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Environmental Economics: A Survey

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I. Introduction

WHEN THE ENVIRONMENTAL revolution arrived in the late 1960s, the economics profession was ready and waiting. Economists had what they saw as a coherent and compelling view of the nature of pollution with a straightforward set of policy implications. The problem of externalities and the associated market failure had long been a part of microeconomic theory and was embedded in a number of standard texts. Economists saw pollution as the consequence of an absence of prices for certain scarce environmental resources (such as clean air and water), and they prescribed the introduction of surrogate prices in the form of unit taxes or “effluent fees” to provide the needed signals to economize on the use of these resources. While much of

the analysis was of a fairly general character, there was at least some careful research underway exploring the application of economic solutions to certain pressing environmental problems (e.g., Allen Kneese and Blair Bower 1968).

The economist’s view had—to the dismay of the profession—little impact on the initial surge of legislation for the control of pollution. In fact, the cornerstones of federal environmental policy in the United States, the Amendments to the Clean Air Act in 1970 and to the Clean Water Act in 1972, *explicitly* prohibited the weighing of benefits against costs in the setting of environmental standards. The former directed the Environmental Protection Agency to set maximum limitations on pollutant concentrations in the atmosphere “to protect the public health”; the latter set as an objective the

“elimination of the discharge of *all* [our emphasis] pollutants into the navigable waters by 1985.”¹

The evolution of environmental policy, both in the U.S. and elsewhere, has inevitably brought economic issues to the fore; environmental regulation has necessarily involved costs—and the question of how far and how fast to push for pollution control in light of these costs has entered into the public debate. Under Executive Order 12291 issued in 1981, many proposed environmental measures have been subjected to a benefit-cost test. In addition, some more recent pieces of environmental legislation, notably the Toxic Substances Control Act (TSCA) and the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), call for weighing benefits against costs in the setting of standards. At the same time, economic incentives for the containment of waste discharges have crept into selected regulatory measures. In the United States, for example, the 1977 Amendments to the Clean Air Act introduced a provision for “emission offsets” that has evolved into the Emissions Trading Program under which sources are allowed to trade “rights” to emit air pollutants. And outside the United States, there have been some interesting uses of effluent fees for pollution control.

This is a most exciting time—and perhaps a critical juncture—in the evolution of economic incentives for environmental protection. The Bush Administration proposed, and the Congress has introduced, a measure for the trading of sulfur emissions for the control of acid rain un-

der the new 1990 Amendments to the Clean Air Act. More broadly, an innovative report from within the U.S. Congress sponsored by Senators Timothy Wirth and John Heinz, *Project 88: Harnessing Market Forces to Protect Our Environment* (Robert Stavins 1988) explores a lengthy list of potential applications of economic incentives for environmental management. Likewise, there is widespread, ongoing discussion in Europe of the role of economic measures for pollution control. Most recently in January of 1991, the Council of the Organization for Economic Cooperation and Development (OECD) has gone on record urging member countries to “make a greater and more consistent use of economic instruments” for environmental management. Of particular note is the emerging international concern with global environmental issues, especially with planetary warming; the enormous challenge and awesome costs of policies to address this issue have focused interest on proposals for “Green Taxes” and systems of tradable permits to contain global emissions of greenhouse gases. In short, this seems to be a time when there is a real opportunity for environmental economists to make some valuable contributions in the policy arena—if, as we shall argue, they are willing to move from “purist” solutions to a realistic consideration of the design and implementation of policy measures.

Our survey of environmental economics is structured with an eye toward its policy potential. The theoretical foundations for the field are found in the theory of externalities. And so we begin in Section II with a review of the theory of environmental regulation in which we explore recent theoretical results regarding the choice among the key policy instruments for the control of externalities: effluent fees, subsidies, and marketable emission permits. Section III takes us

¹ Although standards were to be set solely on the basis of health criteria, the 1970 Amendments to the Clean Air Act did include economic feasibility among its guidelines for setting source-specific standards. Roger Noll has suggested that the later 1977 Amendments were, in fact, more “anti-economic” than any that went before. See Matthew McCubbins, Roger Noll, and Barry Weingast (1989) for a careful analysis of this legislation.

from the theory of externalities to policy applications with a focus on the structuring and implementation of realistic measures for environmental management. This section reviews the work of environmental economists in trying to move from formal theorems to measures that address the variety of issues confronting an environmental regulator. We describe and evaluate briefly, as part of this treatment, the U.S. and European experiences with economic incentives for pollution control. In addition, we explore a series of regulatory issues—centralization versus decentralization of regulatory authority, international effects of domestic environmental policies, and enforcement—matters on which environmental economists have had something to say.

In Section IV, we turn to the measurement of the benefits and costs of environmental programs. This has been a particularly troublesome area for at least two reasons. First, many of the benefits and costs of these programs involve elements for which we do not have ready market measures: health benefits and aesthetic improvements. Second, policy makers, perhaps understandably, have proved reluctant to employ monetary measures of such things as “the value of human life” in the calculus of environmental policy. Environmental economists have, however, made some important strides in the valuation of “nonmarket” environmental services and have shown themselves able to introduce discussion of these measures in more effective ways in the policy arena.

In a survey in this *Journal* some fifteen years ago, Anthony Fisher and Frederick Peterson (1976) justifiably contended that techniques for measuring the benefits of pollution control are “to be taken with a grain of salt” (p. 24). There has been considerable progress on two distinct fronts since this earlier survey. First, environmental (and other) econo-

mists have shown considerable ingenuity in the development of techniques—known as indirect market methods—that exploit the relationships between environmental quality and various marketed goods. These methods allow us to infer the value of improved environmental amenities from the prices of the market goods to which they are, in various ways, related. Second, environmental economists have turned to an approach regarded historically with suspicion in our profession: the direct questioning of individuals about their valuation of environmental goods. Developing with considerable sophistication the so-called “contingent valuation” approach, they have been able to elicit apparently reliable answers to questions involving the valuation of an improved environment. In Section IV, we explore these various methods for the valuation of the benefits and costs of environmental programs and present some empirical findings.

In Section V, we try to pull together our treatment of measuring benefits and costs with a review of cases where benefit-cost analyses have actually been used in the setting of environmental standards. This provides an opportunity for an overall assessment of this experience and also for some thoughts on where such analyses are most needed. We conclude our survey in Section VI with some reflections on the state of environmental economics and its potential contribution to the formulation of public policy.

Before turning to substantive matters, we need to explain briefly how we have defined the boundaries for this survey. For this purpose, we have tried to distinguish between “environmental economics” and “natural resource economics.” The distinguishing characteristic of the latter field is its concern with the intertemporal allocation of renewable and nonrenewable resources. With its origins in the seminal paper by Harold Hotelling

(1931), the theory of natural resource economics typically applies dynamic control methods of analysis to problems of intertemporal resource usage. This has led to a vast literature on such topics as the management of fisheries, forests, minerals, energy resources, the extinction of species, and the irreversibility of development over time. This body of work is excluded from our survey. The precise dividing line between environmental economics and natural resource economics is admittedly a little fuzzy, but in order to keep our task a manageable one, we have restricted our survey to what we see as the two major issues in environmental economics: the regulation of polluting activities and the valuation of environmental amenities.

II. *The Normative Theory of Environmental Regulation*

The source of the basic economic principles of environmental policy is to be found in the theory of externalities. The literature on this subject is enormous; it encompasses hundreds of books and papers. An attempt to provide a comprehensive and detailed description of the literature on externalities theory reaches beyond the scope of this survey. Instead, we shall attempt in this section to sketch an outline of what we see as the central results from this literature, with an emphasis on their implications for the design of environmental policy. We shall not address a number of formal matters (e.g., problems of existence) that, although important in their own right, have little to say about the structure of policy measures for protection of the environment.

A. *The Basic Theory of Environmental Policy*²

The standard approach in the environmental economics literature charac-

terizes pollution as a public "bad" that results from "waste discharges" associated with the production of private goods. The basic relationships can be expressed in abbreviated form as:

$$U = U(X, Q) \quad (1)$$

$$X = X(L, E, Q) \quad (2)$$

$$Q = Q(E) \quad (3)$$

where the assumed signs of the partial derivatives are $U_X > 0$, $U_Q < 0$, $X_L > 0$, $X_E > 0$, $X_Q < 0$, and $Q_E > 0$. The utility of a representative consumer in equation (1) depends upon a vector of goods consumed (X) and upon the level of pollution (Q). Pollution results from waste emissions (E) in the production of X , as indicated in (2). Note that the production function in (2) is taken to include as inputs a vector of conventional inputs (L), like labor and capital, the quantity of waste discharges (E), and the level of pollution (Q). In this formulation, waste emissions are treated simply as another factor of production; this seems reasonable since attempts, for example, to cut back on waste discharges will involve the diversion of other inputs to abatement activities—thereby reducing the availability of these other inputs for the production of goods. Reductions in E , in short, result in reduced output. Moreover, given the reasonable assumption of rising marginal abatement costs, it makes sense to assume the usual curvature properties so that we can legitimately draw isoquants in L and E space and treat them in the usual way.

example, William Baumol (1972), Baumol and Wallace Oates (1988), Paul Burrows (1979), and Richard Cornes and Todd Sandler (1986). We have not included in this survey a literature on conservation and development that has considered issues of irreversibility in the time of development for which the seminal papers are John Krutilla (1967), and Kenneth Arrow and Anthony Fisher (1974). This literature is treated in the Anthony Fisher and Peterson survey (1976) and, more recently, in Anthony Fisher (1981, ch. 5).

² For comprehensive and rigorous treatments of the general ideas presented in this section, see, for

The production function also includes as an argument the level of pollution (Q), since pollution may have detrimental effects on production (such as soiling the output of the proverbial laundry or reducing agricultural output) as well as producing disutility to consumers. The level of pollution is itself some function of the vector of emissions (E) of all the producing units. In the very simplest case, Q might be taken to equal the sum of the emissions over all producers.³

One extension of the model involves the explicit introduction of "defensive" activities on the part of "victims." We might, for example, amend the utility function:

$$U = U[X, F(L, Q)] \quad (4)$$

to indicate that individuals can employ a vector of inputs (L) to lessen, in some sense, their exposure to pollution. The level of pollution to which the individual is actually exposed (F) would then depend upon the extent of pollution (Q) and upon the employment of inputs in defensive activities (L). We could obviously introduce such defensive activities for producers as well. We thus have a set of equations which, with appropriate subscripts, would describe the behavior of the many individual households and firms that comprise the system.

It is a straightforward exercise to maximize the utility of our representative individual (or group of individuals) subject to (2) and (3) as constraints along with a further constraint on resource availabil-

ity. This exercise produces a set of first-order conditions for a Pareto-efficient outcome; of interest here is the condition taking the form:

$$\frac{\partial X}{\partial E} = - \left[\sum \left(\frac{\partial U}{\partial Q} \frac{\partial Q}{\partial E} \right) / \frac{\partial U}{\partial X} + \sum \left(\frac{\partial X}{\partial Q} \frac{\partial Q}{\partial E} \right) \right] \quad (5)$$

Equation (5) indicates that polluting firms should extend their waste discharges to the point at which the marginal product of these emissions equals the sum of the marginal damages that they impose on consumers [the first summation in (5)] and on producers [the second summation in (5)]. Or, put slightly differently, (5) says that pollution-control measures should be pursued by each polluting agent to the point at which the marginal benefits from reduced pollution (summed over all individuals and all firms) equal marginal abatement cost.

Another of the resulting first-order conditions relates to the efficient level of defensive activities:

$$\frac{\partial U}{\partial F} \frac{\partial F}{\partial L} = \frac{\partial U}{\partial X} \frac{\partial X}{\partial L} \quad (6)$$

which says simply that the marginal value of each input should be equated in its use in production and defensive activities.

The next step is to derive the first-order conditions characterizing a competitive market equilibrium, where we find that competitive firms with free access to environmental resources will continue to engage in polluting activities until the marginal return is zero, that is, until $\partial X/\partial E = 0$. We thus obtain the familiar result that because of their disregard for the external costs that they impose on others, polluting agents will engage in socially excessive levels of polluting activities.

The policy implication of this result is

³ This highly simplified model, although useful for our analytical purposes, admittedly fails to encompass the complexity of the natural environment. There is an important literature in environmental economics that develops the "materials-balance" approach to environmental analysis (see Kneese, Robert Ayres, and Ralph d'Arge 1970; Karl-Göran Mäler 1974, 1985). This approach introduces explicitly the flows of environmental resources and the physical laws to which they are subject. Some of these matters will figure in the discussion that follows.

clear. Polluting agents need to be confronted with a "price" equal to the marginal external cost of their polluting activities to induce them to internalize at the margin the full social costs of their pursuits. Such a price incentive can take the form of the familiar "Pigouvian tax," a levy on the polluting agent equal to marginal social damage. In the preceding formulation, the tax would be set equal to the expression in equation (5). Note further that the unit tax (or "effluent fee") must be attached *directly* to the polluting activity, not to some related output or input. Assuming some substitution among inputs in production, the Pigouvian tax would take the form of a levy per unit of waste emissions into the environment—not a tax on units of the firm's output or an input (e.g., fossil fuel associated with pollution).⁴

The derivation of the first-order conditions characterizing utility-maximizing behavior by individuals yields a second result of interest. Inasmuch as defensive activities in the model provide only private benefits, we find that individual maximizing behavior will satisfy the first-order conditions for Pareto efficiency for such activities. Since they are confronted with a given price for each input, individuals will allocate their spending so that a marginal dollar yields the same increment to utility whether it is spent on consumption goods or defensive activities. There is no need for any extra inducement to achieve efficient levels of defensive activities.

Although this is quite straightforward, there are a couple of matters requiring further comment. First, the Pigouvian solution to the problem of externalities has been the subject of repeated attack along Coasian lines. The Ronald Coase

(1960) argument is that in the absence of transactions costs and strategic behavior, the distortions associated with externalities will be resolved through voluntary bargains struck among the interested parties. No further inducements (such as a Pigouvian tax) are needed in this setting to achieve an efficient outcome. In fact, as Ralph Turvey (1963) showed, the introduction of a Pigouvian tax in a Coasian setting will itself be the source of distortions. Our sense, however, is that the Coasian criticism is of limited relevance to most of the major pollution problems. Since most cases of air and water pollution, for example, involve a large number of polluting agents and/or victims, the likelihood of a negotiated resolution of the problem is small—transactions costs are simply too large to permit a Coasian resolution of most major environmental problems. It thus seems to us that a Nash or "independent adjustment" equilibrium is, for most environmental issues, the appropriate analytical framework. In this setting, the Pigouvian cure for the externality malady is a valid one.⁵

Second, there has been no mention of any compensation to the victims of externalities. This is an important point—and a source of some confusion in the literature—for Coase and others have suggested that in certain circumstances compensation of victims for damages by polluting agents is necessary for an efficient outcome. As the mathematics makes clear, this is not the case for our model above. In fact, the result is even stronger: compensation of victims is not permissible (except through lump-sum transfers). Where victims have the opportunity to engage in defensive (or "averting") activities to mitigate the effects of the pollution from which they

⁴ Where it is not feasible to monitor emissions directly, the alternative may be to tax an input or output that is closely related to emissions of the pollutant. This gives rise to a standard sort of second-best problem in taxation.

⁵ For comparative analyses of the bargaining and tax approaches to the control of externalities, see Daniel Bromley (1986), and Jonathan Hamilton, Eytan Sheshinski, and Steven Slutsky (1989).

suffer, compensation cannot be allowed. For if victims are compensated for the damages they suffer, they will no longer have the incentive to undertake efficient levels of defensive measures (e.g., to locate away from polluting factories or employ various sorts of cleansing devices). As is clear in the preceding formulation, the benefits from defensive activities are private in nature (they accrue solely to the victim that undertakes them) and, as a result, economic efficiency requires no incentives other than the benefits they confer on the victim.⁶

The basic theoretical result then (subject to some qualifications to be discussed later) is that the efficient resolution of environmental externalities calls for polluting agents to face a cost at the margin for their polluting activities equal to the value of the damages they produce and for victims to select their own levels of defensive activities with no compensation from polluters. We consider next some policy alternatives for achieving this result.

B. *The Choice Among Policy Instruments*⁷

The analysis in the preceding section has run in terms of a unit tax on polluting

activities. There are, however, other approaches to establishing the proper economic incentives for abatement activities. Two alternative policy instruments have received extensive attention in the literature: unit subsidies and marketable emission permits.

It was recognized early on that a subsidy per unit of emissions reduction could establish the same incentive for abatement activity as a tax of the same magnitude per unit of waste discharges: a subsidy of 10 cents per pound of sulfur emissions reductions creates the same opportunity cost for sulfur emissions as a tax of 10 cents per unit of sulfur discharges. From this perspective, the two policy instruments are equivalent: the regulator can use either the stick or the carrot to create the desired incentive for abatement efforts.

It soon became apparent that there are some important asymmetries between these two policy instruments (e.g., Morton Kamien, Nancy L. Schwartz, and F. Trener Dolbear 1966; D. Bramhall and Edwin Mills 1966; Kneese and Bower 1968). In particular, they have quite different implications for the profitability of production in a polluting industry: subsidies increase profits, while taxes decrease them. The policy instruments thus have quite different implications for the long-run, entry-exit decisions of firms. The subsidy approach will shift the industry supply curve to the right and result in a larger number of firms and higher industry output, while the Pigouvian tax will shift the supply curve to the left with a consequent contraction in the size of the industry. It is even conceivable that the subsidy approach could result in an increase in the total amount of pollution (Baumol and Oates 1988, ch. 14; Stuart Mestelman 1982; Robert Kohn 1985).

The basic point is that there is a further condition, an entry-exit condition, that

⁶ There may, of course, exist cases where defensive activities have "publicness" properties—where the actions of one victim to defend himself against pollution also provide defense for others. In such cases, there is clearly an externality present so that individual maximizing behavior will not yield the efficient levels of defensive activities. For a careful and thorough examination of defensive activities, see Richard Butler and Michael Maher (1986). Incidentally, the general issue of compensation of victims from pollution obviously has much in common with the moral hazard problem in insurance.

⁷ A further policy instrument not discussed in this section but with some potentially useful applications in environmental policy is deposit-refund systems (Peter Bohm 1981). Such systems can shift some of the responsibility for monitoring and effectively place the burden of proof on the source. For under this approach, the source, to recoup its deposit, must demonstrate that its activities have not damaged the environment. See Robert Costanza and Charles Perrings (1990) for a policy proposal under this rubric.

long-run equilibrium must satisfy for an efficient outcome (William Schulze and d'Arge 1974; Robert Collinge and Oates 1982; Daniel Spulber 1985). To obtain the correct number of firms in the long run, it is essential that firms pay not only the cost of the marginal damages of their emissions, but also the total cost arising from their waste emissions. Only if firms bear the total cost of their emissions will the prospective profitability of the enterprise reflect the true social net benefit of entry and exit into the industry.⁸ In sum, unit subsidies are not a fully satisfactory alternative to Pigouvian taxes (Donald Dewees and W. A. Sims 1976).

In contrast, in a world of perfect knowledge, marketable emission permits are, in principle, a fully equivalent alternative to unit taxes. Instead of setting the proper Pigouvian tax and obtaining the efficient quantity of waste discharges as a result, the environmental authority could issue emission permits equal in the aggregate to the efficient quantity and allow firms to bid for them. It is not hard to show that the market-clearing price will produce an outcome that satisfies the first-order conditions both for efficiency in pollution abatement activities in the short run and for entry-exit decisions in the long run. The regulator can, in short,

⁸ In an intriguing qualification to this argument, Martin Bailey (1982) has shown that not only subsidies to polluters, but also compensation to victims, will result in no distortions in resource use where benefits and damages are capitalized into site rents. For a discussion of the Bailey argument, see Baumol and Oates (1988, pp. 230–34). In another interesting extension, Gene Mummy (1980) shows that a combined charges-subsidy scheme can be fully efficient. Under this approach, sources pay a unit tax for emissions above some specified baseline, but receive a unit subsidy for emissions reductions below the baseline. The key provision is that the right to subsidy payments is limited to existing firms (i.e., new sources have a baseline of zero) and that this right can either be sold or be exercised even if the firm chooses to exit the industry. For a useful development of Mummy's insight, see John Pezzey (1990).

set either "price" or "quantity" and achieve the desired result.⁹

This symmetry between the price and quantity approaches is, however, critically dependent upon the assumption of perfect knowledge. In a setting of imperfect information concerning the marginal benefit and cost functions, the outcomes under the two approaches can differ in important ways.

