response whereby both pollution has been diminished and productivity has increased [OECD (1982a, 1982b)]. This is not to say, of course, that policy instruments, which allow a still larger freedom of adjustment and provide stronger incentives for developing new ways of reducing pollution, would not have performed even better.

5.3.3. The effects of regulation on market structure

If industry has an influence on the design of environmental regulation and the larger firms play a prominent role in this process, the result may be unfavorable for the smaller competitors in the industry. Moreover, the use of design standards requiring new production processes or the installation of expensive pollution-abatement technology may hit small firms particularly hard.[56]

If regulation tends to disfavor certain types of firms in an industry, the effect may be that competition is reduced [see Buchanan and Tullock (1975) and Dewees (1983)]. This effect may be particularly serious if mainly innovative firms (e.g. small growing firms) are hit hard by regulation. Moreover, if control is tighter for new firms, competition and innovation in the industry may be reduced still further [OECD (1982b)]. All this would contribute to maintaining a high level of direct as well as indirect compliance costs of regulation in the long run.

5.4. Modifying the performance of regulatory instruments

Some ways to improve the efficiency of the regulatory approach follow from our discussion in the preceding section. First of all, we saw that adding dynamic efficiency aspects to the static ones presented in Section 5.2 suggests that regulatory design be shifted towards forms which allow more freedom of adjustment. Second, advance notice of a given piece of regulation tends to reduce compliance costs. Third, design standards and other inflexible forms of regulation may be less costly to society if government shows a willingness to redesign its rulemaking when new solutions for protecting the environment emerge. In this way, the regulated party may be given an incentive to undertake R&D of new pollution-abatement technologies. In contrast, the use of BAT standards and a tendency to introduce stricter standards for industries that have developed less

[56] See Grabowski and Vernon (1978) for examples from the field of consumer product safety regulation.

harmful production processes are likely to impede innovation. Hence, compliance costs for a given ambient quality are increased or political ambitions with respect to ambient quality may have to be lowered.

In addition to modifications of the type suggested above, the regulatory system may be improved by introducing elements from economic incentive schemes. In this way, the high degree of certainty as to the effects of regulation, which – right or wrong – seems to be decisive for policy choice in the real world, can be obtained along with a stimulus towards efficiency that otherwise may be absent.

First, it should be noted that an economic incentive element is in fact already incorporated into most forms of regulation. If a polluter fails to comply with the directive given to him, he may be fined for doing so. A disadvantage of this regulatory design is, however, that the exact penalty level often is not known beforehand.

The problem of uncertain penalties would be eliminated if regulated parties were confronted with explicit, punitive *non-compliance fees* [see, for example, Viscusi (1979)]. That is, the polluter is formally allowed to exceed the standard given to him and will do so if his compliance costs are high.[57] Although regulation might seem less stringent as a consequence of such a system, it should be noted that this kind of legalized non-compliance allows standards to be set at a more demanding level than otherwise.

In practice, the application of noncompliance fees is often subject to severe imperfections. Thus, the fee is frequently calculated to equal the regulated party's gain from non-compliance; in other words, the fee is not punitive. Given that non-compliance is not always detected and that the regulated party's gain is likely to be underestimated by an outside party such as the government, this kind of policy can hardly be conceived of as rational. For example, it is difficult to see why the polluter would pay any attention to the standard imposed under these circumstances, unless, of course, there were additional and diffuse costs of stigmatization embedded in non-compliance.

As another form of incentive element, effluent charges could be levied on the polluter along with an effluent standard.[58] Assuming that the standard is binding when initially introduced, the effect of the charge would be to promote a future reduction in pollution below the level of the standard. This would increase long-run efficiency, provided, of course, that the value of further reductions in pollution were sufficiently high. Alternatively, reduction in pollution below the

[57] This idea, which in the United States originated as a practical policy in Connecticut and came to Washington with Douglas Costle, former Administrator of EPA, is now part of the Clean Air Act. (Section 120 of the Clean Air Act is devoted to a noncompliance penalty system.) See also, Drayton (1980). It allows EPA administratively to assess, on a source not complying with discharge regulations, a penalty equal to what the agency calculates the source would save through its noncompliance.

[58] For a version of an optimal mixed program of this kind see Baumol and Oates (1975, pp. 162–171).