C. *Environment Policy Under Uncertainty*

In a seminal paper, Martin Weitzman (1974) explored this asymmetry between price and quantity instruments and produced a theorem with important policy implications. The theorem establishes the conditions under which the expected welfare gain under a unit tax exceeds, is equal to, or falls short of that under a system of marketable permits (quotas). In short, the theorem states that in the presence of uncertainty concerning the costs of pollution control, the preferred policy instrument depends on the *relative* steepness of the marginal benefit and cost curves.¹⁰

⁹ The discussion glosses over some quite troublesome matters of implementation. For example, the effects of the emissions of a particular pollutant on ambient air or water quality will often depend importantly on the location of the source. In such cases, the optimal fee must be tailored to the damages per unit of emissions source-by-source. Or, alternatively, in a market for emission permits, the rate at which permits are traded among any two sources will vary with the effects of their respective emissions. In such a setting, programs that treat all sources uniformly can forego significant efficiency gains (Eugene Seskin, Robert Anderson, and Robert Reid 1983; Charles Kolstad 1987). More on all this shortly.

¹⁰ This result assumes linearity of the marginal benefit and cost functions over the relevant range and that the error term enters each function additively. Uncertainty in the benefits function, interestingly, is not enough in its own right to introduce any asymmetries; while it is the source of some expected welfare loss relative to the case of perfect information, there is no difference in this loss as between the two policy instruments. For useful diagrammatic treatments of the Weitzman analysis, see Zvi Adar and James Griffin (1976), Gideon Fishelson (1976), and Baumol and Oates (1988, ch. 5).

The intuition of the Weitzman proposition is straightforward. Consider, for example, the case where the marginal benefits curve is quite steep but marginal control costs are fairly constant over the relevant range. This could reflect some kind of environmental threshold effect where, if pollutant concentrations rise only slightly over some range, dire environmental consequences follow. In such a setting, it is clearly important that the environmental authority have a close control over the quantity of emissions. If, instead, a price instrument were employed and the authority were to underestimate the true costs of pollution control, emissions might exceed the critical range with a resulting environmental disaster. In such a case, the Weitzman theorem tells us, quite sensibly, that the regulator should choose the quantity instrument (because the marginal benefits curve has a greater absolute slope than the marginal cost curve).

Suppose, next, that it is the marginal abatement cost curve that is steep and that the marginal benefits from pollution control are relatively constant over the relevant range. The danger here is that because of imperfect information, the regulatory agency might, for example, select an overly stringent standard, thereby imposing large, excessive costs on polluters and society. Under these circumstances, the expected welfare gain is larger under the price instrument. Polluters will not get stuck with inordinately high control costs, since they always have the option of paying the unit tax on emissions rather than reducing their discharges further.

The Weitzman theorem thus suggests the conditions under which each of these two policy instruments is to be preferred to the other. Not surprisingly, an even better expected outcome can be obtained by using price and quantity instruments in tandem. As Marc Roberts and Michael

Spence (1976) have shown, the regulator can set the quantity of permits at the level that equates expected marginal benefits and costs and then offer a subsidy for emissions reductions in excess of those required by the permits and also a unit tax to provide a kind of "escape hatch" in case control costs turn out to be significantly higher than anticipated. In this way, a combination of price and quantity instruments can, in a setting of imperfect information, provide a larger expected welfare gain than an approach relying on either policy instrument alone (see also Weitzman 1978).¹¹

D. *Market Imperfections*

The efficiency properties of the policy measures we have discussed depend for their validity upon a perfectly competitive equilibrium. This is a suspect assumption, particularly since many of the major polluters in the real world are large firms in heavily concentrated industries: oil refineries, chemical companies, and auto manufacturers. This raises the issue of the robustness of the results to the presence of large firms that are not price takers in their output markets.

James Buchanan (1969) called attention to this issue by showing that the imposition of a Pigouvian tax on a monopolist could conceivably reduce (rather than raise) social welfare. A monopolist restricts output below socially optimal levels, and a tax on waste emissions will lead to yet further contractions in output. The net effect is unclear. The welfare gains from reduced pollution must be offset against the losses from the reduced output of the monopolist.

The first-best response to this conun-

¹¹ Butler and Maher (1982) show that in a setting of economic growth, the shifts in the marginal damage and marginal control cost schedules are likely to be such as to increase substantially the welfare loss from a fixed fee system relative to that from a system of marketable permits.

drum is clear. The regulatory authority should introduce two policy measures: a Pigouvian tax on waste emissions plus a unit subsidy to output equal to the difference between marginal cost and marginal revenue at the socially optimal level of output. Since there are two distortions, two policy instruments are required for a full resolution of the problem. Environmental regulators, however, are unlikely to have the authority (or inclination) to subsidize the output of monopolists. In the absence of such subsidies, the agency might seek to determine the second-best tax on effluents. Dwight Lee (1975) and Andy Barnett (1980) have provided the solution to this problem by deriving formally the rule for the second-best tax on waste emissions. The rule calls for a unit tax on emissions that is somewhat less than the unit tax on a perfectly competitive polluter (to account for the output effect of the tax):

$$t^* = t_c - \left| (P - MC) \frac{dX}{dE} \right| \quad (7)$$

Equation (7) indicates that the second-best tax per unit of waste emissions (t^*) equals the Pigouvian tax on a perfectly competitive firm (t_c) minus the welfare loss from the reduced output of the monopolist expressed as the difference between the value of a marginal unit of output and its cost times the reduction in output associated with a unit decrease in waste emissions. It can be shown by the appropriate manipulation of (7) that the second-best tax on the monopolist varies directly with the price elasticity of demand. The rationale is clear: where demand is more price elastic, the price distortion (i.e., the divergence between price and marginal cost) tends to be smaller so that the tax on effluent need not be reduced by so much as where demand is more price inelastic.

It seems unlikely, however, that the

regulator will have either the information needed or the authority to determine and impose a set of taxes on waste emissions that is differentiated by the degree of monopoly power. Suppose that the environmental authority is constrained to levying a uniform tax on waste discharges and suppose that it determines this tax in a Pigouvian manner by setting it equal to marginal social damages from pollution, completely ignoring the issue of market imperfections. How badly are things likely to go wrong? Oates and Diana Strassmann (1984) have explored this question and, using some representative values for various parameters, conclude that the complications from monopoly and other noncompetitive elements are likely to be small in magnitude; the losses from reduced output will typically be "swamped" by the allocative gains from reduced pollution. They suggest that, based on their estimates, it is not unreasonable simply to ignore the matter of incremental output distortions from effluent fees.¹² Their analysis suggests further that the failure of polluting agents to minimize costs because of more complex objective functions (a la Williamson), public agencies of the Niskanen sort, or because of regulatory constraints on profits need not seriously undermine the case for pricing incentives for pollution control. This subject needs further study, especially since many of the principal participants in the permit market for trading sulfur allowances under the new Amendments to the Clean Air Act will be regulated firms.

E. *On the Robustness of the Pigouvian Prescription: Some Further Matters*

Although the literature has established certain basic properties of the Pi-

¹² For more on this issue, see Peter Asch and Joseph Seneca (1976), Walter Misiolek (1980), and Burrows (1981).

gouvian solution to the problem of externalities, there are some remaining troublesome matters. One concerns the information requirements needed to implement the approach. Developing reliable measures of the benefits and costs of environmental amenities is, as we shall see shortly, a difficult undertaking. To determine the appropriate Pigouvian levy, moreover, we not only need measures of existing damages and control costs, but we need to develop measures of the incremental costs and benefits over a substantial range. For the proper Pigouvian levy is not a tax equal to marginal social damages at the *existing* level of pollution; it is a tax equal to marginal damages at the *optimal outcome*. We must effectively solve for the optimal level of pollution to determine the level of the tax. As an alternative, we might set the tax equal to the existing level of damages and then adjust it as levels of pollution change in the expectation that such an iterative procedure will lead us to the socially optimal outcome. But even this is not guaranteed (Baumol and Oates 1988, ch. 7).

There is, moreover, a closely related problem. In the discussion thus far, we have examined solely the first-order conditions for efficient outcomes; we have not raised the issue of satisfying any second-order conditions. As Baumol and David Bradford (1972) have shown, this is a particularly dangerous omission in the presence of externalities.¹³ In fact, they demonstrate that if a detrimental externality is of sufficient strength, it *must* result in a breakdown of the convexity-concavity conditions required for an optimal outcome. As a result, there may easily exist a multiplicity of local maxima from which to choose—with no simple rule to determine the first-best out-

come.¹⁴ Under such circumstances, equilibrium prices may tell us nothing about the efficiency of current output or the direction in which to seek improvement.

There are thus reasons for some real reservations concerning the direct application of the Pigouvian analysis to the formulation of environmental policy. It is to this issue that we turn next.

III. *The Design and Implementation of Environmental Policy*

A. *Introduction: From Theory to Policy*

Problems of measurement and the breakdown of second-order conditions (among other things) constitute formidable obstacles to the determination of a truly first-best environmental policy. In response to these obstacles, the literature has explored some second-best approaches to policy design that have appealing properties. Moreover, they try to be more consistent with the procedures and spirit of decision making in the policy arena.

Under these approaches, the determination of environmental policy is taken to be a two-step process: first, standards or targets for environmental quality are set, and, second, a regulatory system is designed and put in place to achieve these standards. This is often the way environmental decision making proceeds. Under the Clean Air Act, for example, the first task of the EPA was to set standards in the form of maximum

¹³ See also Richard Portes (1970), David Starrett (1972), J. R. Gould (1977), and Burrows (1986).

¹⁴ This problem is further compounded by the presence of defensive activities among victims of pollution. The interaction among abatement measures by polluters and defensive activities by victims can be a further source of nonconvexities (Hirofumi Shibata and Steven Winrich 1983; Oates 1983). Yet another source of nonconvexities can be found in the structure of subsidy programs that offer payments for emissions reductions to firms in excess of some minimum size (Raymond Palmquist 1990).

permissible concentrations of the major air pollutants. The next step was to design a regulatory plan to attain these standards for air quality.

In such a setting, systems of economic incentives can come into play in the second stage as effective regulatory instruments for the achievement of the predetermined environmental standards. Baumol and Oates (1971) have described such a system employing effluent fees as the "charges and standards" approach. But marketable permit systems can also function in this setting—a so-called "permits and standards" approach (Baumol and Oates 1988, ch. 12)¹⁵

The chief appeal of economic incentives as the regulatory device for achieving environmental standards is the large potential cost-savings that they promise. There is now an extensive body of empirical studies that estimate the cost of achieving standards for environmental quality under existing command-and-control (CAC) regulatory programs (e.g., Scott Atkinson and Donald Lewis 1974; Seskin, Anderson, and Reid 1983; Alan Krupnick 1983; Adele Palmer et al. 1980; Albert McGartland 1984). These are typically programs under which the environmental authority prescribes (often in great detail) the treatment procedures that are to be adopted by each source. The studies compare costs under CAC programs with those under a more cost effective system of economic incentives. The results have been quite striking: they indicate that control costs under existing programs have often been several times

the least-cost levels. (See Thomas Tietenberg 1985, ch. 3, for a useful survey of these cost studies.)

The source of these large cost savings is the capacity of economic instruments to take advantage of the large differentials in abatement costs across polluters. The information problems confronting regulators under the more traditional CAC approaches are enormous—and they lead regulators to make only very rough and crude distinctions among sources (e.g., new versus old firms). In a setting of perfect information, such problems would, of course, disappear. But in the real world of imperfect information, economic instruments have the important advantage of economizing on the need for the environmental agency to acquire information on the abatement costs of individual sources. This is just another example of the more general principles concerning the capacity of markets to deal efficiently with information problems.¹⁶

The estimated cost savings in the studies cited above result from a more cost effective allocation of abatement efforts within the context of existing control technologies. From a more dynamic perspective, economic incentives promise additional gains in terms of encouraging the development of more effective and less costly abatement techniques. As John Wenders (1975) points out in this context, a system that puts a value on any discharges remaining after control (such as a system of fees or marketable permits) will provide a greater incentive to R&D efforts in control technology than will a regulation that specifies some given level of discharges (see also Wesley Magat 1978, and Scott Milliman and Raymond Prince 1989).

¹⁵ This is admittedly a highly simplified view of the policy process. There is surely some interplay in debate and negotiations between the determination of standards and the choice of policy instruments. More broadly, there is an emerging literature on the political economy of environmental policy that seeks to provide a better understanding of the process of instrument choice—see, for example, McCubbins, Noll, and Weingast (1989), and Robert Hahn (1990).

¹⁶ There is also an interesting literature on incentive-compatible mechanisms to obtain abatement cost information from polluters—see, for example, Evan Kwerel (1977).

B. *The Choice of Policy Instruments Again*¹⁷

Some interesting issues arise in the choice between systems of effluent fees and marketable emission permits in the policy arena (John H. Dales 1968; Dewees 1983; David Harrison 1983). There is, of course, a basic sense in which they are equivalent: the environmental authority can, in principle, set price (i.e., the level of the effluent charge) and then adjust it until emissions are reduced sufficiently to achieve the prescribed environmental standard, or, alternatively, issue the requisite number of permits directly and allow the bidding of polluters to determine the market-clearing price.

However, this basic equivalence obscures some crucial differences between the two approaches in a policy setting; they are by no means equivalent policy instruments from the perspective of a regulatory agency. A major advantage of the marketable permit approach is that it gives the environmental authority direct control over the quantity of emissions. Under the fee approach, the regulator must set a fee, and if, for example, the fee turns out to be too low, pollution will exceed permissible levels. The agency will find itself in the uncomfortable position of having to adjust and readjust the fee to ensure that the environmental standard is attained. Direct control over quantity is to be preferred since the standard itself is prescribed in quantity terms.

This consideration is particularly important over time in a world of growth and inflation. A nominal fee that is adequate to hold emissions to the requisite levels at one moment in time will fail to

do so later in the presence of economic growth and a rising price level. The regulatory agency will have to enact periodic (and unpopular) increases in effluent fees. In contrast, a system of marketable permits automatically accommodates itself to growth and inflation. Since there can be no change in the aggregate quantity of emissions without some explicit action on the part of the agency, increased demand will simply translate itself into a higher market-clearing price for permits with no effects on levels of waste discharges.

Polluters (that is, *existing* polluters), as well as regulators, are likely to prefer the permit approach because it can involve lower levels of compliance costs. If the permits are auctioned off, then of course polluters must pay directly for the right to emit wastes as they would under a fee system. But rather than allocating the permits by auction, the environmental authority can initiate the system with a one-time distribution of permits to existing sources—free of charge. Some form of “grandfathering” can be used to allocate permits based on historical performance. Existing firms thus receive a marketable asset, which they can then use either to validate their own emissions or sell to another polluter.¹⁸ And finally, the permit approach has some advantages in terms of familiarity. Regulators have long-standing experience with permits, and it is a much less radical change to make permits effectively transferable than to introduce a wholly new system of regulation based on effluent fees. Mar-

¹⁷ For a useful, comprehensive survey of the strengths and weaknesses of alternative policy instruments for pollution control, see Bohm and Clifford Russell (1985).

¹⁸ In an interesting simulation study, Randolph Lyon (1982) finds that the cost of permits to sources under an auction system can be quite high; for one of the auction simulations, he finds that aggregate payments for permits will exceed treatment costs. Lyon's results thus suggest potentially large gains to polluting firms from a free distribution of permits instead of their sale through an auction. These gains, of course, are limited to current sources. Polluting firms that arrive on the scene at a later date will have to purchase permits from existing dischargers.

ketable permits thus have some quite appealing features to a regulatory agency—features that no doubt explain to some degree the revealed preference for this approach (in the U.S. at least) over that of fees.

Effluent charges have their own appeal. They are sources of public revenue, and, in these days of large budget deficits, they promise a new revenue source to hard-pressed legislators. From an economic perspective, there is much to be said for the substitution of fees for other sources of revenues that carry sizable excess burdens (Lee and Misiulek 1986). In a study of effluent charges on emissions of particulates and sulfur oxides from stationary sources into the atmosphere, David Terkla (1984) estimates, based on assumed levels of tax rates, that revenues in 1982 dollars would range from \$1.8 to \$8.7 billion and would, in addition, provide substantial efficiency gains (\$630 million to \$3.05 billion) if substituted for revenues from either the federal individual income tax or corporation income tax.

Moreover, the charges approach does not depend for its effectiveness on the development of a smoothly functioning market in permits. Significant search costs, strategic behavior, and market imperfections can impede the workings of a permit market (Hahn 1984; Tietenberg 1985, ch. 6). In contrast, under a system of fees, no transfers of permits are needed—each polluter simply responds directly to the incentive provided by the existing fee. There may well be circumstances under which it is easier to realize a cost-effective pattern of abatement efforts through a visible set of fees than through the workings of a somewhat distorted permit market. And finally, there is an equity argument in favor of fees (instead of a free distribution of permits to sources). The Organization for Economic Cooperation and Development (OECD), for example, has adopted the

“Polluter Pays Principle” on the grounds that those who use society’s scarce environmental resources should compensate the public for their use.

There exists a large literature on the design of fee systems and permit markets to attain predetermined levels of environmental quality. This work addresses the difficult issues that arise in the design and functioning of systems of economic incentives—issues that receive little or only perfunctory attention in the purely theoretical literature but are of real concern in the operation of actual policy measures. For example, there is the tricky matter of spatial differentiation. For most pollutants, the effect of discharges on environmental quality typically has important spatial dimensions: the specific location of the source dictates the effects that its emissions will have on environmental quality at the various monitoring points. While, in principle, this simply calls for differentiating the effluent fee according to location, in practice this is not so easy. The regulatory agency often does not have the authority or inclination to levy differing tax rates on sources according to their location. Various compromises including the construction of zones with uniform fees have been investigated (Tietenberg 1978; Seskin, Anderson, and Reid 1983; Kolstad 1987).

Similarly, problems arise under systems of transferable permits where (as is often the case) the effects of the emissions of the partners to a trade are not the same. (The seminal theoretical paper is W. David Montgomery 1972.) Several alternatives have been proposed including zoned systems that allow trades only among polluters within the specified zones, ambient permit systems under which the terms of trade are determined by the relative effects of emissions at binding monitors, and the pollution-offset system under which trades are sub-

ject to the constraint of no violations of the prevailing standard at any point in the area (Atkinson and Tietenberg 1982; Atkinson and Lewis 1974; Hahn and Noll 1982; Krupnick, Oates, and Eric Van de Verg 1983; McGartland and Oates 1985; McGartland 1988; Tietenberg 1980, 1985; Walter Spofford 1984; Baumol and Oates 1988, ch. 12). For certain pollutants, these studies make clear that a substantial portion of the cost-savings from economic-incentive approaches will be lost if spatial differentiation is not, at least to some degree, built into the program (Robert Mendelsohn 1986).

The actual design of systems of economic incentives inevitably involves some basic compromises to accommodate the range of complications to the regulatory problem (Albert Nichols 1984). It is instructive to see how some of these issues have been dealt with in practice.

C. *Experience with Economic Incentives for Environmental Management*¹⁹

In the United States proposals for effluent fees have met with little success; however, there has been some limited experience with programs of marketable permits for the regulation of air and water quality. In Europe, the experience (at least until quite recently) has been the reverse: some modest use of effluent charges but no experience with transferable permits. We shall provide in this section a brief summary of these measures along with some remarks on their achievements and failures.

Largely for the reasons mentioned in the preceding section, policy makers in the U.S. have found marketable permits preferable to fees as a mechanism for providing economic incentives for pollution

¹⁹ The OECD (1989) has recently provided a useful "catalog" and accompanying discussion of the use of economic incentives for environmental protection in the OECD countries.

control.²⁰ The major program of this genre is the EPA's Emission Trading Program for the regulation of air quality. But there are also three other programs worthy of note: the Wisconsin system of Transferable Discharge Permits (TDP) for the management of water quality, the lead trading program (known formally as "interrefinery averaging"), and a recent program for the trading of rights for phosphorus discharges into the Dillon Reservoir in Colorado.²¹

By far the most important of these programs in terms of scope and impact, Emissions Trading has undergone a fairly complicated evolution into a program that has several major components. Under the widely publicized "Bubble" provision, a plant with many sources of emissions of a particular air pollutant is subjected to an overall emissions limitation. Within this limit, the managers of the plant have the flexibility to select a set of controls consistent with the aggregate limit, rather than conforming to specified treatment procedures for each source of discharges with the plant. Under the "Netting" provision, firms can avoid stringent limitations on new sources

²⁰ One case in which there has been some use of fees in the U.S. is the levying of charges on industrial emissions into municipal waste treatment facilities. In some instances these charges have been based not only on the quantity but also on the strength or quality of the effluent. The charges are often related to "average" levels of discharges and have had as their primary objective the raising of funds to help finance the treatment plants. Their role as an economic incentive to regulate levels of emissions has apparently been minor (see James Boland 1986; Baumol and Oates 1979, pp. 258-63). There are also a variety of taxes on the disposal of hazardous wastes, including land disposal taxes in several states.

²¹ Tietenberg's book (1985) is an excellent, comprehensive treatment of the Emissions Trading Program. Robert Hahn and Gordon Hester have provided a series of recent and very valuable descriptions and assessments of all four of these programs of marketable permits. See Hahn and Hester (1989a, 1989b), and Hahn (1989). For analyses of the Wisconsin TDP system, see William O'Neil (1983), and O'Neil et al. (1983).

of discharges by reducing emissions from other sources of the pollutant within the facility. Hahn and Hester (1989b) report that to date there have been over 100 approved Bubble transactions in the U.S. and a much larger number of Netting "trades" (somewhere between 5,000 and 12,000). The estimated cost savings from these trades have been quite substantial; although the estimates exhibit a very wide range, the cost savings probably amount to several billion dollars.

There are provisions under Emissions Trading for external trades across firms—mainly under the Offset provision which allows new sources in nonattainment areas to "offset" their new emissions with reductions in discharges by existing sources. Offsets can be obtained through either internal (within plant) or external trades. Hahn and Hester (1989b) indicate that there have been about 2,000 trades under the Offset policy; only about 10 percent of them have been external trades—the great bulk of offsets have been obtained within the plant or facility.

Emissions Trading, as a whole, receives mixed marks. It has significantly increased the flexibility with which sources can meet their discharge limitations—and this has been important for it has allowed substantial cost savings. The great majority of the trades, however, have been internal ones. A real and active market in emissions rights involving different firms has not developed under the program (in spite of the efforts of an active firm functioning as a broker in this market). This seems to be largely the result of an extensive and complicated set of procedures for external trades that have introduced substantial levels of transactions costs into the market and have created uncertainties concerning the nature of the property rights that are being acquired. In addition, the program has been grafted onto an elaborate set of command-and-control style

regulations which effectively prohibit certain kinds of trades. Many potentially profitable trades simply have not come to pass.²²

Likewise, the experience under the Wisconsin TDP system has involved little external trading. The program establishes a framework under which the rights to BOD discharges can be traded among sources. Since the program's inception in 1981 on the Fox River, there has been only one trade: a paper mill which shifted its treatment activities to a municipal wastewater treatment plant transferred its rights to the municipal facility. The potential number of trades is limited since there are only about twenty major sources (paper mills and municipal waste treatment plants) along the banks of the river. But even so, preliminary studies (O'Neil 1983; O'Neil et al. 1983) indicated several potentially quite profitable trades involving large cost savings. A set of quite severe restrictions appears to have discouraged these transfers of permits. Trades must be justified on the basis of "need"—and this does not include reduced costs! Moreover, the traded rights are granted only for the term of the seller's discharge permit (a maximum period of five years) with no assurance that the rights will be renewed. The Wisconsin experience seems to be one in which the conditions needed for the emergence of a viable market in discharge permits have not been established.

In contrast, EPA's "interrefinery averaging" program for the trading of lead rights resulted in a very active market over the relatively short life of the program. Begun in 1982, the program allowed refiners to trade the severely lim-

²² In an interesting analysis of the experience with Emissions Trading, Roger Raufer and Stephen Feldman (1987) argue that some of the obstacles to trading could be circumvented by allowing the leasing of rights.

ited rights to lead additives to gasoline. The program expired in 1986, although refiners were permitted to make a use of rights that were "banked" through 1987. Trading became brisk under the program: over the first half of 1987, for example, around 50 percent of all lead added to gasoline was obtained through trades of lead rights, with substantial cost savings reported from these trades. Although reliable estimates of cost-savings for the lead-trading program are not available, Hahn and Hester (1989b) surmise that these savings have run into the hundreds of millions of dollars. As they point out, the success of the program stemmed largely from the absence of a large body of restrictions on trades: refiners were essentially free to trade lead rights and needed only to submit a quarterly report to EPA on their gasoline production and lead usage. There were, moreover, already well established markets in refinery products (including a wide variety of fuel additives) so that refinery managers had plenty of experience in these kinds of transactions.²³

Finally, there is an emerging program in Colorado for the trading of rights to phosphorous discharges into the Dillon Reservoir. This program is noteworthy in that among those that we have discussed, it is the only one to be designed and introduced by a local government. The plan embodies few encumbrances to trading; the one major restriction is a 2:1 trading ratio for point/nonpoint trading, introduced as a "margin of safety" because of uncertainties concerning the effectiveness of nonpoint source controls. The program is still in its early stages: although no trades have been approved, some have been requested.

The U.S. experience with marketable

permits is thus a limited one with quite mixed results. In the one case where the market was allowed to function free of heavy restrictions, vigorous trading resulted with apparently large cost savings. In contrast, under Emissions Trading and the Wisconsin TDP systems, stringent restrictions on the markets for trading emissions rights appear to have effectively increased transaction costs and introduced uncertainties, seriously impeding the ability of these markets to realize the potentially large cost savings from trading. Even so, the cost savings from Emissions Trading (primarily from the Netting and Bubble provisions) have run into several billion dollars. Finally, it is interesting that these programs seem not to have had any significant and adverse environmental effects; Hahn and Hester (1989a) suggest that their impact on environmental quality has been roughly "neutral."

In light of this experience, the prospects, we think, appear favorable for the functioning of the new market in sulfur allowances that is being created under the 1990 Amendments to the Clean Air Act. This measure, designed to address the acid rain problem by cutting back annual sulfur emissions by 10 million tons, will permit affected power plants to meet their emissions reduction quotas by whatever means they wish, including the purchase of "excess" emissions reductions from other sources. The market area for this program is the nation as a whole so that there should be a large number of potential participants in the market. At this juncture, plans for the structure and functioning of the market do not appear to contain major limitations that would impede trading in the sulfur allowances. There remains, however, the possibility that state governors or public utility commissions will introduce some restrictions. There is the further concern that regulated firms may not behave

²³ We should also note that various irregularities and illegal procedures were discovered in this market—perhaps because of lax oversight.

in a strictly cost-minimizing fashion, thereby compromising some of the cost-effectiveness properties of the trading scheme. But as we suggested earlier, this may not prove to be a serious distortion.

The use of effluent fees is more prevalent in Europe where they have been employed extensively in systems of water quality management and to a limited extent for noise abatement (Ralph Johnson and Gardner Brown, Jr. 1976; Bower et al. 1981; Brown and Hans Bressers 1986; Brown and Johnson 1984; Tietenberg 1990). There are few attempts to use them for the control of air pollution. France, Germany, and the Netherlands, for example, have imposed effluent fees on emissions of various water pollutants for over two decades. It should be stressed that these fee systems are not pure systems of economic incentives of the sort discussed in economics texts. Their primary intent has not been the regulation of discharges, but rather the raising of funds to finance projects for water quality management. As such, the fees have typically been low and have tended to apply to "average" or "expected" discharges rather than to provide a clear cost signal at the margin. Moreover, the charges are overlaid on an extensive command-and-control system of regulations that mute somewhat further their effects as economic incentives.

The Netherlands has one of the oldest and most effectively managed systems of charges—and also the one with relatively high levels of fees. There is some evidence suggesting that these fees have, in fact, had a measurable effect in reducing emissions. Some multiple regression work by Hans Bressers (1983) in the Netherlands and surveys of industrial polluters and water board officials by Brown and Bressers (1986) indicate that firms have responded to the charges with significant cutbacks in discharges of water borne pollutants.

In sum, although there is some experi-

ence with systems of fees for pollution control, mainly of water pollution, these systems have not, for the most part, been designed in the spirit of economic incentives for the regulation of water quality. Their role has been more that of a revenue device to finance programs for water quality management.

These systems, it is worth noting, have addressed almost exclusively so-called "point-source" polluters. Non-point source pollution (including agricultural and urban runoff into waterways) has proved much more difficult to encompass within systems of charges or permits. Winston Harrington, Krupnick, and Henry Peskin (1985) provide a useful overview of the potential role for economic incentives in the management of non-point sources. This becomes largely a matter of seeking out potentially effective second-best measures (e.g., fees on fertilizer use), since it is difficult to measure and monitor "discharges" of pollutants from these sources. Kathleen Segerson (1988) has advanced an ingenious proposal whereby such sources would be subject to a tax (or subsidy payment) based, not on their emissions, but on the observed level of environmental quality; although sources might find themselves with tax payments resulting from circumstances outside their control (e.g., adverse weather conditions), Segerson shows that such a scheme can induce efficient abatement and entry/exit behavior on the part of non-point sources.

D. Legal Liability as an Economic Instrument for Environmental Protection

An entirely different approach to regulating sources is to rely on legal liability for damages to the environment. Although we often do not include this approach under the heading of economic instruments, it is clear that a system of "strict liability," under which a source is financially responsible for damages,

embodies important economic incentives.²⁴ The imposition of such liability effectively places an "expected price" on polluting activities. The ongoing suits, for example, following upon the massive Exxon-Valdez oil spill suggest that such penalties will surely exert pressures on potential polluters to engage in preventive measures.

Under this approach, the environmental authority, in a setting of uncertainty, need not set the values of any price or quantity instruments; it simply relies on the liability rule to discipline polluters. Two issues are of interest here. The first is the capacity, in principle, for strict liability to mimic the effects of a Pigouvian tax. And the second is the likely effectiveness, in practice, of strict liability as a substitute for other forms of economic incentives. There is a substantial literature in the economics of the law that addresses these general issues and a growing number of studies that explore this matter in the context of environmental management (see, for example, Steven Shavell 1984a, 1984b; Segerson 1990).

It is clear that strict liability can, in principle, provide the source of potential damages with the same incentive as a Pigouvian tax. If a polluter knows that he will be held financially accountable for any damages his activities create, then he will have the proper incentive to seek methods to avoid these damages. Strict liability serves to internalize the external costs—just as does an appropriate tax. Strict liability is unlike a tax, however, in that it provides compensation to victims. The Pigouvian tax possesses an important asymmetry in a market sense: it is a charge to the polluter—but not a payment to the victim. And, as noted

earlier, such payments to victims can result in inefficient levels of defensive activities. Strict liability thus does not get perfect marks on efficiency grounds, even in principle, for although it internalizes the social costs of the polluter, it can be a source of distortions in victims' behavior.

The more important concern, in practice, is the effectiveness of legal liability in disciplining polluter behavior. Even if the basic rule is an efficient one in terms of placing liability on the source of the environmental damage, the actual "price" paid by the source may be much less than actual damages because of imperfections in the legal system: failures to impose liability on responsible parties resulting from uncertainty over causation, statutes of limitation, or high costs of prosecution.²⁵ There is the further possibility of bankruptcy as a means of avoiding large payments for damages. The evidence on these matters is mixed (see Segerson 1990), but it seems to suggest that legal liability has functioned only very imperfectly.

An interesting area of application in the environmental arena involves various pieces of legislation that provide strict liability for damages from accidental spills of oil or leakage of hazardous wastes. The Comprehensive Environmental Responses, Compensation, and Liability Act (CERCLA) of 1980 and its later amendments (popularly known as "Superfund") are noteworthy for their broad potential applicability (Thomas Grigalunas and James Opaluch 1988). Such measures may well provide a useful framework for internalizing the external

²⁴ The major alternative to strict liability is a negligence rule under which a polluter is liable only if he has failed to comply with a "due standard of care" in the activity that caused the damages. Under strict liability, the party causing the damages is liable irrespective of the care exercised in the polluting activity.

²⁵ As one reviewer noted, in these times of heightened environmental sensitivity, liability determinations could easily exceed actual damages in some instances. However, this seems not to have happened in the recent Exxon-Valdez case. The case was settled out of court with Exxon agreeing to pay some \$900 million over a period of several years. Some observers believe that this falls well short of the true damages from the Exxon-Valdez oil spill in Alaska.

costs of spills (Opaluch and Grigalunas 1984). In particular, the liability approach appears to have its greatest appeal in cases like those under Superfund where damages are infrequent events and for which monitoring the level of care a firm takes under conventional regulatory procedures would be difficult.²⁶

E. *Environmental Federalism*

In addition to the choice of policy instrument, there is the important issue of the locus of regulatory authority. In the case of fees, for example, should a central environmental authority establish a uniform fee applicable to polluters in all parts of the nation or should decentralized agencies set fee levels appropriate to their own jurisdictions? U.S. environmental policy exhibits considerable ambivalence on this matter. Under the Clean Air Act in 1970, the U.S. Congress instructed the Environmental Protection Agency to set uniform national standards for air quality—maximum permissible concentrations of key air pollutants applicable to all areas in the country. But two years later under the Clean Water Act, the Congress decided to let the individual states determine their own standards (subject to EPA approval) for water quality. The basic question is “Which approach, centralized decision making or environmental federalism, is the more promising?”

Basic economic principles seem to suggest, on first glance, a straightforward answer to this question. Since the benefits and costs of reduced levels of most forms of pollution are likely to vary (and vary substantially) across different jurisdic-

tions, the optimal level of effluent fees (or quantities of marketable permits) will also vary (Sam Peltzman and T. Nicolaus Tideman 1972). The first-best outcome must therefore be one in which fees or quantities of permits are set in accord with local circumstances, suggesting that an optimal regulatory system for pollution control will be a form of environmental federalism.

Some environmental economists have raised an objection to this general presumption. John Cumberland (1981), among others, has expressed the concern that in their eagerness to attract new business and jobs, state or local officials will tend to set excessively lax environmental standards—fees that are too low or quantities of permits that are too high. The fear is that competition among decentralized jurisdictions for jobs and income will lead to excessive environmental degradation. This, incidentally, is a line of argument that has appeared elsewhere in the literature on fiscal federalism under the title of “tax competition.” The difficulty in assessing this objection to decentralized policy making is that there exists little systematic evidence on the issue; most of the evidence is anecdotal in character, and, until quite recently, there has been little theoretical work addressing the phenomenon of interjurisdictional competition.²⁷

In a pair of recent papers, Oates and Robert Schwab (1988a, 1988b) have set forth a model of such competition in which “local” jurisdictions compete for a mobile national stock of capital using both tax and environmental policy instruments. Since the production functions

²⁶ A more complicated and problematic issue relates to the permission of the courts to sue under Superfund for damages from toxic substances using “the joint and several liability doctrine.” Under this provision, *each* defendant is potentially liable for an amount up to the *entire* damage, irrespective of his individual contribution. For an analysis of this doctrine in the Superfund setting, see Tietenberg (1989).

²⁷ Two recent studies, one by Virginia McConnell and Schwab (1990), and the other by Timothy Bartik (1988c), find little evidence of strong effects of existing environmental regulations on the location decisions of firms within the U.S. This, of course, does not preclude the possibility that state and local officials, in *fear* of such effects, will scale down standards for environmental quality.

are neoclassical in character, an increase in a jurisdiction's capital stock raises the level of wages through an associated increase in the capital-labor ratio. In the model, local officials simultaneously employ two policy tools to attract capital: a tax rate on capital itself which can be lowered or even set negative (a subsidy) to raise the return to capital in the jurisdiction, and a level of allowable pollutant emissions (or, alternatively, an effluent fee). By increasing the level of permissible waste discharges either directly or by lowering the fee on emissions, the local authority increases the marginal product of capital and thereby encourages a further inflow of capital. The model thus involves two straightforward tradeoffs: one between wage income and tax revenues, and the other between wage income and local environmental quality. The analysis reveals that in a setting of homogeneous worker-residents making choices by simple majority rule, jurisdictions select the socially optimal levels of these two policy instruments. The tax rate on capital is set equal to zero, and the level of environmental quality is chosen so that the willingness to pay for a cleaner environment is equal to marginal abatement cost. The analysis thus supports the case for environmental federalism: decentralized policy making is efficient in the model.²⁸

In one sense, this is hardly a surprising result. Since local residents care about the level of environmental quality, we should not expect that they would wish to push levels of pollution into the range where the willingness to pay to avoid environmental damage exceeds the loss in wage income from a cleaner environment. At the same time, this result is

not immune to various "imperfections." If, for example, local governments are constrained constitutionally to use taxes on capital to finance various local public goods, then it is easy to show that not only will the tax rate on capital be positive, but officials will select socially excessive levels of pollution. Likewise, if Niskanen bureaucrats run the local public sector, they will choose excessively lax environmental standards as a mechanism to attract capital so as to expand the local tax base and public revenues. Finally, there can easily be conflicts among local groups of residents with differing interests (e.g., workers vs. non-workers) that can lead to distorted outcomes (although these distortions may involve too little or too much pollution).

The basic model does at least suggest that there are some fundamental forces promoting efficient decentralized environmental decisions. If the regions selected for environmental decision making are sufficiently large to internalize the polluting effects of waste discharges, the case for environmental federalism has some force. Exploration of this issue is admittedly in its infancy—in particular, there is a pressing need for some systematic empirical study of the effects of "local" competition on environmental choices.²⁹

F. *Enforcement Issues*

The great bulk of the literature on the economics of environmental regulation simply assumes that polluters comply with existing directives: they either keep their discharges within the prescribed limitation or, under a fee scheme, report accurately their levels of emissions and pay the required fees.

²⁸ Using an alternative analytical framework in which local jurisdictions "bid" against one another for polluting firms in terms of entry fees, William Fischel (1975) likewise finds that local competition produces an efficient outcome.

²⁹ For some other recent theoretical studies of interjurisdictional fiscal competition, see Jack Mintz and Henry Tulkens (1986), John Wilson (1986), David Wildasin (1989), and George Zodrow and Peter Mieszowski (1986).

Sources, in short, are assumed *both* to act in good faith and to have full control over their levels of discharges so that violations of prescribed behavior do not occur.

Taking its lead from the seminal paper by Gary Becker (1968) on the economics of crime and punishment, a recent literature has addressed enforcement issues as they apply to environmental regulations.³⁰ As this literature points out, violations of environmental regulations can have two sources: a polluter can willfully exceed his discharge limitation (or under-report his emissions under a fee system) to reduce compliance costs or a stochastic dimension to discharges may exist so that the polluter has only imperfect control over his levels of emissions. In such a setting, the regulatory problem becomes a more complicated one. Not only must the regulatory agency set the usual policy parameters (emissions limitations or fees), but it must also decide upon an enforcement policy which involves both monitoring procedures and levels of fines for violations.

The early literature explored these enforcement issues in a wholly static framework. The seminal papers, for example, by Paul Downing and William Watson (1974) and by Jon Harford (1978), established a number of interesting results. Downing and Watson show that the incorporation of enforcement costs into the analysis of environmental policy suggests that optimal levels of pollution control will be less than when these costs are ignored. Harford obtains the especially interesting result that under a system of effluent fees, the level of *actual* dis-

charges is independent both of the level of the fine for underreporting and of the probability of punishment (so long as the slope of the expected penalty function with respect to the size of the violation is increasing and the probability of punishment is greater than zero). The polluter sets the level of actual wastes such that marginal abatement cost equals the effluent fee—the efficient level! But he then, in general, underreports his discharges with the extent of underreporting varying inversely with the level of fines and the probability of punishment.