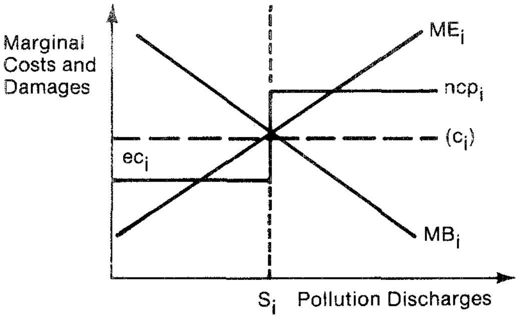

Figure 10.2. Combination of charge, standard, and non-compliance penalty.

level of the standard could be subsidized. The overall effects of a subsidy would differ from those of an equally large effluent charge, unless the income effect of the charge/subsidy could be disregarded and the shadow price of a dollar of government revenue equalled a dollar.

In principle, such a combination of standards and charges (subsidies) would provide the advantages of both systems, i.e. the relative certainty of a maximum limit to pollution and an incentive for the polluter to seek ways to reduce pollution still further in the short as well as in the long run. Furthermore, the standard could be combined with both (a subsidy or) an effluent charge (*ec*) and a noncompliance penalty (*ncp*), exceeding the level of the effluent charge.[59] Given sufficient information about the (nonconstant) marginal environmental effects of discharges and about the frequency distribution of the curve displaying the polluter's marginal benefits of discharges, such a system could be more efficient than a pure system of uniform effluent charges (see Figure 10.2). However, a prerequisite for such an outcome is that discharges from one polluter do not significantly alter the marginal environmental effects of discharges from another.

To sum up: although the actual use of environmental regulation appears to be based largely on factors unrelated to efficiency, there are as we have seen a number of instances in which efficiency aspects call for regulation instead of economic incentive schemes. But when emphasis is placed on long term efficiency and on the strength of the profit motive for seeking innovations in pollution abatement, economic incentives become more important. How much more im-

[59] See Roberts and Spence (1976). A fee-subsidy system was developed by James Smith and his colleagues at the City of Philadelphia Air Management Services and is reported in *Feasibility Study: A Fee/Subsidy System for Controlling Sulfur Dioxide Emissions*; a multiple-volume set of working papers by various authors [Philadelphia Air Management Services (1981)]. The aim here is zero net revenue raising (zero net transfer cost to polluters) and the mechanism is a combination of a specified emission level for each source, a fee for emissions over that level, and a subsidy for reductions in emissions below the chosen level.

portant is a matter of belief in the innovating capacity of the polluters and special firms developing pollution abatement equipment. However, combining standards and economic incentive schemes opens up a possibility to extract some of the best from both approaches. Such a combination can be obtained with a system of marketable permits, as discussed in Section 3. Or, a set of charges or subsidies and/or noncompliance penalties can be added to the set of standards. But for this to be meaningful, the standards must be of a type that allows some freedom of adjustment.

6. Moral suasion

As we saw in the preceding section, the choice of environmental policy instruments may be influenced by a number of "non-economic" factors. As a special but probably not unusual case, the policy-maker is confronted with a situation where there are definite constraints on the set of policy instruments. The origin of such constraints may be found in the political interpretations of public opinion. Thus, for example, it may become clear or interpreted as clear that charges on polluters are out-of-bounds, politically speaking, whereas subsidies are not or vice versa.

Estimates of compliance costs, employment effects, etc. made by interest groups often play a prominent role in the formation of such constraints. Typically, these estimates are based on insights that outsiders, and among them the government, cannot check. In particular, the effect of pollution charges on employment and the volume of exports of an industry may be greatly exaggerated by industry representatives without anyone else being able to prove that these estimates are biased and even less, of course, to ascertain the extent of which they are biased. [See Sonstelie and Portney (1983) for possible solutions to some of these problems.]

Thus, political constraints on environmental policy (to be distinguished from observing other goals of economic policy such as distribution goals) may be in force and turn the choice of optimal policy into a second best problem. In the limiting case, all stringent political actions to meet certain government policy goals may be blocked by such constraints. In that case, only actions that are voluntary on the part of the polluters are open to government influence. We now turn to a brief discussion of this "policy of moral suasion", which has occasionally been used and, in some cases, has proved to be effective.

Government initiatives to influence behavior on a voluntary basis can hardly be expected to be effective in all instances of environmental protection. If the environmental hazards are not conspicuous or dramatic enough, moral pressure may not materialize among any significant number of people. Similarly, when it is generally felt that the formal or moral responsibility rests with an identifiable party, others may not be easily convinced to take action. But many cases of environmental degradation are characterized by a lack of well-defined property rights and hence by unclear responsibility. In such cases, protection of the environment may be seen as a moral concern for people in general.