Arun Malik (1990) has extended this line of analysis to the functioning of systems of marketable permits. He establishes a result analogous to Harford's: under certain circumstances, noncompliant polluters will emit precisely the same level of wastes for a given permit price as that discharged by an otherwise identical compliant firm. The conditions, however, for this equivalence are fairly stringent ones. More generally, Malik shows that noncompliant behavior will have effects on the market-clearing price in the permit market—effects that will compromise to some extent the efficiency properties of the marketable permit system.

One implication of this body of work is the expectation of widespread noncompliance on the part of polluters. But as Harrington (1988) points out, this seems not to be the case. The evidence we have from various spot checks by EPA and GAO suggests that most industrial polluters seem to be in compliance most of the time.³¹ Substantial compliance seems

³⁰ Russell, Harrington, and William Vaughan (1986, ch. 4) provide a useful survey of the enforcement literature in environmental economics up to 1985. Harrington (1988) presents a concise, excellent overview both of the more recent literature and of the “stylized facts” of actual compliance and enforcement behavior. See also Russell (1990).

³¹ Interestingly, noncompliance seems to be more widespread among municipal waste treatment plants than among industrial sources! (Russell 1990, p. 256). Some of the most formidable enforcement problems involve federal agencies. The GAO (1988), for example, has found the Department of Energy's nuclear weapons facilities to be a source of major concern; the costs of dealing with environmental contamination associated with these facilities are estimated at more than \$100 billion.

to exist in spite of modest enforcement efforts: relatively few “notices of violation” have been issued and far fewer polluters have actually been fined for their violations. Moreover, where such fines have been levied, they have typically been quite small. And yet in spite of such modest enforcement efforts, “cheating” is not ubiquitous—violations are certainly not infrequent, but they are far from universal.

This finding simply doesn't square at all well with the results from the static models of polluter behavior.³² An alternative line of modeling (drawing on the tax-evasion literature) seems to provide a better description of polluter behavior; it also has some potentially instructive normative implications. This approach puts the problem in a dynamic game-theoretic framework. Both polluters and regulators react to the activities of one another in the previous period. In a provocative paper, Harrington (1988) models the enforcement process as a Markov decision problem. Polluters that are detected in violation in one period are moved to a separate group in the next period in which they are subject to more frequent inspection and higher fines. Polluting firms thus have an incentive to comply in order to avoid being moved into the second group (from which they can return to the original group only after a period during which no violations are detected). In such a framework, firms may be in compliance even though they would be subject to no fine for a violation. Following up on Russell's analysis (Russell, Harrington, and Vaughan 1986, pp. 199–216), Harrington finds that the addition of yet a third group, an absorbing state from which the polluter can never emerge, can result in a “spectacu-

lar reduction in the minimum resources required to achieve a given level of compliance” (p. 47). In sum, the dynamic game-theoretic approach can produce compliance in cases in which the expected penalty is insufficient to prevent violations in a purely static model. Moreover, it suggests some potentially valuable guidelines for the design of cost-effective enforcement procedures. Enforcement is an area where economic analysis may make some quite useful contributions.

G. *The Effects of Domestic Environmental Policy on Patterns of International Trade*

The introduction of policy measures to protect the environment has potential implications not only for the domestic economy but also for international trade. Proposed environmental regulations are, in fact, often opposed vigorously on the grounds that they will impair the “international competitiveness” of domestic industries. The increased costs associated with pollution control measures will, so the argument goes, result in a loss of export markets and increased imports of products of polluting industries.

These potential effects have been the subject of some study. It is clear, for example, that the adoption of costly control measures in certain countries will, in principle, alter the international structure of relative costs with potential effects on patterns of specialization and world trade. These trade effects have been explored in some detail, making use of standard models of international trade (Kazumi Asako 1979; Baumol and Oates 1988, ch. 16; Anthony Koo 1974; Martin McGuire 1982; John Merrifield 1988; Rüdiger Pethig 1976; Pethig et al. 1980; Horst Siebert 1974; James Tobey 1989; Ingo Walter 1975). In particular, there has been a concern that the less developed countries, with their emphasis on

³² Perhaps public opprobrium is a stronger disciplinary force than economists are typically inclined to believe!

economic development rather than environmental protection, will tend over time to develop a comparative advantage in pollution-intensive industries. In consequence, they will become the "havens" for the world's dirty industries; this concern has become known as the "pollution-haven hypothesis" (Walter and Judith Ugelow 1979; Walter 1982).

Some early studies made use of existing macro-econometric models to assess the likely magnitudes of these effects. These studies used estimates of the costs of pollution control programs on an industry basis to get some sense of the effects of these programs on trade and payments flows. Generally, they found small, but measurable, effects (d'Arge and Kneese 1971; Walter 1974).

We are now in a position to examine historically what has, in fact, happened. To what extent have environmental measures influenced the pattern of world trade? Have the LDC's become the havens of the world's dirty industries? Two recent studies, quite different in character, have addressed this issue directly. H. Jeffrey Leonard (1988), in what is largely a case study of trade and foreign-investment flows for several key industries and countries, finds little evidence that pollution-control measures have exerted a systematic effect on international trade and investment. After examining some aggregate figures, the policy stances in several industrialized and developing countries, and the operations of multinational corporations, Leonard concludes that "the differentials in the costs of complying with environmental regulations and in the levels of environmental concern in industrialized and industrializing countries have not been strong enough to offset larger political and economic forces in shaping aggregate international comparative advantage" (p. 231).

Tobey (1989, 1990) has looked at the

same issue in a large econometric study of international trade patterns in "pollution-intensive" goods. After controlling for the effects of relative factor abundance and other trade determinants, Tobey cannot find any effects of various measures of the stringency of domestic environmental policies. Tobey estimates two sets of equations that explain, respectively, patterns of trade in pollution-intensive goods and changes in trade patterns from 1970 to 1984. In neither set of equations do the variables measuring the stringency of domestic environmental policy have the predicted effect on trade patterns.

Why have domestic environmental measures not induced "industrial flight;" and the development of "pollution havens?" The primary reason seems to be that the costs of pollution control have not, in fact, loomed very large even in heavily polluting industries. Existing estimates suggest that control costs have run on the order of only 1 to 2½ percent of total costs in most pollution-intensive industries; H. David Robison (1985, p. 704), for example, reports that total abatement costs per dollar of output in 1977 were well under 3 percent in all industries with the sole exception of electric utilities where they were 5.4 percent. Such small increments to costs are likely to be swamped in their impact on international trade by the much larger effects of changing differentials in labor costs, swings in exchange rates, etc. Moreover, nearly all the industrialized countries have introduced environmental measures—and at roughly the same time—so that such measures have not been the source of significant cost differentials among major competitors. There seems not to have been a discernible movement in investment in these industries to the developing countries because major political and economic uncertainties have apparently loomed much larger

in location decisions than have the modest savings from less stringent environmental controls.

In short, domestic environmental policies, at least to this point in time, do not appear to have had significant effects on patterns of international trade. From an environmental perspective, this is a comforting finding, for it means that there is little force to the argument that we need to relax environmental policies to preserve international competitiveness.

H. *Command-and-Control vs. Economic Incentives: Some Concluding Observations*

Much of the literature in environmental economics, both theoretical and empirical, contrasts in quite sharp and uncompromising terms the properties of systems of economic incentives with the inferior outcomes under existing systems of command-and-control regulations. In certain respects, this literature has been a bit misleading and, perhaps, unfair. The term command-and-control encompasses a very broad and diverse set of regulatory techniques—some admittedly quite crude and excessively costly. But others are far more sophisticated and cost sensitive. In fact, the dividing line between so-called CAC and incentive-based policies is not always so clear. A program under which the regulator specifies the exact treatment procedures to be followed by polluters obviously falls within the CAC class. But what about a policy that establishes a fixed emissions limitation for a particular source (with no trading possible) but allows the polluter to select the form of compliance? Such flexibility certainly allows the operation of economic incentives in terms of the search for the least-cost method of control.

The point here is that it can be quite misleading to lump together in a cavalier

fashion “CAC” methods of regulatory control and to contrast them as a class with the least-cost outcomes typically associated with systems of economic incentives. In fact, the compromises and “imperfections” inherent in the design and implementation of incentive-based systems virtually guarantee that they also will be unable to realize the formal least-cost result.

Empirical studies contrasting the cost effectiveness of the two general approaches have typically examined the cost under each system of attaining a specified *standard* of environmental quality—which typically means ensuring that at no point in an area do pollutant concentrations exceed the maximum level permissible under the particular standard. As Atkinson and Tietenberg (1982) and others have noted, CAC systems typically result in substantial “over-control” relative to incentive-based systems. Since it effectively assigns a zero shadow price to any environmental improvements over and above the standard, the least-cost algorithm attempts to make use of any “excess” environmental capacity to increase emissions and thereby reduce control costs. The less cost-sensitive CAC approaches generally overly restrict emissions (relative to the least-cost solution) and thereby produce pollutant concentrations at nonbinding points that are less than those under the least-cost outcome. In sum, at most points in the area, environmental quality (although subject to the same overall standard) will be higher under a CAC system than under the least-cost solution. So long as there is some value to improved environmental quality beyond the standard, a proper comparison of benefits and costs should give the CAC system credit for this increment to environmental quality. One recent study (Oates, Paul Portney, and McGartland 1989) which does just this for a major air pollutant finds that a rela-

tively sophisticated CAC approach produces results that compare reasonably well to the prospective outcome under a fully cost effective system of economic incentives.

Our intent is not to suggest that the economist's emphasis on systems of economic incentives has been misplaced, but rather to argue that policy structure and analysis is a good deal more complicated than the usual textbooks would suggest (Nichols 1984). The applicability of systems of economic incentives is to some extent limited by monitoring capabilities and spatial complications. In fact, in any meaningful sense the "optimal" structure of regulatory programs for the control of air and water pollution is going to involve a combination of policy instruments—some making use of economic incentives and others not. Careful economic analysis has, we believe, an important role to play in understanding the workings of these systems. But it can make its best contribution, not through a dogmatic commitment to economic incentives, but rather by the careful analysis of the whole range of policy instruments available, insuring that those CAC measures that are adopted are effective devices for controlling pollution at relatively modest cost (Kolstad 1986).

At the same time, it is our sense that incentive-based systems have much to contribute to environmental protection—and that they have been much neglected in part because of the (understandable) predisposition of regulators to more traditional policy instruments.³³ There are strong reasons for believing, with supporting evidence, that this neglect has seriously impaired our efforts both to realize our objectives for improved environmental quality and to do

so at the lowest cost. A general realization of this point seems to be emerging with a consequent renewed interest in many countries in the possibility of integrating incentive-based policies into environmental regulations—a matter to which we shall return in the concluding section.

IV. *Measuring the Benefits and Costs of Pollution Control*

As we suggested in the previous sections, effluent fees and transferable permits are capable, in principle, of achieving a given pollution standard at least cost. Eventually, however, economists must ask whether environmental standards have been set at appropriate levels: does the marginal cost of achieving the ozone standard in the Los Angeles basin exceed the marginal benefits? The answer to this question requires that we measure the benefits and costs of pollution control.

While the measurement of control costs is itself no simple task, environmental economists have turned most of their attention to the benefit side of the ledger. Of central concern has been the development of methodologies to measure the benefits of goods—such as clean air or water—that are not sold in markets. These techniques fall into two categories: indirect market methods, which attempt to infer from actual choices, such as choosing where to live, the value people place on environmental goods; and direct questioning approaches, which ask people to make tradeoffs between environmental and other goods in a survey context. We shall review both approaches, and then discuss the application of these methods to valuing the benefits of pollution control. In particular, we will try to highlight areas where benefits have been successfully measured, as well as areas where good benefit estimates are

³³ See Steven Kelman (1981) for a fascinating—if somewhat dismaying—study of the politics and ideology of economic incentives for environmental protection.

most needed. But first we must be clear about the valuation of changes in environmental quality.

A. *Defining the Value of a Change in Environmental Quality*

We noted at the beginning of this review that pollution may enter both consumers' utility functions and firms' production functions. (See equations (1) and (2).) To elaborate on how this might occur we introduce a *damage function* that links pollution, Q , to something people value, S ,

$$S = S(Q). \quad (8)$$

For a consumer, S might be time spent ill or expected fish catch; for a firm it might be an input into production, such as the stock of halibut. We assume that S replaces Q in the utility and production functions (equations (1) and (2)).

There are two cases of interest here. First, if the consumer (or firm) views S as out of his control, we can define the value of a change in S (which may be easier to measure than the value of a change in Q), and then predict the change in S resulting from a change in Q . For example, if people view reductions in visibility associated with air pollution as beyond their control, one can predict the reduction in visibility from (8) and concentrate on valuing visibility. This is commonly known as the damage function approach to benefit estimation.

The second case is more complicated. It may sometimes be possible to mitigate the effects of pollution through the use of inputs, Z . For example, medicine may exist to alleviate respiratory symptoms associated with air pollution. In this instance, equation (8) must be modified to

$$S = S(Q, Z), \quad (9)$$

and it is Q rather than S that must be valued, because S is no longer exogenous.

For the case of a firm, the value of a change in Q (or S) is the change in the firm's profits when Q (or S) is altered. This amount is the same whether we are talking about the firm's willingness to pay (*WTP*) for an improvement in environmental quality or its willingness to accept (*WTA*) compensation for a reduction in environmental quality.

For a consumer, in contrast, the value of a change in Q (or S) depends on the initial assignment of property rights. If consumers are viewed as having to pay for an improvement in environmental quality, for example, from Q^0 to Q^1 , the most they should be willing to pay for this change is the reduction in expenditure necessary to achieve their original utility level when Q improves. Formally, if $e(P, S(Q^0), U^0)$ denotes the minimum expenditure necessary to achieve pre-improvement utility U^0 at prices P and environmental quality Q^0 , then the most people would be willing to pay (*WTP*) for the improvement in environmental quality to Q^1 is

$$WTP = e(P, S(Q^0), U^0) - e(P, S(Q^1), U^0). \quad (10)$$

If, on the other hand, consumers are viewed as having rights to the higher level of environmental quality and must be compensated for a reduction in Q , then the smallest amount they would be willing to accept is the additional amount they must spend to achieve their original utility level when Q declines. Formally, willingness to accept (*WTA*) compensation for a reduction in Q from Q^1 to Q^0 is given by

$$WTA = e(P, S(Q^0), U^1) - e(P, S(Q^1), U^1), \quad (11)$$

where U^1 is the utility level achieved at the higher level of environmental quality.

In general, willingness to accept com-

compensation for a reduction in Q will be higher than willingness to pay for an increase in Q of the same magnitude. As W. Michael Hanemann (1991) has recently shown, the amount by which WTA exceeds WTP varies directly with the income elasticity of demand for S and inversely with the elasticity of substitution between S and private goods. If the income elasticity of demand for S is zero or if S is a perfect substitute for a private good, WTP should equal WTA . If, however, the elasticity of substitution between S and private goods is zero, the difference between WTA and WTP can be infinite. It is therefore important to determine which valuation concept, WTP or WTA , is appropriate for the problem at hand.

The preceding definitions of the value of a change in environmental quality do not by themselves characterize all of the welfare effects of environmental policies. Improvements in environmental quality may alter prices as well as air or water quality, and these price changes must be valued in addition to quality changes.

In contrast to valuing quality changes, valuing price changes is relatively straightforward. WTP for a reduction in price is just the reduction in expenditure necessary to achieve U^0 (the consumer's original utility level) when prices are reduced. As is well known, this is just the area to the left of the relevant compensated demand function (i.e., the one that holds utility at U^0) between the two prices. Willingness to accept compensation for a price increase is the increase in expenditure necessary to achieve U^1 , the utility level enjoyed at the lower price, when price is increased.

Unlike the case of a quality change, WTA compensation for a price increase exceeds WTP for a price decrease only by the amount of an income effect. As long as expenditure on the good in question is a small fraction of total expendi-

ture, the difference between the two welfare measures will be small. Moreover, approximating WTP or WTA by consumer surplus—the area to the left of the Marshallian demand function will produce an error of no more than 5 percent in most cases (Robert Willig 1976).³⁴

One problem with the definitions of the value of a change in environmental quality (equations (10) and (11)) is that not all environmental benefits can be viewed as certain. Reducing exposure to a carcinogen, for example, alters the probability that persons in the exposed population will contract cancer, and it is this probability that must be valued.

To define the value of a quality change under uncertainty, suppose that the value of S associated with a given Q is uncertain. Specifically, suppose that two values of S are possible: S^0 and S^1 . For example, S^0 might be 360 healthy days per year and S^1 no healthy days (death). Q no longer determines S directly, but affects π , the probability that S^0 occurs. If the individual is an expected utility maximizer and if $V(M, S^i)$, $i = 0, 1$, denotes his expected utility in each state (M being income), willingness to pay for a change in Q from Q^0 to Q^1 is the most one can take away from the individual and leave him at his original expected utility level (Michael Jones-Lee 1974).

$$\begin{aligned} & \pi(Q^0)V(M, S^0) + [1 - \pi(Q^0)]V(M, S^1) \\ & = \pi(Q^1)V(M - WTP, S^0) \\ & + [1 - \pi(Q^1)]V(M - WTP, S^1). \end{aligned} \quad (12)$$

For a small change in Q , WTP is just the difference in utility between the two states, divided by the expected marginal utility of money,

³⁴ Sufficient conditions for this to hold are that (1) consumer surplus is no more than 90 percent of income; (2) the ratio of consumer surplus to income, multiplied by one-half the income elasticity of demand, is no more than 0.05.

$$WTP = \frac{[V(M, S^0) - V(M, S^1)]}{\pi V_M^0 + (1 - \pi)V_M^1} \cdot \frac{\partial \pi}{\partial Q} dQ. \quad (13)$$

An important point to note here is that the value of the change in Q is an ex ante value: changes in Q are valued before the outcomes are known. For example, suppose that reducing exposure to an environmental carcinogen is expected to save two lives in a city of 1,000,000 persons. The ex ante approach views this as a 2-in-one-million reduction in the probability of death for each person in the population. The ex post approach, by contrast, would value the reduction in two lives with certainty.

We are now in a position to discuss the principal methods that have been used to value changes in pollution.

B. Indirect Methods for Measuring the Benefits of Environmental Quality

Economists have employed three approaches to valuing pollution that rely on observed choices: the averting behavior approach, the weak complementarity approach, and the hedonic price approach.

1. *The Averting Behavior Approach.* The averting behavior approach relies on the fact that in some cases purchased inputs can be used to mitigate the effects of pollution.³⁵ For example, farmers can increase the amount of land and other inputs to compensate for the fact that ozone reduces soybean yields. Or, for another, residents of smoggy areas can take medicine to relieve itchy eyes and runny noses.

As long as other inputs can be used to compensate for the effects of pollution,

³⁵ In terms of the notation above, either (9) applies, or other inputs can be substituted for S in production; see equation (2).

the value of a small change in pollution can be measured by the value of the inputs used to compensate for the change in pollution. If, for example, a reduction in one-hour maximum ozone levels from 0.16 parts per million (ppm) to 0.11 ppm reduces the number of days of respiratory symptoms from 6 to 5, and if an expenditure on medication of \$20 has the same effect, then the value of the ozone reduction is \$20.

Somewhat more formally, if $S = S(Q, Z)$, willingness to pay for a marginal change in Q may be written as the marginal rate of substitution between an averting good and pollution, times the price of the averting good (Paul Courant and Richard Porter 1981).

$$WTP = -p_1 \frac{\partial S / \partial Q}{\partial S / \partial z_1}, \quad (14)$$

where z_1 is medication. Marginal WTP can thus be estimated from the production function alone.

To value a nonmarginal change in pollution, one must know both the cost function for the good affected by pollution and the marginal value function for that good. For example, in the case of health damages, a large improvement in air quality will shift the marginal cost of healthy days to the right (see Figure 1) and the value of the change is given by the area between the two marginal cost curves, bounded by the marginal value of healthy time. When the good in question is not sold in markets, as is the case for health, estimating the marginal value function is, however, difficult.³⁶

³⁶ If S were sold in markets, estimation of the marginal value function would be simple, assuming one could observe the price of S and assuming that the price was exogenous to any household. The problem is that, for a good produced by the household itself, one cannot observe the price (marginal cost) of the good—it must be estimated from the marginal cost function. Furthermore, the price is endogenous, since it depends on the level of S .

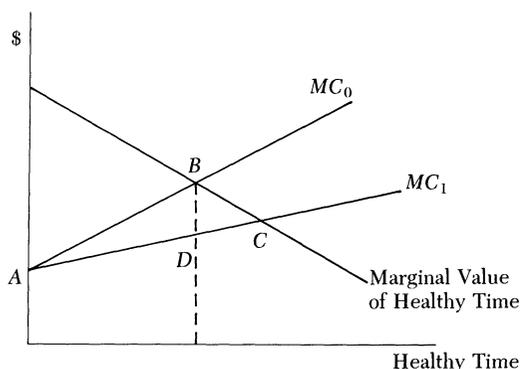


Figure 1. Morbidity Benefits of a Nonmarginal Pollution Reduction

An alternative approach, suggested by Bartik (1988a), is to use the change in the cost of producing the original level of S , i.e., the area between the marginal cost functions to the left of S^0 (area ABD in Figure 1), to approximate the value of the environmental quality change. For an improvement in Q , this understates the value of the change because it does not allow the individual to increase his chosen value of S . When the marginal cost of S increases, the relevant area will overstate the value of the welfare decrease. The advantage of this approximation is that it can be estimated from knowledge of the cost function alone.

The usefulness of the averting behavior approach is clearly limited to cases where other inputs can be substituted for pollution. Most pollution damages suffered by firms occur in agriculture, forestry, and fishing. In the case of agriculture, irrigation can compensate for the effects of global warming on crop yields. Likewise, capital (boats and gear) and labor can compensate for fish populations depleted as a result of water pollution.

In the case of pollution damages suffered by households, averting behavior has been used to value health damages and the soiling damages caused by air pollution. Households can avoid health damages either by avoiding exposure to

pollution in the first place, or by mitigating the effects of exposure once they occur. For example, the deleterious effects of water pollution can be avoided by purchasing bottled water (V. Kerry Smith and William Desvousges 1986b), and pollutants in outdoor air may be filtered by running an air-conditioner (Mark Dickie and Shelby Gerking 1991).

Two problems, however, arise in applying the averting behavior method in these cases. First, in computing the right-hand-side of (14), the researcher must know what the household imagined the benefit of purchasing water ($\partial S/\partial z_1$) to be, since it is the *perceived* benefits of averting behavior that the household equates to the marginal cost of this behavior. Second, when the averting input produces joint products, as in the case of running an air-conditioner, the cost of the activity cannot be attributed solely to averting behavior. Inputs that mitigate the effects of pollution include medicine and doctors' visits (Gerking and Linda Stanley 1986); however, use of the latter often runs into the joint product problem—a doctor's visit may treat ailments unrelated to pollution, as well as pollution related illness.

2. *The Weak Complementarity Approach.* While the averting behavior approach exploits the substitutability between pollution and other inputs into production, the weak complementarity approach values changes in environmental quality by making use of the complementarity of environmental quality, e.g., cleaner water, with a purchased good, e.g., visits to a lake. Suppose that a specified improvement in water quality at a lake resort results in an increase in a household's demand for visits to the resort from ED to AB (see Figure 2). One can view the value of access to the lake at the original quality level Q^0 as the value of being able to visit the lake at a cost of C rather than at some cost E .

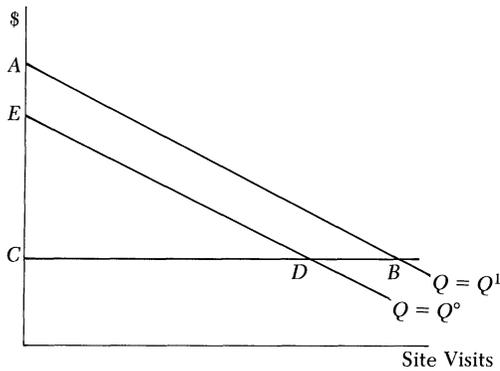


Figure 2. The Effect of a Change in Environmental Quality on the Demand for Visits to a Recreation Site

The value of access to the lake is thus the area EDC .³⁷ The increase in the value of access when Q changes (area $ABDE$) is the value of the water quality improvement.

For area $ABDE$ to measure the value of the water quality improvement, environmental quality must be weakly complementary to the good in question (Mäler 1974; Nancy Bockstael and Kenneth McConnell 1983). This means that (1) the marginal utility of environmental quality (water quality) must be zero if none of the good is purchased (no visits are made to the lake); (2) there is a price above which none of the good is purchased (no visits are made). If (1) did not hold, there would be additional benefits to a change in water quality not reflected in the demand for visits.

In practice, the weak complementarity approach has been used most often to value the attributes of recreation sites—either water quality, or a related attri-

bute, such as fish catch.³⁸ Although site visits do not have a market price, their cost can be measured by summing the cost of traveling to the site, including the time cost, as well as any entrance fees.

A problem in measuring the demand for site visits as a function of site quality is that there is no variation in site quality among persons who visit a site. A popular solution to this problem is the varying parameters model, which assumes that site quality enters recreation demand functions multiplied by travel cost or income, both of which vary across households.³⁹ In the first stage of the model, the demand for visits to site i is regressed on the cost of visiting the site and on income. In the second stage the coefficients from stage one are regressed on quality variables at site i . This is equivalent to estimating a set of demand functions in which visits to site i depend on the quality of the i th site, the cost of visiting the i th site, income, and interactions between travel cost and quality, and income and quality.

One drawback of this approach is that it allows visits to a given site to depend only on the cost of visiting that site—the cost of visiting substitute sites is not considered. This is equivalent to assuming that, except for the quality variables that enter the model in stage two, all sites are perfect substitutes. The varying parameters model may, therefore, give misleading results if one wishes to value quality changes at several sites.

A second approach to valuing quality changes is to use a discrete choice model. This approach examines the choice of

³⁷ Strictly speaking EDC should be measured using the consumer's compensated demand function. When measuring the value of access to a good, use of the Marshallian demand function may no longer provide a good approximation to the welfare triangle since the choke prices of the Marshallian and compensated demand functions may vary substantially. The Willig bounds do not apply in this case.

³⁸ Surveys of recreation demand models may be found in Mendelsohn (1987) and also in John Braden and Kolstad (1991). Bockstael, Hanemann, and Catherine Kling (1987) discuss their application to valuing environmental quality at recreation sites.

³⁹ This solution was first used by Vaughan and Russell (1982) and has also been used by V. Kerry Smith, Desvousges, and Matthew McGivney (1983), and V. Kerry Smith and Desvousges (1986a).

which site to visit on a given day as a function of the cost of visiting each site, and the quality of each site. If the choice of which site to visit on the first recreation day can be viewed as independent of which site to visit on the i th, a simple discrete choice model, such as the multinomial logit, can be applied to the choice of site, conditional on participation (Clark Binkley and Hanemann 1978; Daniel Feenberg and Mills 1980). The choice of whether to participate and, if so, on how many days, is made by comparing the maximum utility received from taking a trip with the utility of the best substitute activity on that day.⁴⁰

The advantage of the discrete choice model is that the probability of visiting any one site depends on the costs of visiting all sites and the levels of quality at all sites. The drawback of the model is that the decision to take a trip or not and, if so, which site to visit, is made independently on each day of the season. The number of trips made to date influence neither which site the individual chooses to go to on a given day, nor whether he takes a trip at all.⁴¹ Thus, these models must be combined with models that predict the total number of trips taken.

3. *Hedonic Market Methods.* The

⁴⁰ If one estimates a discrete choice model of recreation decisions, the value of a change in environmental quality at site i is no longer measured as indicated in Figure 2 (Hanemann 1984). Because utility is random from the viewpoint of the researcher, compensating variation for a change in quality at a recreation site on a given day equals the change in utility conditional on visiting the site times the probability that the site is visited, plus the change in the probability of visiting the site times the utility received from the site.

⁴¹ One solution to this problem, proposed by Edward Morey (1984), is to estimate a share model, which allocates the recreation budget for a season among different sites. The drawback of this model is that the share of the budget going to each site is assumed to be positive, whereas, in reality, a household may not visit all sites.

third method used by economists to value environmental quality, or a related output such as mortality risk, exploits the concept of hedonic prices—the notion that the price of a house or job can be decomposed into the prices of the attributes that make up the good, such as air quality in the case of a house (Ronald Ridker and John Henning 1967), or risk of death in the case of a job (Richard Thaler and Sherwin Rosen 1976). The hedonic price approach has been used primarily to value environmental disamenities in urban areas (air pollution, proximity to hazardous waste sites), which are reflected both in housing prices and in wages. It has also been used to value mortality risks by examining the compensation workers receive for voluntarily assuming job risks. Finally, the hedonic travel cost approach has been used to value recreation sites. We discuss each approach in turn.

Urban Amenities. Air quality and other environmental amenities can be valued in an urban setting by virtue of being tied to residential location: they are part of the bundle of amenities—public schools, police protection, proximity to parks—that a household purchases when buying a house.

The essence of the hedonic approach is to try to decompose the price of a house (or of residential land) into the prices of individual attributes, including air quality. This is done using an hedonic price function, which describes the equilibrium relationship between house price, p , and attributes, $A = (a_1, a_2, \dots, a_n)$. The marginal price of an attribute in the market is simply the partial derivative of the hedonic price function with respect to that attribute. In selecting a house, consumers equate their marginal willingness to pay for each attribute to its marginal price (S. Rosen 1974; A. Myrick Freeman 1974). This implies that

the gradient of the hedonic price function, evaluated at the chosen house, gives the buyer's marginal willingness to pay for each attribute.

Somewhat more formally, utility maximization in an hedonic market calls for the marginal price of an attribute to equal the household's marginal willingness to pay for the attribute,

$$\partial p / \partial a_i = \partial \theta / \partial a_i, \quad (15)$$

where θ is the household's bid function, the most one can take away from the household in return for the collection of amenities, A , and keep its utility constant. Equation (15) implies that, in equilibrium, the marginal willingness to pay for an attribute can be measured by its marginal price, computed from the hedonic price function.

If a large improvement in environmental quality is contemplated in one section of a city—an improvement large enough to alter housing prices—the derivative of the hedonic price function no longer measures the value of the amenity change. In the short run, before households adjust to the amenity change and prices are altered, the value of the amenity change is the area under the household's marginal bid function—the right hand side of (15)—between the old and new levels of air quality. To value the amenity change in the long run, however, one must take into account the household's adjustment to the amenity change *and* to any price changes that may result. The area under the marginal bid function (the short-run welfare measure) is, however, a lower bound to the long-run benefits of the amenity change (Bartik 1988b).

Empirical applications of the hedonic approach have typically focused either on valuing marginal amenity changes, which requires estimating only the hedonic price function, or on computing the short-run benefits of nonmarginal amen-

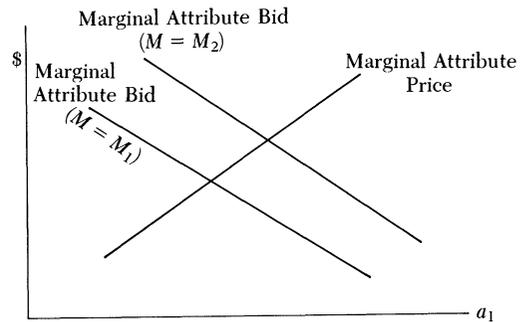


Figure 3. The Identification Problem in an Hedonic Market

ity changes, which requires estimating marginal bid functions. S. Rosen originally suggested that this be done by regressing marginal attribute price, computed from the gradient of the hedonic price function, on the arguments of the marginal bid function. This procedure, however, may encounter an identification problem which is caused by the fact that the arguments of the marginal attribute bid function determine marginal attribute price as well.

An example of the identification problem, provided by James Brown and Harvey Rosen (1982), occurs when the hedonic price function is quadratic and the marginal value functions are linear in attributes. In the case of a single amenity, a_1 ,

$$\partial p / \partial a_1 = \beta_0 + \beta_1 a_1 \quad (16)$$

$$\partial \theta / \partial a_1 = b_0 + b_1 a_1 + b_2 M. \quad (17)$$

In this case regressing $\beta_0 + \beta_1 a_1$ on a_1 and M will reproduce the parameters of the marginal price function, i.e., $\hat{b}_0 = \beta_0$, $\hat{b}_1 = \beta_1$ and $\hat{b}_2 = 0$. This is illustrated graphically in Figure 3. The problem is that the marginal price function does not shift independently of the marginal bid function. Shifts in the latter, due, say, to differences in income, thus trace out points on the marginal price function.

To achieve identification in this ex-

ample, one can introduce functional form restrictions, such as adding a_1^2 to the marginal price function, but not to the marginal value function, which will cause $\partial p/\partial a_i$ to shift independently of $\partial\theta/\partial a_i$ (Mendelsohn 1984). Another solution is to estimate hedonic price functions for several markets, so that the coefficients of the marginal price function vary across cities (Palmquist 1984; Robert Ohsfeldt and Barton Smith 1985; Ohsfeldt 1988). For this to work, households in all cities must have identical preferences; however, the distribution of measured household characteristics and/or the supply of amenities must vary across cities so that the hedonic price function and its gradient vary from one city to another. In the case of several a_i 's, one can impose exclusion restrictions on the a_i 's that enter each marginal value function (Dennis Epple 1987) so that marginal prices vary independently of the variables that enter the marginal value function.

In view of the problems in estimating marginal attribute bid functions, it is important to note that an upper bound to the long-run benefits of an amenity improvement can be obtained from the hedonic price function alone. Yoshitsugu Kanemoto (1988) has shown that the change in prices in the improved area predicted by the hedonic price function is an upper bound to the long-run benefits of an amenity improvement. Thus, from knowledge of the hedonic price function alone one can obtain (1) the exact value of a marginal attribute change, and (2) an upper bound to the long-run value of an attribute change.

Wage-Amenity Studies. The analysis of hedonic housing markets, by focusing on housing market equilibrium within a city, implicitly ignores migration among cities. If one takes a long-run view and assumes that workers can move freely from one city to another, then data on

compensating wage differentials across cities can be used to infer the value of environmental amenities (Glenn Blomquist, Mark Berger, and John Hoehn 1988; Maureen Cropper and Amalia Arriaga-Salinas 1980; V. Kerry Smith 1983). Intuitively, the value people attach to urban amenities should be reflected in the higher wages they require to live in less desirable cities.

When migration is possible, consumers choose the city in which they live to maximize utility; however, wage income, as well as amenities, vary from one city to another (S. Rosen 1979; Jennifer Roback 1982).⁴² Household equilibrium requires that utility be identical in all cities.

The fact that consumers in all cities must enjoy the same level of utility implies that wages and land rents must adjust to compensate for amenity differences. The marginal value of an amenity change to a consumer is thus the sum of the partial derivatives of an hedonic wage function and an hedonic property value function (Roback 1982).

Hedonic Labor Markets. The fact that risk of death is a job attribute traded in hedonic labor markets has provided economists with an alternative to the averting behavior approach as a means of valuing mortality risk (Thaler and S. Rosen 1976). The theory behind this approach is simple: other things equal, workers in riskier jobs must be compensated with higher wages for bearing this risk. As in the case of hedonic housing markets, the worker chooses his job by equating the marginal cost of working in a less risky job—the derivative of the hedonic price function—to the marginal benefit, the value (in dollars) of the resulting increase in life expectancy.

There are three problems in using the compensating wage approach. One is

⁴² In most models wages, lot size, and amenities vary among, but not within, cities.

that compensating wage differentials exist only if workers are informed of job risks. Thus, the absence of compensating differentials need not mean that workers do not value reducing the risk of death. A second problem is that compensating differentials appear to exist only in unionized industries (William Dickens 1984; Douglas Gegax, Gerking, and Schulze 1985). This suggests that the wage differential approach may provide estimates of the value of a risk reduction only for certain segments of the population. This problem is compounded by the fact that the least risk averse individuals work in risky jobs. Third, if workers have biased estimates of job risks, or if the objective measures of job risk used in most wage studies over- or understate workers' risk perceptions, market wage premia will yield biased estimates of the value of a risk reduction.

The Hedonic Travel Cost Approach. Yet another area in which the hedonic approach has been applied is in valuing the attributes of recreation sites (G. Brown and Mendelsohn 1984). In valuing sites, the analog to the hedonic price function is obtained by regressing the cost of travelling to a recreation site on the attributes of the site, such as expected fish catch, clarity of water, and water color. However, because this relationship is not the result of market forces, there is nothing to guarantee that the marginal cost of an attribute is positive. More desirable sites may be located closer to population centers rather than farther away from them.⁴³ In this case, the individual's choice of site will not be described by (13), and care must be taken when inferring values from marginal attribute costs (V. Kerry Smith, Palmquist, and Paul Jankus 1990).

⁴³ The problem may be reduced by using only sites actually visited from a given origin in estimating the hedonic travel cost function.

C. *The Contingent Valuation Method*

While the indirect market approaches we have described above can be used to value many of the benefits of pollution reduction, there are important cases in which they cannot be used. When no appropriate averting or mitigating behavior exists, indirect methods cannot be used to estimate the morbidity benefits of reducing air pollution. Recreation benefits may be difficult to measure since there may not be enough variation in environmental quality across sites in a region to estimate the value of water quality using the travel cost approach.

There is, in addition, an entire category of benefits—*nonuse values*—which cannot even in principle be measured by indirect market methods. Nonuse values refer to the benefits received from knowing that a good exists, even though the individual may never experience the good directly. Examples include preserving an endangered species or improving visibility at the Grand Canyon for persons who never plan to visit the Grand Canyon.

This suggests that direct questioning can play a role in valuing the benefits of pollution control. Typically, direct questioning or contingent valuation studies ask respondents to value an output, such as a day spent hunting or fishing, rather than a change in pollution concentrations per se. Examples of commodities that have been valued using the contingent valuation method (CVM) include improvements in water quality to the point where the water is fishable or swimmable (Richard Carson and Robert Mitchell 1988), improvements in visibility resulting from decreased air pollution (Alan Randall, Berry Ives, and Clyde Eastman 1974; Schulze and David Brookshire 1983; Decision Focus 1990), the value of preserving endangered species (James Bowker and John Stoll 1988;

Kevin Boyle and Richard Bishop 1987), and days free of respiratory symptoms (George Tolley et al. 1986b; Dickie et al. 1987).

Any contingent valuation study must incorporate (1) a description of the commodity to be valued; (2) a method by which payment is to be made; and (3) a method of eliciting values. In studies that value recreation-related goods, hypothetical payment may take the form of a user fee or an increase in taxes; in the case of improved visibility, a charge on one's utility bill, since power plant pollution can contribute to air quality degradation. To determine the maximum a person is willing to pay for an improvement in environmental quality, the interviewer may simply ask what this amount is (an open-ended survey), or he may ask whether or not the respondent is willing to pay a stated amount (a closed-ended survey). The yes/no answer does not yield an estimate of each respondent's willingness to pay; however, the fraction of respondents willing to pay at least the stated amount gives a point on the cumulative distribution function of willingness to pay for the commodity (Trudy Cameron and Michelle James 1987).

There seems to be general agreement that closed-ended questions are easier for respondents to answer and therefore yield more reliable information than open-ended questions, especially when the commodity valued is not traded in conventional markets. Asking an open-ended question about a good that respondents have never been asked to value, such as improved visibility, often yields a distribution of responses that has a large number of zero values and a few very large ones. This may reflect the fact that respondents have nothing to which to anchor their responses, and are unwilling to go through the reasoning necessary to discover the value they place on the good. Answering a yes/no question is, by

contrast, a much easier task, and one that parallels decisions made when purchasing goods sold in conventional markets.

It must be acknowledged that, despite advances made in contingent valuation methodology during the last 15 years, many remain skeptical of the method. Perhaps the most serious criticism is that responses to contingent valuation questions are hypothetical—they represent professed, rather than actual, willingness to pay. This issue has been investigated in at least a dozen studies that compare responses to contingent valuation questions with actual payments for the same commodity.