Attempts to influence the behavior of *individuals and households* can gain support from existing attitudes and social valuations related to the environmental issue involved. Thus, an attempt to make people abstain from buying fur coats to protect endangered species may receive firm backing from people who cannot afford to buy them. In other cases, where voluntary actions to protect the environment are conspicuous, such actions may be supported by feelings of cooperation and shared interests. This is probably more true for non-government initiatives than for the government-initiated attempts to influence behavior, which we are concerned with here. But this distinction may be less relevant for certain countries or local areas with a tradition of consensus on a large number of social issues. Thus, for example, attempts to make people voluntarily return used mercury and nickel-cadmium batteries to sellers have been fairly successful in some countries [see, for example, OECD (1981)]. A more general problem is that, unless new habits have had time to be formed, moral suasion may be effective only for a short period – as long as the arguments seem new and compelling.[60]

The likelihood of persuading *firms* to take voluntary action of reduce pollution (without the backing of a threat of harsher measures) is even smaller. Firms under the pressure of competition can be assumed to pay attention to arguments without a legal or economic content only when their costs of reducing pollution are negligible. Exceptions will be found when a conspicuous attempt to take moral arguments into account would serve the purpose of sales promotion, as when consumers have been building up a demand for new products with less negative effects on the environment (such as low phosphate detergents). But in such cases, unless the new product happens to be as effective, attractive and inexpensive as the original one, it is the consumers who pay the costs.

It should be noted in this connection that the relation between voluntary actions and constraints on policy may be the opposite of the one assumed here. Thus, firms may support voluntary programs among consumers or take voluntary actions on their own as an offensive measure to block the government from using more stringent and more effective policy instruments in the future.

So far we have discussed whether it is worthwhile for the government to undertake moral suasion when other instruments are blocked. But as pointed out by Baumol and Oates (1979), there are instances when such a policy is in fact more efficient than other instruments. First, this may occur when the monitoring required for economic incentive schemes or regulation is ruled out as being

[60] See, however, Baumol and Oates (1979, ch. 19) for examples where voluntary actions have remained in effect for longer periods of time.

technically infeasible or prohibitively expensive. For example, improper disposal of hazardous material into the sewage system is difficult to control by such methods, as is littering or careless use of open fire in wilderness areas. Here, moral suasion may be more efficient than realistic versions of other instruments. That this approach can also be quite effective is supported to some extent by experience from campaigns against littering in Scandinavia and "Smokey the Bear" forest-fire prevention campaigns in the United States.

Second, in certain cases of environmental catastrophies or immediate risks of such catastrophies (e.g. extremely hazardous smog levels), ordinary policy instruments may be too cumbersome or simply too slow. Again, there are examples which show that government appeals for voluntary actions can work fast and have an important impact in such voluntary situations.

To sum up, there are indications that, in certain cases, it may be worthwhile for the government to rely on moral suasion when alternative measures are blocked for political reasons. In addition, even when more sophisticated policy alternatives are available, there are cases when moral suasion emerges as an efficient policy instrument.

7. Concluding remarks

The message of this chapter may be seen as either negative or positive, depending on the perspective of the reader. The negative version is that no general statements can be made about the relative desirability of alternative policy instruments once we consider such practical complications as that location matters, that monitoring is costly, and that exogenous change occurs in technology, regional economies, and natural environmental systems. The positive way of stating this result is to stress that all the alternatives are promising in some situations. Even design standards have a place in the armamentarium of the environmental policy-maker. If the classic case for the absolute superiority of effluent charges is flawed by the simplicity of the necessary assumptions, the arguments for the superiority of rigid forms of regulations suffer equally from unstated assumptions and static views of the world. There is no substitute for careful analysis of the available alternatives in the specific policy context at issue.

That said, however, we are still tempted to stress the advantages of economic incentive systems in the long-run context, at least as a complement to a regulatory approach. The extra push toward the development of new production and discharge reduction technology provided by these instruments seems likely to dwarf in importance the short-run, and to some extent illusory, advantages to be gained by specifying actions or stigmatizing pollution at any non-zero level. Furthermore, we believe it worthwhile expanding the fields of application contemplated for such relatively unexplored instruments as deposit–refund systems.

Some exploration and experimentation can be done in real policy problems, but in many instances realistically complicated models will, we anticipate, provide insights currently lacking because of the simplicity of available theoretical models and the narrowness of actual experience.

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