How close hypothetical values are to actual ones depends on whether the commodity is a public or private good, on the elicitation technique used, and on whether it is willingness to pay (*WTP*) for the good or willingness to accept compensation (*WTA*) that is elicited. Most experiments comparing hypothetical and actual *WTP* for a private good (strawberries or hunting permits) have found no statistically significant difference between mean values of hypothetical and actual willingness to pay (Dickie, Ann Fisher, and Gerking 1987; Bishop and Thomas Heberlein 1979; Bishop, Heberlein, and Mary Jo Kealy 1983). Such is not the case when hypothetical and actual *WTA* are compared. In three experiments involving willingness to accept compensation for hunting permits, Bishop and Heberlein (1979) and Bishop, Heberlein, and Kealy (1983) found that actual *WTA* was statistically significantly lower than hypothetical *WTA* in two out of three cases. Hypothetical and actual *WTP* have also been found to differ when the commodity valued is a public good (Kealy, Jack Dovidio, and Mark L. Rockel 1987).

Other criticisms of the CVM have focused on: (1) the possibility that individuals may behave strategically in answering

questions—either overstating *WTP* if this increases the likelihood that an improvement is made, or understating *WTP* if it reduces their share of the cost (the free-rider problem); (2) the fact that individuals may not be sufficiently familiar with the commodity to have a well-defined value for it; and (3) the fact that *WTP* for a commodity is often an order of magnitude less than willingness to accept (*WTA*) compensation for the loss of the commodity.

The possibility that respondents behave strategically has been tested in laboratory experiments by examining whether announced *WTP* for a public good varies with the method used to finance the public good. Studies by Bohm (1972), Bruce Scherr and Emerson Babb (1975), and Vernon Smith (1977, 1979) suggest that strategic behavior is not a problem, possibly because of the effort that effective strategic behavior requires.

If the commodity to be valued is not well understood, contingent valuation responses are likely to be unreliable: responses tend to exhibit wide variation, and respondents may even prefer less of a good to more! One interpretation of this result is that people really do not have values for the commodity in question—they are created by the researcher in the course of the survey (Thomas Brown and Paul Slovic 1988). This is a serious criticism: Do people really know enough about groundwater contamination or biodiversity to place a value on either good?

Fortunately, it is possible to defend against this criticism by seeing how responses vary with the amount of information that is provided about the commodity being valued. If values are well defined, they should not, on average, vary with small changes in the amount of information.

One of the most striking and challenging findings emerging from this work is

that willingness to pay for an environmental improvement is usually *many times lower* than willingness to accept compensation to forego the same improvement (Judd Hammack and G. Brown 1974; Bishop and Heberlein 1979; Robert Rowe, d'Arge, and Brookshire 1980; Jack Knetsch and J. A. Sinden 1984). This is sometimes interpreted as evidence that the method of eliciting responses is unsatisfactory; however, as we noted above, there is no reason why *WTA* for a quality (public good) decrease should not exceed *WTP* for an increase of the same magnitude, provided that there are few substitutes for the public good.⁴⁴ An alternative explanation for the *WTA/WTP* discrepancy that has been offered by some economists (Donald Coursey, John Hovis, and Schulze 1987; Brookshire and Coursey 1987) is that individuals are simply not as familiar with the sale of an item as with its purchase. These authors find that, in experiments where individuals were allowed to submit bids or offers for the same commodity, *WTA* approached *WTP* after several rounds of transactions.⁴⁵

D. Applications of Valuation Techniques

Having described the main techniques used to value environmental amenities, we now wish to give the reader a feel for the way in which these

⁴⁴The explanation of the discrepancy between *WTA* and *WTP* offered by psychologists—that monetary losses from some reference point are valued more highly than monetary gains (Daniel Kahneman and Amos Tversky 1979)—also suggests that this disparity has nothing to do with flaws in the contingent valuation method.

⁴⁵None of these explanations, however, seems to account for results obtained by Kahneman, Knetsch, and Thaler (1990). They find that, even for common items such as coffee mugs and ballpoint pens, sellers have reservation prices that are higher, much higher on average, than buyers' bid prices. This disparity does not disappear after several rounds of trading. The initial distribution of property rights (the "endowment effect") may, therefore, matter, even for goods with many substitutes.

TABLE 1
TOTAL ANNUALIZED ENVIRONMENTAL COMPLIANCE COSTS, BY MEDIUM, 1990
(Millions of 1986 dollars)

<i>Medium</i>	<i>Costs</i>	<i>Major Statutes</i>
Air and Radiation, Total	28,029	
Air	27,588	Clean Air Act (CAA)
Radiation	441	Radon Pollution Control Act
Water, Total	42,410	
Water Quality	38,823	Clean Water Act (CWA)
Drinking Water	3,587	Safe Drinking Water Act
Land, Total	26,547	
RCRA	24,842	Resource Conservation and Recovery Act (RCRA)
Superfund	1,704	Comprehensive Environmental Response, Compensation and Liability Act (CERCLA)
Chemicals, Total	1,579	
Toxic Substances	600	Toxic Substances Control Act (TSCA)
Pesticides	979	Federal Insecticide, Fungicide and Rodenticide Act (FIFRA)
Total Costs	100,167	

Note: These represent the costs of complying with all federal pollution control laws, assuming full implementation of the law (USEPA 1990).

techniques have been used to value the benefits of pollution control. We shall begin with an overview of the types of benefits associated with the major pieces of environmental legislation. We then turn to a description and assessment of actual benefit estimation.

Table 1 lists the major pieces of environmental legislation in the U.S. and the estimated costs of complying with each statute in 1990. With the exception of the Clean Water Act, the primary goal of U.S. environmental legislation is to protect the health of the population. According to the Clean Air Act, ambient standards for the criteria air pollutants are to be set to protect the health of the most sensitive persons in the population.⁴⁶ The goal of the Safe Drink-

ing Water Act is, similarly, to provide a margin of safety in protecting the country's drinking water supplies from toxic substances, while the goal of the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) is to prevent adverse effects to human health and to the environment from the use of pesticides.

Each of the statutes in Table 1 also results in certain nonhealth benefits. The Clean Air Act provides important aesthetic benefits in the form of increased visibility, and the 1990 Amendments to the Act, designed to reduce acid rain, may yield ecological and water quality benefits. The Clean Water Act—whose goal is to make all navigable water bodies fishable and swimmable—yields recreational and ecological benefits. Both Acts yield benefits to firms in agriculture, forestry, and commercial fishing. FIFRA, the primary law governing pesticide us-

⁴⁶ The criteria air pollutants are particulate matter, sulfur oxides, nitrogen oxides, carbon monoxide, lead, and ozone.

age, is designed to protect animal as well as human health.

In addition to the pollution problem addressed by the major environmental statutes, there is increasing concern about the effects of emissions of greenhouse gases, including carbon dioxide, chlorofluorocarbons (CFCs) and methane. Studies suggest that emissions of these gases may contribute to increases in mean temperature, especially in the Northern Hemisphere, changes in precipitation, and sea level rises that could average 65 cm by the end of the next century. The main effects of these changes are likely to be felt in agriculture, in animal habitat, and in human comfort.

In light of the preceding discussion, we review empirical work for four categories of nonmarket benefits: health, recreation, visibility, and ecological benefits. We also discuss the benefits of pollution control to agriculture.

1. *The Health Benefits of Pollution Control.* The statutes listed in Table 1 contribute to improved human health in several ways. By reducing exposure to carcinogens—in the air, in drinking water, and in food—environmental legislation reduces the probability of death at the end of a latency period—the time that it takes for cancerous cells to develop. Mortality benefits are also associated with control of noncarcinogenic air pollutants, which reduces mortality especially among sensitive persons in the population, e.g., angina sufferers or persons with chronic obstructive lung disease. Lessening children's exposure to lead in gasoline or drinking water avoids learning disabilities and other neurological problems associated with lead poisoning. Finally, controlling air pollution reduces illness—ranging from minor respiratory symptoms associated with smog (runny nose, itchy eyes) to more serious respiratory infections, such as pneumonia and influenza. Water borne disease (e.g.,

giardiasis) may also cause acute illness.

Reductions in risk of death have been valued using three methods: averting behavior, hedonic analysis, and contingent valuation. The most common approach to valuing changes in risk of death due to environmental causes is hedonic wage studies. The results of these studies are typically expressed in terms of the value per "statistical life" saved. If reducing exposure to some substance reduces current probability of death by 10^{-5} for each of 200,000 persons in a population, it will save two statistical lives ($10^{-5} \times 200,000$). If each person is willing to pay \$20 for the 10^{-5} risk reduction, then the value of a statistical life is the sum of these willingnesses to pay ($\$20 \times 200,000$), divided by the number of statistical lives saved, or \$2,000,000.

Recent compensating wage studies (Ann Fisher, Daniel Violette, and Lauraine Chestnut 1989) generate mean estimates of the value of a statistical life that fall within an order of magnitude of one another: \$1.6 million to \$9 million (\$1986), with most studies yielding mean estimates between \$1.6 million and \$4.0 million. Contingent valuation studies that value reductions in job-related risk of death (Gerking, Menno DeHaan, and Schulze 1988) or reductions in risk of auto death (Jones-Lee, M. Hammerton, and P. R. Philips 1985) fall in the same range.

Averting behavior studies—based on seat belt use (Blomquist 1979) or the use of smoke detectors (Rachel Dardis 1980)—yield estimates of the value of a statistical life that are an order of magnitude lower than the studies cited above. These studies, however, estimate the value of a risk reduction for the person who just finds it worthwhile to undertake the averting activity. This is because buckling a seat belt or purchasing a smoke detector are 0-1 activities. They are undertaken provided that their marginal benefit equals or exceeds their marginal cost, with equality of marginal ben-

efit and marginal cost holding only for the marginal purchaser. If 80 percent of all persons use smoke detectors, the value of the risk reduction to the marginal purchaser may be considerably lower than the mean value.

There are, however, other problems in using the indirect market approaches we have reviewed here to value changes in environmental risks. One problem is that the risks valued in labor market and averting behavior studies are more voluntary than many environmental risks. Work by Slovic, Baruch Fischhoff, and Sarah Lichtenstein (1980, 1982) suggests that willingness to pay estimates obtained in one context may not be transferable to the other. Second, death due to an industrial accident is often instantaneous, whereas death resulting from environmental contaminants may come from cancer and involve a long latency period. Deaths due to cancer thus occur in the future and cause fewer years of life to be lost than deaths in industrial accidents. At the same time, however, cancer is one of the most feared causes of death.

In a study designed to value reductions in chemical contaminants (trihalomethanes) in drinking water, Mitchell and Carson (1986) found that the former effect seems to be important: the value of a statistical life associated with a reduction in risk of death 30 years hence was only \$181,000 (\$1986). This is lower than the value of a statistical life associated with current risk of death for two reasons: (1) the number of expected life years lost is smaller if the risk occurs 20 years hence, and (2) the individual may discount the value of future life years lost (Cropper and Frances Sussman 1990; Cropper and Paul Portney 1990).

In spite of these difficulties, valuing mortality risks is an area in which economists have made important contributions. The notion that, *ex ante*, individuals are willing to spend only a certain

amount to reduce risks to life makes possible rational debate and analysis in the policy arena over tradeoffs in risk reduction. Moreover, estimates of the value of a statistical life are in sufficiently close agreement to permit their use in actual benefit-cost calculations (subject, perhaps, to some sensitivity analysis).

The valuation of morbidity has been less successful. Estimates of the value of reductions in respiratory symptoms come from two sources: averting behavior studies and contingent valuation studies. The averting behavior approach has been used to value illnesses associated with both water and air pollution. It has been more successful in the case of water pollution because an averting behavior exists (buying bottled water) that is closely linked to water pollution (Abdalla 1990; Harrington, Krupnick, and Walter Spofford 1989). By contrast, the averting behaviors used to value air pollution—running an air-conditioner in one's home or car—are in most cases not undertaken primarily because of pollution. The use of doctor visits (purpose unspecified) to mitigate the effects of air pollution suffers from a similar shortcoming.

Contingent valuation studies of respiratory symptoms (coughing, wheezing, sinus congestion) have encountered two problems. The first concerns what is to be valued. Ideally, one would like to value a change in air pollution which, after defensive behavior is undertaken, might cause a change in the level of the symptom experienced. The individual's willingness to pay for the pollution change includes the value of the change in illness after mitigating behavior is undertaken, plus the cost of the mitigating behavior. This suggests that a symptom day be valued after mitigating actions have been taken. A second problem is that the respondent must be encouraged to consider carefully his budget constraint. Failure to handle these problems has led to unbelievably high average

values of a symptom day. In more careful studies, mean willingness to pay to eliminate one day of coughing range from \$1.39 (\$1984) (Dickie et al. 1987) to \$42.00 (\$1984) (Edna Loehmann et al. 1979); for a day of sinus congestion \$1.88 (Dickie et al.) to \$52.00 (Loehmann et al.).

An alternative approach to valuing morbidity is to use the cost of illness—the cost of medical treatment plus lost earnings—which, as Harrington and Portney (1976) have shown, is a lower bound to willingness to pay for the change in illness. Mean willingness to pay for symptom reduction is usually three to four times higher than the traditional cost of illness. Berger et al. (1987) report a mean *WTP* of \$27 to eliminate a day of sinus congestion, compared with an average cost of illness of \$7. The corresponding figures for throat congestion are \$44 and \$14.

Studies of willingness to pay to reduce the risk of chronic disease are few (W. Kip Viscusi, Magat, and Joel Huber 1988, is a notable exception), and cost of illness estimates are more prevalent in valuing chronic illness (Ann Bartel and Paul Taubman 1979; Barbara Cooper and Dorothy Rice 1976). Viscusi, Magat, and Huber estimate the value of a statistical case of chronic bronchitis to be \$883,000, approximately one-third of the value of a statistical life. This may be contrasted with cost of illness estimates of \$200,000 per case of chronic lung disease (Cropper and Krupnick 1989).

As the preceding discussion indicates, more work is needed in the area of both morbidity and mortality valuation. Because of the difficulty in finding activities that mitigate the effects of air pollution, contingent valuation studies would seem to be a more promising approach to valuing morbidity. If new studies are done, they should value combinations of symptoms rather than individual symptoms, since pollution exposures often trigger

multiple symptoms, and since the value of jointly reducing several symptoms is generally less than the sum of the values of individual symptom reductions. In the case of mortality risks, more refined estimates are needed that take into account the timing of the risk, the degree of voluntariness, and the cause of death. The timing issue is especially crucial here: the benefits of environmental programs to reduce exposure to carcinogens, such as asbestos, are not realized until the end of a latency period—perhaps 40 years in the case of asbestos. Since the exposed population is 40 years older, fewer life-years are saved, compared with programs that save lives immediately.⁴⁷

2. *The Recreation Benefits of Pollution Control.* Reductions in water pollution may enhance the quality of recreation experiences by allowing (or improving) swimming, boating, or fishing. Most studies of the recreation benefits of water pollution control have focused on fishing-related benefits, and it is on them that we concentrate our attention.

Travel cost studies have taken one of three approaches to valuing the fishing benefits of improved water quality. In some studies (V. Kerry Smith and Desvousges 1986a), measures of water quality such as dissolved oxygen are valued directly. That is, water quality variables directly enter equations that describe the choice of recreation site or demand functions for site visits.⁴⁸ This approach is clearly useful if one wishes to link the valuation study to pollution control poli-

⁴⁷ While some studies have attempted to take the latency period and number of life-years saved into account (Josephine Mauskopf 1987), this is not the general practice (Cropper and Portney 1990).

⁴⁸ This approach is also used when the recreation activity studied is swimming or viewing, activities where perceptions of water quality are likely to be linked to water clarity and odor. It has, for example, been applied in studies of beach visits in Boston (Bockstael, Hanemann, and Kling 1987) and lake visits in Wisconsin (George Parsons and Kealy 1990).

cies, such as policies to reduce biochemical oxygen demand (BOD), a measure of the oxygen required to neutralize organic waste. A second approach is to relate site visits (or choice of site) to fish catch. Fish catch is clearly more closely associated with motives for visiting a site than is dissolved oxygen; however, it must be linked to changes in the fish population, which must, in turn, be linked to changes in ambient water quality.

A third approach is to treat changes in water quality as effectively eliminating or creating recreation sites. This approach has been used in valuing the effects of acid rain on fishing in Adirondack lakes: reductions in pH below certain thresholds have been treated as eliminating acres of surface area for fishing of particular species (John Mullen and Frederic Menz 1985). It is also the approach used by Vaughan and Russell (1982) in valuing the benefits of the Clean Water Act. They treat the benefits of moving all point sources to the Best Practical Control Technology Currently Available (BPT) as an increase in the number of acres of surface water that support game fish (bass, trout) as opposed to rough fish (carp, catfish). The Clean Water Act is thus viewed as increasing the number of recreation sites, rather than raising fish catch at existing sites.

Regardless of the form of water recreation valued, an improvement in water quality has two effects: it increases the utility of people who currently use the resource, and it may increase participation rates (number of days spent fishing). Varying parameter models that value changes in water quality or fish catch using the shift in demand for site visits (see Figure 2) capture both effects. Discrete choice models measure the effect of a quality improvement on a given recreation day, but do not estimate the effect of quality changes on the total number of days spent fishing; however, these

models are typically used in conjunction with models that predict the total number of trips. Treating changes in water quality as altering the supply of available sites captures participation effects but not improvements in quality at existing sites.

In addition to travel cost models, contingent valuation studies have been used to value improvements in fish catch or water quality. Because it is difficult to ask consumers to value changes in dissolved oxygen levels or fecal coliform count—another measure of water quality—without linking these water quality measures to the type of activities they support, many CVM studies use the RFF Water Quality Ladder (Vaughan and Russell 1982), which relates a water quality index to the type of water use—boating, fishing (rough fish), fishing (game fish), swimming—that can be supported by various levels of the index. It is these activity levels that are valued by respondents. The water quality ladder has been used both to value water quality at specific sites (e.g., the Monongahela River, by V. Kerry Smith and Desvousges 1986a) and at all sites throughout the country (Carson and Mitchell 1988).

It is interesting to compare estimates of the value of water quality improvements obtained by the travel cost and contingent valuation approaches. Carson and Mitchell (1988) report that households are, on average, willing to pay \$80 per year (in 1983 dollars) for an improvement in water quality throughout the U.S. from boatable to fishable (capable of supporting game fish). V. Kerry Smith and Desvousges (1986a) report a mean value of \$25 per household for the same improvement in a five-county region in western Pennsylvania. The difference between these estimates reflects the fact that non-use values are important: households care about clean water in areas where they do not live. Even the \$25 estimate for western Pennsylvania re-

flects nonuse values, since only one-third of the households surveyed engaged in some form of water based recreation.

Because they do not capture nonuse values, travel cost estimates of the value of improving water quality are not directly comparable with those obtained using the CVM. Using a varying parameter model, V. Kerry Smith and Desvousges (1986a) find the value of an improvement in water quality from boatable to fishable to be between \$0.06 and \$30.00 per person per day (\$1983) for 30 Army Corps of Engineers sites. This value may be contrasted with estimates of \$5 to \$10 per person per day (\$1983) obtained by Vaughan and Russell.

The preceding discussion suggests two problems that arise in valuing water quality benefits that do not arise in valuing health effects. The first is an aggregation problem. Suppose that one wishes to value the benefits of water quality improvements in a river basin, and suppose that the travel cost approach is used to measure use values associated with an improvement in dissolved oxygen or fish catch. The nonuse values associated with these improvements could be measured using a contingent valuation study. However, while the responses of nonusers could be added to values obtained from the travel cost approach, it would, in practice, be hard to separate use from nonuse values in the responses of fishermen.

The second problem is one of transferring results from a water quality study done in one geographic area to another area. While one can easily control for differences in willingness to pay in the two regions associated with differences in income and population, the value of water quality improvements is also likely to vary with the particular aesthetic and other characteristics of the region—and such characteristics are intrinsically hard to measure. Thus, whereas one can value

a day of coughing independently of location, it is harder to value a generic fishing day.

This raises important questions concerning priorities for research in the area of recreation benefits.⁴⁹ Future research can proceed using a contingent valuation approach in which use and nonuse values are elicited simultaneously for sites in the respondent's region. The problem here is to have the respondent value an improvement to recreation that is sufficiently specific that it can be related to changes in pH levels from acid rain or changes in levels of dissolved oxygen associated with the adoption of BPT. The advantage of this approach is that it would capture both use and nonuse values. The advantage of the travel cost approach is that it could use endpoints more closely related to pollution (such as dissolved oxygen); however, it would not yield estimates of nonuse values.

3. *The Visibility Benefits of Pollution Control.* Reductions in air pollution, by increasing visibility, may improve the quality of life in urban areas as well as at recreation sites. Since the number of persons affected by improvements in visibility is large—at least as great as the number of persons whose health is affected by air pollution—the potential value of such benefits is great.

One can view the results of hedonic property value studies performed in the 1970s and early 1980s as evidence that people value the visibility benefits of pollution control. In these studies housing prices were regressed on measures of ambient air quality such as particulates or sulfates, which are negatively correlated

⁴⁹ It should be emphasized that, while there exist several dozen studies of water quality benefits in a recreation context, many studies analyze the same data. Thus, empirical estimates of water quality benefits exist for only a few areas of the country—lakes in Wisconsin and the Adirondacks, beaches in Boston and on the Chesapeake Bay, recreation sites in western Pennsylvania.

with visibility. The studies, most of which found significant negative effects of air pollution on housing prices, thus provide indirect evidence that people are willing to pay for improved visibility.⁵⁰ For example, John Trijonis et al. (1984) estimated based on differences in housing prices that households in San Francisco were willing, on average, to pay \$200 per year for a 10 percent improvement in visibility.

The difficulty in using these studies to estimate benefits, however, is that the coefficient of air pollution (or visibility) captures all reasons why households may prefer to live in nonpolluted areas—including both improved health and reduced soiling. Indeed, the reason why property value studies have become less popular as a method of valuing the benefits of pollution control is that it is difficult to know what the pollution coefficient captures and, therefore, difficult to aggregate benefit estimates obtained from these studies with those obtained from other approaches. Such aggregation is necessary because residential property value studies capture benefits only at home and not at the other locations the household frequents.

For these reasons contingent valuation seems the most promising method for valuing visibility. Because visibility benefits vary regionally, CVM studies can most usefully be classified according to whether they measure urban visibility benefits or benefits at recreation sites, and according to whether the locations studied are in the Eastern or in the Western United States. The former distinction is important because visibility benefits at recreation sites—especially national parks—are likely to have a substantial nonuse component; consequently, the relevant population for which benefits

are computed may be considerably larger than for urban visibility benefits. The East/West distinction is important both because of differences in baseline visibility and because of qualitative differences in the nature of visibility impairments, e.g., haze versus brown cloud.

There are two key problems in any contingent valuation study of visibility. One is presenting changes in visibility that are both meaningful to the respondent and that can be related to pollution control policies. The other is separating the respondent's valuation of health effects from his valuation of visibility changes.

Most CVM studies define increased visibility as an improvement in visual range—the distance at which a large, black object disappears from view. Visual range is both correlated with people's perceptions of visibility and with ambient concentrations of certain pollutants (fine nitrate and sulfate aerosols). Differences in visual range are presented in a series of pictures in which all other conditions—weather, brightness, the objects photographed—are, ideally, kept constant.

It has long been recognized (Brookshire et al. 1979) that, in responding to such pictures, people assume that the health effects of pollution diminish as visibility improves. Health effects are therefore inherently difficult to separate from visibility changes. The best way to handle this problem is to ask respondents what they assume health effects to be and then to control for these effects.

Unfortunately, existing CVM studies of visibility benefits—especially those for urban areas—have failed to treat the issues raised above in a satisfactory manner. With this limitation in mind, it is nonetheless of interest to contrast the magnitude of benefits associated with improvements in urban air quality with estimates obtained from hedonic property

⁵⁰ Freeman (1979a) provides an excellent summary of early studies.

value studies. Studies of visibility improvements in eastern U.S. cities (Tolley et al. 1986a; Douglas Rae 1984) have estimated that households would pay approximately \$26 annually for a 10 percent improvement in visibility.⁵¹ Loehmann Boldt, D., and Chaikin, K. (1981) reports an annual average willingness to pay per household of \$101 for a 10 percent improvement in visibility in San Francisco. Both figures are considerably lower than estimates implied by property value studies.

Studies in recreation areas have focused on major national parks, including the Grand Canyon (Decision Focus 1990; Schulze and Brookshire 1983), because of the possibility of large nonuse values attached to visibility benefits at these sites. Two conclusions emerge from these studies. First, nonuse values appear to be large relative to use values. Use values associated with an improvement in visibility at the Grand Canyon from 70 to 100 miles are under \$2.00 per visitor party per day (\$1988) (Schulze and Brookshire 1983; K. K. MacFarland et al. 1983). By contrast, Schulze and Brookshire found that a random sample of households were willing to pay \$95 per year (\$1988) to prevent a deterioration in visibility at the Grand Canyon from the 50th percentile to the 25th percentile.

Second, the embedding, or superadditivity, problem is potentially quite serious. This refers to the fact that, in general, an individual's willingness to pay for simultaneous improvements in visibility at several sites should be less than the sum of his willingness to pay for isolated improvements at each site (Hoehn and Randall 1989). In a follow-up study to Schulze and Brookshire (1983), Tolley

et al. (1986a) found respondents were willing to pay only \$22 annually for the same visibility improvement at the Grand Canyon when this was valued at the same time as visibility improvements in Chicago (the site of the interviews) and throughout the East coast.

4. *The Ecological Benefits of Pollution Control.*⁵² By the ecological benefits of pollution control, we mean reduced pollution of animal and plant habitats, such as rivers, lakes, and wetlands. Because the benefits of clean water to recreational fisherman or larger populations of deer to hunters are captured in recreation studies, the benefits discussed in this section are the nonuse benefits associated with reduced pollution of ecosystems.

It should be clear to the reader that valuing this category of benefits poses serious conceptual problems. One is defining the commodity to be valued. Does one value reductions in pollution concentrations, increases in animal populations, or some more subtle index of the health of an ecosystem? Two approaches can be taken here. The "top down" approach asks the respondent to value the preservation of an ecosystem, such as 100 acres of wetland (John Whitehead and Blomquist 1991). The "bottom up" approach values the preservation of particular species inhabiting the wetland, such as geese and other birds.

Regardless of the approach taken, several problems must be faced. One difficulty is defining what substitutes are assumed to exist, whether for a particular species or for a wetland (Whitehead and Blomquist 1991). Presumably the value

⁵¹ This figure, reported by Chestnut and Rowe (1989), is an average of mean willingness to pay for each city surveyed by Tolley and Rae, based on Chestnut and Rowe's reanalysis of the data.

⁵² Outside environmental economics, there is a considerable literature in environmental ethics that explores the issue of nonhuman rights and their policy implications. From this perspective, the economist's benefit-cost calculation with its wholly anthropocentric orientation is an excessively narrow and illegitimate framework for analysis. Kneese and Schulze (1985) provide an excellent treatment of this set of issues.

placed on the preservation of 10,000 geese depends on the size of the goose population. A related problem arises when programs are valued one at a time; in general, the value attached to preserving several species at the same time is less than the sum of the values attached to preserving each species in isolation. This implies that the totality of what is to be preserved should be valued: one cannot compute this by summing the values attached to individual components.

To date, most studies of endangered species have valued individual species in isolation. For example, Bowker and Stoll (1988) estimate that households are, on average, willing to pay \$22 per year (\$1983) to preserve the whooping crane, while Boyle and Bishop (1987) find that non-eagle watchers are willing to spend \$11 per year to preserve the bald eagle in the state of Wisconsin. These values are appropriate if one is considering a program to preserve either of these species in isolation; however, the values should not be added together if one is contemplating preserving both species.

Even if one decides to value a wetland (of given size) and defines the nature of substitutes, an important question remains: do people really have well-defined, or in the terminology of psychologists, "crystallized" values for these commodities? Since respondents in CVM studies are likely to be less familiar with ecological benefits than with health and recreation benefits, responses are likely to depend critically on the information given to respondents in the survey itself (Karl Samples, John Dixon, and Marcia Gown 1986). This problem, however, is widely recognized, and recent studies have taken pains to see how responses are influenced by the amount of information provided.

5. *The Agricultural Benefits of Pollution Control*. Although we have empha-

sized the nonmarket benefits of pollution control, some benefits accrue directly to firms, and can be measured by examining shifts in the supply curves for the affected outputs. The industries that are most subject to ambient air and water pollution are forestry, fishing, and agriculture. We focus on agriculture because it is the sector that is likely to experience the largest benefits from pollution control.

Reductions in ozone concentrations and, possibly, in acid rain, should increase the yields of field crops such as soybeans, corn, and wheat. In addition, reductions in greenhouse gases, to the extent that they prevent increases in temperature and decreases in precipitation in certain areas, should also increase crop yields.

In measuring the effects on agricultural output of changes in pollution concentrations or climate, two approaches can be taken. The damage function approach translates a change in environmental conditions into a yield change, assuming that farmers take no actions to mitigate the effects of the change. The yield change shifts the supply curve for the crop in question, and the corresponding changes in consumer and producer surpluses are calculated.⁵³ This is the predominant approach used thus far to analyze the effects of global climate change (Sally Kane, John Reilly, and Tobey 1991). It has also been used in some studies of the effects of ozone on field crops (Richard Adams, Thomas Crocker, and Richard Katz 1984; Raymond Kopp et al. 1985; Kopp and Krupnick 1987).

The averting behavior approach allows farmers to adjust to the change in pollution/climate by altering their input mix and/or by adjusting the number of acres

⁵³ In calculating the welfare effects of a shift in supply, one must be careful to take into account the effects of agricultural price support programs, which distort market prices. See Erik Lichtenberg and David Zilberman (1986).

planted. In some applications, a profit function is estimated in which the environmental pollutant enters as a parameter (James Mjelde et al. 1984; Philip Garcia et al. 1986). The value of the change in Q can then be computed directly from the profit function. If the resulting shift in supply is big enough to alter market price, the welfare effects of these price changes must also be computed.

A more common approach is to solve for the effect of the change in pollution on output using a mathematical programming model whose coefficients have not been econometrically estimated (Adams, Scott Hamilton, and Bruce McCarl 1986; Scott Hamilton, McCarl, and Adams 1985). The effect of output changes on price is then computed separately.

While benefit estimates that allow farmers to adjust to changes in pollution are clearly preferable on theoretical grounds to estimates that do not allow such adjustments, it is important to ask how much of a difference this is likely to make empirically, especially as the damage function approach is much easier to implement. For changes in temperature and precipitation, damages are likely to be greatly overstated if opportunities for mitigating behavior (e.g., irrigation) are ignored.⁵⁴ On the other hand, mitigating behavior does not seem to make a great deal of difference in the case of ozone damage (Scott Hamilton, McCarl, and Adams 1985).

Estimates of annual damage to field crops from a 25 percent increase in ozone are in the neighborhood of \$2 billion (\$1980)—not negligible, but small relative to estimates of health damages. It is also interesting to note that most of

these damages are borne by consumers. Producers in most cases gain from yield decreases due to the resulting increases in prices!

Kane, Reilly, and Tobey (1991) obtain similar results when estimating the welfare effects of global climate change on agriculture: reductions in the yields of field crops (wheat, corn, soybeans, and rice) in the U.S., Canada, China, and the USSR benefit producers worldwide due to increases in commodity prices. Consumers, however, lose. Thus, although the aggregate losses to producers and consumers worldwide are small (about one-half of one percent of world GDP), food-importing countries such as China suffer large welfare losses (equal to 5.5 percent of GDP) while food exporters such as Argentina enjoy welfare gains.

E. Measuring the Costs of Pollution Control

Table 1, which lists the costs of the major environmental statutes, may give the reader the impression that measuring the costs of pollution control is a straightforward matter. Such is not the case.

To begin with, the costs of pollution control must be measured using the same concepts that are used to measure the benefits of pollution control: the change in consumer and producer surpluses associated with the regulations and with any price and/or income changes that may result. The figures in Table 1 represent, for the most part, expenditures on cleaner fuels or abatement control equipment by firms. They do not represent the change in firms' profits, and thus ignore any adjustments firms may make to these expenditures. The figures also ignore the price and output effects associated with reducing emissions. At the very least, one would want to take into account the price changes likely to result within a sector because of environmental regulations—for example, one would

⁵⁴ We base this statement on the results of the RFF MINK project (Norman Rosenberg et al. 1990), which examines damages associated with climate change—specifically, a return to the climate of the dust bowl—in Missouri, Iowa, Nebraska, and Kentucky, under alternate adjustment scenarios.

want to measure the welfare effects of an increase in electricity prices resulting from the 10 million ton reduction in SO₂ emissions by electric utilities projected under the 1990 Amendments to the Clean Air Act.

We note that, at least in the short run, the effect of ignoring these adjustments is to overstate the cost of environmental regulations. Abatement expenditures overstate the loss in firms' profits if firms can pass on part of their cost increase to consumers. Consumers in turn can avoid some of the welfare effects of price increases of "dirty" goods by substituting "clean" goods for "dirty" ones.

When environmental regulations affect sectors, such as electricity production, that are important producers of intermediate goods, it may be important to measure the impacts that environmental regulations have throughout the economy. Computable general equilibrium models, preferably those in which supply and demand functions have been econometrically estimated, may be needed to measure correctly the social costs of environmental regulation.

Michael Hazilla and Kopp (1990) have used an econometrically estimated CGE model of the U.S. economy to compute the social costs of the Clean Air and Clean Water Acts, as implemented in 1981. The effects of these regulations on firms are modeled as an upward shift in firms' cost functions, to which firms can adjust by altering their choice of inputs and outputs. It is interesting to contrast the estimates of social costs obtained from this approach with EPA's estimates of compliance costs. The EPA estimated the costs of complying with the Clean Air and Clean Water Acts in 1981 to be \$42.5 billion (1981 dollars). Hazilla and Kopp estimate the costs to be \$28.3 billion; the lower figure reflects the substitution possibilities that the expenditure approach ignores.

In the long run, however, the social costs of the Clean Air and Clean Water Acts exceed simple expenditure estimates because of the effects of decreases in income on saving and investment. In their analysis of the effects of environmental regulation on U.S. economic growth, Dale Jorgenson and Peter Wilcoxon (1990a) measure this effect. Using a CGE model of the U.S. economy, they estimate that mandated pollution controls reduced the rate of GNP growth by .191 percentage points per annum over the period 1973–85.

V. *The Costs and Benefits of Environmental Programs*

The value of a symptom-day or a statistical life is, of course, only one component in evaluating a pollution control strategy. To translate unit benefit values into the benefits of an environmental program requires three steps: (1) the emissions reduction associated with the program must be related to changes in ambient air or water quality; (2) the change in ambient environmental quality must be related to health or other outcomes through a dose-response function; (3) the health or nonhealth outcomes must be valued. The information required for the first two tasks is considerable, especially if one wants to evaluate a major piece of legislation such as the Clean Air Act or Clean Water Act.

In this section we review attempts to estimate the benefits and costs of environmental programs. Of central interest are cases in which benefit-cost analyses have actually been used in setting environmental standards; in addition, we discuss instances in which such analyses have not been used but should be. This leads naturally to a discussion of priorities for research in the area of benefit and cost measurement.

A. *The Use of Benefit-Cost Analysis in Setting Environmental Standards*

Executive Order 12291, signed in 1981, requires that benefit-cost analyses be performed for all major regulations (defined as those having annual costs in excess of \$100 million). Furthermore, the order requires, *to the extent permitted by law*, that regulations be undertaken only if the benefits to society exceed the costs.

One consequence of Executive Order 12291 is the undertaking of benefit-cost analyses for all major environmental regulations; however, the extent to which benefits and costs can be considered in making regulations is limited by the enabling statutes. Of the major environmental statutes only two, the Toxic Substances Control Act (TSCA) and the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) explicitly require that benefits and costs be weighed in setting standards.⁵⁵ Some standards—specifically, those pertaining to new sources under the Clean Air Act and to the setting of effluent limitations under the Clean Water Act—allow costs to be taken into account, but do not suggest that benefits and costs be balanced at the margin. In contrast, the National Ambient Air Quality Standards and regulations for the disposal of hazardous waste under RCRA and CERCLA are to be made without regard to compliance costs.

In spite of these limitations, benefit-cost analyses have been used in EPA's rulemaking process since 1981. Between February of 1981 and February of 1986, EPA issued 18 major rules (USEPA 1987), including reviews of National Ambient Air Quality Standards for three pollutants—nitrogen dioxide, particulate

matter, and carbon monoxide—effluent standards for water pollutants in the iron and steel and chemicals and plastics industries, and regulations to ban lead in gasoline, as well as certain uses of asbestos.⁵⁶ Regulatory Impact Analyses (RIAs) were prepared for 15 of these rules.

In five of the RIAs, both benefits and costs were monetized; however, benefits could legally be compared with costs only in the case of lead in gasoline. In this case, the benefits in terms of engine maintenance alone were judged to exceed the costs by \$6.7 billion over the period 1985–92, and the regulation was issued. In two other cases—the PM standard and effluent limitations for iron and steel plants—the benefits exceeded the costs of the proposed regulation and the regulation was implemented, although EPA denied that it weighed benefits against costs in reaching its decision. The remaining cases are more difficult to evaluate. The clean water benefits of proposed effluent guidelines for chemicals and plastics manufacturers were judged to exceed regulatory costs in some sections of the country but not in others. EPA recommended that these guidelines be implemented. Of several alternative standards for emissions of particulate matter by surface coal mines, only one was found to yield positive net benefits, and these were small (\$300,000). Eventually, no regulation was issued by EPA.

The preceding review suggests that benefit-cost analysis has not entirely been ignored in setting environmental standards, but its use has been selective. In part, this is the result of law—EPA was allowed to weigh benefits against costs for only 5 of the 18 major regulations that it issued between 1981 and

⁵⁵ Some portions of the Clean Air Act, specifically, those pertaining to aircraft emissions, motor vehicle standards and fuel standards, also require that marginal benefits and costs be balanced.

⁵⁶ A complete listing of the regulations may be found in USEPA (1987). Also included were regulations governing the disposal of used oil, and standards regarding land disposal of hazardous waste.

1986.⁵⁷ One could argue that the government should not invest resources in a full blown benefit-cost analysis if the results of such an analysis cannot be used in regulating the polluting activity. But this would be a mistake. Even where the explicit use of a benefit-cost test is prohibited, such studies can be informative and useful. In their own way, they are likely to influence the views of legislators and regulators. In particular, the issue is often one of amending standards—either raising them or lowering them. Benefit-cost information on such adjustments, although not formally admissible, may well have some impact on decisions to revise standards. In addition, simply demonstrating the feasibility and potential application of such studies may lead to their explicit introduction into the policy process at a later time.

B. *The Need for Benefit-Cost Analyses of Environmental Standards*

We turn now to a set of priorities for benefit-cost analyses of environmental regulation: which of existing environmental programs require closest scrutiny and what benefit techniques must be developed in order to perform these analyses? We begin with an enumeration of these programs, as we see them, and then offer some thoughts on the analysis of each of them.

There are, broadly, two areas in which careful benefit-cost analyses are most needed. One is for statutes whose total costs are thought to exceed their total benefits. A widely cited example is the Clean Water Act (CWA), which will soon be up for renewal. Freeman (1982) sug-

gests that the recreational use values associated with the adoption of BPT are small, relative to the costs presented in Table 1. Justification for these standards must then rest on other grounds. A second example where costs may exceed benefits involves the extent of cleanup of Superfund sites under CERCLA. While the cost of cleaning up these sites is predicted to run into the hundreds of billions of dollars, the health benefits of these cleanups are thought by many to be modest (Curtis Travis and Carolyn Doty 1989). Current law does not require an explicit benefit-cost analysis of remedial alternatives at each Superfund site, but, in our view, it probably should.

The second general class of cases in which careful benefit-cost analyses are needed is where environmental standards are sufficiently stringent to push control efforts onto the steep portion of the marginal cost of abatement curve. Even though the total costs of these standards may exceed their total benefits (see Figure 4), society might experience a gain in welfare from relaxing the standard if the marginal benefits of abatement are considerably below the marginal costs at the level of the standard. In terms of Figure 4, we need to know whether the marginal benefit function is MB_2 or MB_1 . There are several instances of actual policies that appear to fall within this class: (1) the ground-level ozone standard, in areas that are currently out of compliance with the standard; (2) certain provisions in RCRA for disposal of hazardous waste; and (3) the 1990 acid rain amendments to the Clean Air Act. In addition to these existing laws, proposals for significant reductions in CO_2 emissions may entail high marginal costs, suggesting a close scrutiny of benefits.

Turning first to the Clean Water Act, we note that evaluating the CWA will require computing the use (recreation) and nonuse (ecological) benefits of im-

⁵⁷ For the other four regulations where a comparison of costs and benefits was allowed—the three toxic substances (TSCA) regulations and the setting of emission standards for light duty trucks—benefits were quantified but not monetized. In the case of PCB's the cost per catastrophe avoided was computed; in the case of asbestos, the cost per life saved.

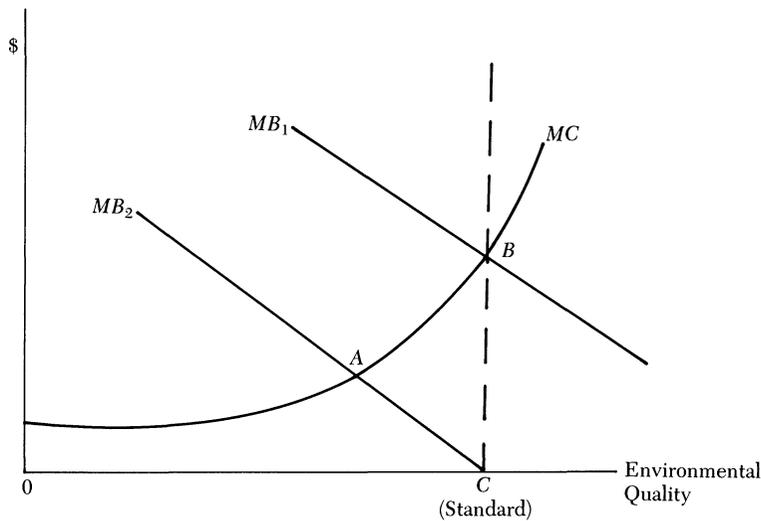


Figure 4. Welfare Loss from Setting Incorrect Standards

proved water quality. As we noted above, one can either use a contingent valuation approach that captures both values, or one can attempt to capture use values using travel cost methods and measure nonuse values separately. Whichever approach is used, we emphasize the regional character of the costs and benefits of improved water quality; benefit estimates must, in consequence, be available at this level of disaggregation. The contingent valuation method avoids two problems inherent in the use of travel cost models. First, unless the transferability problem can be solved, travel cost models will have to be estimated for each river or lake throughout the U.S.! And, second, if a contingent valuation survey of nonuse values is to be added to travel cost measures of use values, it may be hard to get users to separate use from nonuse values.

A key issue in valuing the benefits of Superfund cleanups is how to value health risks—usually risks of cancer—that will not occur until the distant future. Many Superfund sites pose very low health risks today, primarily because

there is no current route of exposure to toxic waste. People could, however, be exposed to contaminated soils or groundwater if substances were to leak from storage containers in the future. This involves valuing future risks to persons currently alive as well as to persons yet unborn. While some research has been done in this area (Mauskopf 1987; Cropper and Portney 1990; Cropper and Sussman 1990), there are few empirical studies that examine either the value that people place on reducing future risks to themselves or the rate at which they discount lives saved in future generations. Estimates of these values are also crucial if one is to analyze regulations governing the current disposal of hazardous waste under RCRA, as well as other regulations that affect exposure to carcinogens (e.g., air toxics and pesticide regulations).

An additional problem is how to incorporate uncertainty regarding estimates of health risks into the analysis. While most valuation studies treat the probability of an adverse outcome as certain, in reality there is great uncertainty about health risks, especially the risk of contracting

cancer from exposure to environmental carcinogens. This uncertainty has two sources: uncertainty about actual exposures received, and uncertainty about the effects of a given exposure.⁵⁸ The standard procedure in risk assessments is to “correct” for this uncertainty by presenting a point estimate based on very conservative assumptions (Nichols and Richard Zeckhauser 1986). It would, however, be more appropriate to incorporate the distribution of cancer risk into the analysis.

Existing estimates of the marginal costs and marginal benefits of achieving the one-hour ozone standard in areas that are currently out of attainment suggest that marginal costs exceed marginal benefits (Krupnick and Portney 1991). Estimates of the health benefits of ozone control have, however, focused on the value of reducing restricted-activity or symptom days. There is some evidence that ozone may exacerbate the rate at which lung tissue deteriorates, contributing to chronic obstructive lung disease (COPD). Since, for healthy individuals, the probability of contracting COPD is uncertain, what must be valued is a change in the risk of contracting chronic lung disease corresponding to a change in ozone concentrations.

The objective of the provisions of the 1990 Amendments to the Clean Air Act aimed at reducing SO₂ and NO₂ is to reduce acid rain, primarily in the Eastern U.S. and Canada. Although the 10-million-ton reduction in sulfur emissions specified in the amendments is likely to have some health benefits, most of the anticipated benefits are ecological or rec-

reational, resulting from an increase in the pH of lakes.⁵⁹ There are also likely to be visibility benefits (reduced haze) in the Eastern U.S. This underscores the need for better estimates of the value of improved visibility, especially in urban areas. It will also be necessary to measure the ecological benefits associated with reduced acid rain, especially as these are likely to differ qualitatively from the ecological benefits associated with the CWA.

Finally, we note that in the area of global climate change, considerable attention has been devoted to measuring the costs of reducing greenhouse gas emissions, especially through the use of a tax on the carbon content of fuels (Jorgenson and Wilcoxon 1990b). Little, however, is known about the benefits of reducing greenhouse gases, even if one assumes that the link between CO₂ and climate change is certain.⁶⁰

The benefits of preventing these climate changes differ from the benefits associated with conventional air and water pollutants in two respects. First, many—though by no means all—of the effects of climate change are likely to occur through markets. These include effects on agriculture and forestry, as well as changes in heating and cooling costs. While this should make benefits easier to measure, the problem is that the effects of CO₂ emissions are not likely to be felt for decades. This implies that valuing such damages is difficult. A damage function approach, which ignores adaptation possibilities, is clearly inappropriate; however, predicting technological possibilities for adaptation is not easy.

Second, the benefits of reducing greenhouse gases will not be felt until the next century. The problem here is that, even at a discount rate of only 3

⁵⁸ Estimates of the effect of a given exposure usually come from rodent bioassays, which are used to estimate a dose-response function. In addition to uncertainty regarding the parameters of the dose-response function, there is uncertainty as to how these estimates should be extrapolated from rodents to man.

⁵⁹ For a dissenting view see Portney (1990).

⁶⁰ A useful beginning here is the work of William Nordhaus (1990).

percent, one dollar of benefits received 100 years from now is worth only 5 cents today. This problem has typically been addressed by suggesting that benefits should be discounted at a very low rate, if at all. An alternative approach is to make transfers to future generations to compensate them for our degradation of the environment, rather than to alter the discount rate.

C. *The Distribution of Costs and Benefits*

In addition to examining the costs and benefits of environmental legislation, it is of interest to know who pays for pollution abatement and who benefits from it. Typically, studies of the distributional effects of environmental programs emphasize the distribution of benefits and costs by income class.

To determine how the benefits of environmental programs are distributed across different income classes, we must measure how the programs alter the physical environments of different income groups. In one study of the distributional effects of programs aimed at raising the level of national air quality, Leonard Gianessi, Peskin, and Edward Wolff (1979) found striking locational differentials in benefits; not surprisingly, most of the benefits from efforts to improve air quality are concentrated in the more industrialized urban areas (largely the heavily industrialized cities of the East) with fewer benefits accruing to rural residents. Even within metropolitan areas, air quality may differ substantially. Since the poor often live in the most polluted parts of urban areas, they might be thought to be disproportionately large beneficiaries of programs that reduce air pollution—and there is evidence that this is, indeed, the case (Asch and Seneca 1978; Jeffrey Zupan 1973). While this may be true, certain indirect effects can follow that offset such benefits. For ex-

ample, cleaner air in what was a relatively dirty area may increase the demand for residences there and drive up rents, thereby displacing low-income renters. All in all, this is a complicated issue. At any rate, Gianessi, Peskin, and Wolff find that within urban areas the distribution of benefits may be slightly pro-poor, but, as we shall see next, this is likely to be offset (or more than offset) by a regressive pattern of the costs of these programs.⁶¹

We are on somewhat more solid ground on the distribution of the costs of environmental programs (G. B. Christiansen and Tietenberg 1985). There exist data on the costs of pollution control by industry with which one can estimate how costs have influenced the prices of various classes of products and how, in turn, these increased prices have reduced the real incomes of different income classes. In one early study of this kind, Gianessi, Peskin, and Wolff (1979) examined the distributive pattern of the costs of the Clean Air Act and found that lower-income groups bear costs that constitute a larger fraction of their income than do higher-income classes. (See also Nancy Dorfman and Arthur Snow 1975; Gianessi and Peskin 1980.) Three independent studies of automobile pollution control costs all reach similar findings of regressivity (Dorfman and Snow 1975; Harrison 1975; Freeman 1979b).

In a more recent study, Robison (1985) uses an input-output model to estimate the distribution of costs of industrial pollution abatement. Assuming that the costs of pollution control in each industry are passed on in the form of higher prices, Robison traces these price in-

⁶¹ Moreover, there is some persuasive evidence from observed voting patterns on proposed environmental measures (Robert Deacon and Perry Shapiro 1975; Fischel 1979) indicating that higher income individuals are willing to pay more for a cleaner environment than those with lower incomes.

creases through the input-output matrix to determine their impact on the pattern of consumer prices. Robison's model divides individuals into twenty income classes. For each class, estimates are available of the pattern of consumption among product groups. This information, together with predictions of price increases for each product, is used to estimate the increase in the prices of goods consumed by each income group. Robison finds that the incidence of control costs is quite regressive. Costs as a fraction of income fall over the entire range of income classes; they vary from 0.76 percent of income for the lowest income class to 0.16 percent of income for the highest income class.

It is true that these studies relate to existing environmental programs and do not measure directly the potential distributional effects of a system of economic incentives such as effluent fees. But our sense is that the pattern of control costs across industries would be roughly similar under existing and incentive-based programs. It is the same industries under both regimes that will have to undertake the bulk of the abatement measures. Our conjecture thus is that the pattern of costs for our major environmental programs is likely to be distinctly regressive in its incidence, be they of the command-and-control or incentive-based variety.

While the distributional effects of environmental programs may not be altogether salutary, we do not wish to exaggerate their importance. We emphasize that the primary purpose of environmental programs is, in economic terms, an efficient allocation of resources. Environmental measures, as Freeman (1972) has stressed, are not very well suited to the achievement of redistributional objectives. But an improved environment provides important benefits for all income classes—and we will be doing no groups a favor by opposing environmental pro-

grams on distributional grounds. At the same time, there are opportunities to soften some of the more objectionable redistributive consequences of environmental policies through the use of measures like adjustment assistance for individuals displaced from jobs in heavily polluting industries and the reliance on the more progressive forms of taxation to finance public spending on pollution control programs.

VI. *Environmental Economics and Environmental Policy: Some Reflections*

As suggested by the lengthy (and only partial) list of references and citations in this survey, environmental economics has been a busy field over the past two decades. Environmental economists have reworked existing theory, making it more rigorous and clearing up a number of ambiguities; they have devised new methods for the valuation of benefits from improved environmental quality; and they have undertaken numerous empirical studies to measure the costs and benefits of actual or proposed environmental programs and to assess the relative efficiency of incentive-based and CAC policies. In short, the "intellectual structure" of environmental economics has been both broadened and strengthened since the last survey of the field by Fisher and Peterson in this *Journal* in 1976.

But what about the contribution of environmental economics to the design and implementation of environmental policy? This is not an easy question to answer. We have seen some actual programs of transferable emissions permits in the United States and some use of effluent charges in Europe. And with the enactment of the 1990 Amendments to the Clean Air Act, the U.S. has introduced a major program of tradable allowances to control sulfur emissions—moving this

country squarely into the use of incentive-based approaches to regulation in at least one area of environmental policy.⁶² But, at the same time, effluent charge and marketable permit programs are few in number and often bear only a modest resemblance to the pure programs of economic incentives supported by economists. As we noted in the introduction, certain major pieces of environmental legislation prohibit the use of economic tests for the setting of standards for environmental quality, while other directives require them! The record, in short, is a mixed and somewhat confusing one: it reveals a policy environment characterized by a real ambivalence (and, in some instances, an active hostility) to a central role for economics in environmental decision making.⁶³

What is the potential and the likelihood of more attention to the use of eco-

⁶² Under this provision, the U.S. will address the acid rain problem by cutbacks in sulfur emissions over the next decade of 10 million tons (about a 50 percent reduction). This is to be accomplished through a system of tradable allowances under which affected power plants will be allowed to meet their emissions reductions by whatever means they choose—including the purchase of “excess” emissions reductions from other sources that choose to cut back by more than their required quota. Also noteworthy is the U.S. procedure to implement reductions in chlorofluorocarbon emissions under the Montreal Protocol. Under this measure, EPA has effectively grandfathered the U.S. quota among existing producers and importers; from these baselines, firms are allowed to trade allowances (Hahn and McGarland 1989).

⁶³ Some recent studies of actual environmental decision making are consistent with this “mixed” view. Magat, Krupnick, and Harrington (1986), for example, in a study of EPA determination of effluent standards under the Clean Water Act Amendments of 1972, found that “simple rules based either on economic efficiency or the goal of distributional equity did not dominate the rulemaking process” (p. 154). Their analysis did find that standards across industry subcategories reflected to some extent differences in compliance costs among firms. In contrast, Cropper et al. (1992) find that EPA decisions on pesticide regulation have, in fact, reflected a systematic balancing of environmental risks and costs of control. Economic factors, it appears, have mattered in some classes of decisions and not in others.

omic analysis and economic incentives in environmental management? It is easy to be pessimistic on this matter. There is still some aversion, both in the policy arena and across the general public, to the use of “market methods” for pollution control. While we were working on this survey, one of the leading news magazines in the U.S. ran a lengthy feature story entitled “The Environment: Cleaning Up Our Mess—What Works, What Doesn’t, and What We Must Do to Reclaim our Air, Land, and Water” (Gregg Easterbrook 1989, in *Newsweek*). A central argument in the article is that the attempt to place environmental policy on a solid “scientific” footing has been a colossal error that has handcuffed efforts to get on with pollution control. Proceeding “on the assumption that environmental protection is a social good transcending cost-benefit calculations” (p. 42), Easterbrook argues that we should not place a high priority on scientific work on the complicated issues of measuring benefits and costs and of providing carefully designed systems of incentives, but should get on with enacting pollution control measures that are technologically feasible. In short, we should control what technology enables us to control without asking too many hard questions and holding up tougher legislation until we know all the answers.

Such a position has a certain pragmatic appeal. As we all know, our understanding of complicated ecological systems and the associated dose-response relationships is seriously incomplete. And as our survey has indicated, our ability to place dollar values on improvements in environmental quality is limited and imprecise. Nevertheless, we have some hard choices to make in the environmental arena—and whatever guidance we can obtain from a careful, if imprecise, consideration of benefits and costs should not be ignored.

We stress, moreover, that the role for economic analysis in environmental policy making is far more important now than in the earlier years of the "environmental revolution." When we set out initially to attack our major pollution problems, there were available a wide array of fairly direct and inexpensive measures for pollution control. We were, in short, operating on relatively low and flat segments of marginal abatement cost (MAC) curves. But things have changed. As nearly all the cost studies reveal, marginal abatement cost functions have the typical textbook shape. They are low and fairly flat over some range and then begin to rise, often quite rapidly. Both the first and second derivatives of these abatement cost functions are positive—and rapidly increasing marginal abatement costs often set in with a vengeance.

We now find ourselves operating, in most instances, along these rapidly rising portions of MAC functions so that decisions to cut pollution yet further are becoming more costly. In such a setting, it is crucial that we have a clear sense of the relative benefits and costs of alternative measures. It will be quite easy, for example, to enact new, more stringent regulations that impose large costs on society, well in excess of the benefits, health or otherwise, to the citizenry. As Portney (1990) has suggested, this may well be true of the new measures to control urban air pollution and hazardous air pollutants under the most recent Amendments to the Clean Air Act. Portney's admittedly rough estimates suggest that the likely range of benefits from these new provisions falls well short of the likely range of their cost.

Economic analysis can be quite helpful in getting at least a rough sense of the relative magnitudes at stake. This is not, we would add, a matter of sophisticated measures of "exact consumer surplus" but simply of measuring as best we can

the relevant areas under crude approximations to demand curves (compensated or otherwise). In addition to measurement issues, this new setting for environmental policy places a much greater premium on the use of cost-effective regulatory devices, for the wastes associated with the cruder forms of CAC policies will be much magnified.⁶⁴

In spite of the mixed record, it is our sense that we are at a point in the evolution of environmental policy at which the economics profession is in a very favorable position to influence the course of policy. As we move into the 1990s, the general political and policy setting is one that is genuinely receptive to market approaches to solving our social problems. Not only in the United States but in other countries as well, the prevailing atmosphere is a conservative one with a strong disposition toward the use of market incentives, wherever possible, for the attainment of our social objectives. Moreover, as we have emphasized in this survey, we have learned a lot over the past twenty years about the properties of various policy instruments and how they work (or do not work) under different circumstances. Economists now know more about environmental policy and are in a position to offer better counsel on the design of measures for environmental management.

This, as we have stressed, takes us from the abstract world of pure systems of fees or marketable permits. Environmental economists must be (and, we be-

⁶⁴ Following our earlier discussion of the Weitzman theorem, we note its implication for the issue under discussion here: a preference for price over quantity instruments. So long as there is little evidence of any dramatic threshold effects or other sources of rapid changes in marginal benefits from pollution control, the steepness of the MAC function suggests that regulatory agencies can best protect against costly error by adopting effluent fees rather than marketable emission permits (Hadi Dowlatabadi and Harrington 1989; Oates, Portney, and McGartland 1989).

lieve, are) prepared to come to terms with detailed, but important, matters of implementation: the determination of fee schedules, issues of spatial and temporal variation in fees or allowable emissions under permits, the life of permits and their treatment for tax purposes, rules governing the transfer of pollution rights, procedures for the monitoring and enforcement of emissions limitations, and so on. In short, economists must be ready to "get their hands dirty."

But the contribution to be made by environmental economists can be a valuable one. And there are encouraging signs in the policy arena of a growing receptiveness to incentive-based approaches to environmental management. As we noted in the introduction, both in the United States and in the OECD countries more generally, there have been recent expressions of interest in the use of economic incentives for protection of the environment. As we were finishing the final draft of this survey, the Council of the OECD issued a strong and lengthy endorsement of incentive-based approaches, urging member countries to "make a greater and more consistent use of economic instruments" for environmental management (OECD 1991).

Finally, we note the growing awareness and concern with global environmental issues. Many pollutants display a troublesome tendency to spill over national boundaries. While this is surely not a new issue (e.g., transnational acid rain), the thinning of the ozone shield and the prospect of global warming are pressing home in a more urgent way the need for a global perspective on the environment. The potential benefits and costs of programs to address these issues, particularly global warming, are enormous—and they present a fundamental policy challenge. The design and implementation of workable and cost-effective measures on a global scale are formidable

problems, to put it mildly. And they call for an extension of existing work in the field to the development of an "open economy environmental economics" that incorporates explicitly the issues arising in an international economy linked by trade, financial, and environmental flows.⁶⁵

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⁶⁵ For a useful effort to develop a research perspective and agenda on the economic analysis of global change, see U.S. National Oceanic and Atmospheric Administration, National Science Foundation and National Aeronautics and Space Administration (1991).

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