ECONOMIC INCENTIVES TO ENCOURAGE HAZARDOUS WASTE MINIMIZATION AND SAFE DISPOSAL

by

Richard B. Belzer and Albert L. Nichols

Energy and Environmental Policy Center
John F. Kennedy School of Government
Harvard University
Cambridge, Massachusetts 02138
November 9, 1988

Cooperative Agreement No. CR813491-01-2

Submitted to:

U.S. Environmental Protection Agency
Office of Policy, Planning and Evaluation
Washington, DC 20410

Project Officers:

Greg Michaels
Ron McHugh

DISCLAIMER

The information in this document has been funded wholly or in part by the United States Environmental Protection Agency under cooperative agreement CR813491-01-2 to the Energy and Environmental Policy Center, John F. Kennedy School of Government, Harvard University. It has not been subjected to the Agency's peer and administrative review, has not been approved for publication as an EPA document, and does not necessarily reflect the views of the Agency. No official endorsement by the Agency or any of its officials should be inferred.

ABSTRACT

This report examines opportunities for the application of economic incentive instruments for the management and reduction of hazardous wastes. The analysis concludes that as long as firms comply with existing regulations and bear potential tort liability, no additional incentives appear to be warranted. There may be significant residual external damages, however, if full compliance cannot be ensured. Neither waste-minimization standards nor simple economic incentives appear promising as regulatory strategies aimed at noncompliant firms. An innovative economic incentive instrument is examined in which the overt identification of noncompliers is not required.

This instrument is applied to the case of used lubricating oil. An empirical methodology is developed in which the effects of the incentive are modeled based on exogeneous parameters which include: (1) the relative price-responsiveness among waste generators, (2) the level of unit transactions costs, and (3) the level of risk associated with existing or projected disposal practices. The analysis shows that the conditions under which the instrument may offer net external benefits do not appear to exist in the case of used oil, but may well be present in more hazardous waste streams. Suggestions for future research and additional case studies are offered.

TABLE OF CONTENTS

Chapter 4: THE IMPACTS OF NONCOMPLIANCE AND ENFORCEMENT ON RISK AND WASTE MINIMIZATION

Chapter 5: SUBSIDIES TO ENCOURAGE COMPLIANCE AND WASTE MINIMIZATION

Chapter 6: COMBINING TAXES AND SUBSIDIES INTO A COORDINATED REGULATORY INSTRUMENT

Chapter 7:
AN APPLICATION OF COMBINED TAX-SUBSIDY INSTRUMENTS: A DEPOSIT-REFUND SYSTEM APPLIED TO USED OIL

Chapter 8:
SUGGESTIONS FOR FUTURE RESEARCH

LIST OF FIGURES

LIST OF TABLES

Chapter 1

INTRODUCTION

In the 1984 amendments to the Resource, Conservation and Recovery Act (RCRA), Congress declared "it to be national policy of the United States that, wherever feasible, the generation of hazardous waste be reduced or eliminated as expeditiously as possible."[1] Most observers readily agree with the Congress that modifying production processes to reduce the amounts of toxic byproducts or treating wastes to render them harmless are, at least in principle, far more satisfactory approaches than more traditional methods of disposing of wastes. The Congressional Office of Technology Assessment (OTA 1986; OTA 1987) has been a particularly strong proponent, calling on the Environmental Protection Agency (EPA) to shift its focus from regulating wastes to discouraging their creation in the first place. At Congress's direction, the EPA recently completed a major study of opportunities for waste minimization (EPA 1986a).

Waste minimization also has strong appeal among various interest groups. For environmentalists, the attraction is obvious; wastes that are not produced pose no direct threat to human health or the environment. The appeal also is strong for citizens' groups concerned about the siting of hazardous waste treatment, storage, and disposal facilities (TSDFs) in their communities; they can respond to the question of "If not here, where?" by arguing that industry can eliminate (or at least sharply reduce) the need for such facilities by avoiding the crea-

[1] RCRA Section 1003(b).

tion of waste in the first place. For industries currently generating hazardous wastes, waste minimization is attractive as a possible route out of the increasingly high costs of disposal in conformance with RCRA and state rules and the potentially crushing burden of tort liability.

Competing (and Confusing) Definitions

There seems to be a broad consensus that it is desirable to reduce the amount of waste generated. However, this consensus breaks down once the nature and form of new governmental programs comes up. Indeed, there is considerable confusion and disagreement concerning precisely what types of activities should be encouraged, and by whom. These definitional conflicts are well illustrated by noting certain areas in which the EPA and the OTA disagree.

Waste Minimization, Waste Reduction, and Similar Terminology

The OTA has argued fervently for a fundamental reorientation of the nation's environmental policy, one that focuses on preventing pollution rather than treating or disposing it at the end of the pipe. The OTA advocates, among other things, the establishment of a new, high-level Office of Waste Reduction within the EPA. This office would have broad responsibilities that cross the media-specific program boundaries upon which the agency has long been organized. Furthermore, the OTA is quite restrictive in its definition of the kinds of actions that should qualify as what is calls "waste reduction:" The mass <u>and</u> the toxicity of waste must be simultaneously reduced, and this must be accomplished <u>only</u> within the production process and not added on at the end.

The OTA claims that this definition accurately reflects the Congressional intent of the waste minimization policy objective established in the 1984 RCRA Amendments. This is debatable because the statutory language is hardly explicit -- a point that the OTA readily concedes (OTA 1986: 46). Furthermore, since the Congress defined 39 terms in RCRA Section 1004, but refrained from defining waste minimization, waste reduction, or any similar concept that might operationalize its broad policy statement, one might just as easily conclude that the Congress deliberately left the issue ambiguous.

In the 1984 RCRA Amendments, the EPA was directed to submit a report

> on the feasibility and desirability of establishing standards of performance or of taking other additional actions under this Act to require the generators of hazardous waste to reduce the volume or quantity and toxicity of the hazardous waste they generate...[2]

The words "volume" and "quantity" are widely viewed as synonyms; hence the conjunction "or" between them. But these terms are connected to "toxicity" with the conjunction "and," implying (certainly in the OTA's view) simultaneity. Nonetheless, in the EPA's Report to Congress the agency construed this as an error:

> EPA does not interpret that language to indicate that Congress rejected volume reduction alone (with no change in the toxicity of hazardous constituents) as being a legitimate form of waste minimization. A generator that reduces the volume of its hazardous waste, even if the composition of its waste does not change, is accomplishing beneficial waste minimization. EPA believes that waste concentration may occasionally be a useful waste minimization technique.[3]

[2] RCRA Section 8002(r).

[3] EPA 1986a: iv.

What the OTA calls "waste reduction" is closely related to what the EPA (and many others) define as "source reduction," a term that has several years of historical usage but, to further muddle the debate, does not appear in RCRA.

The Universe of Wastes To Be Covered

Another area of disagreement is the universe of wastes to be covered by any new waste minimization regulations or programs. The EPA sticks to the statutory and regulatory definitions implied by RCRA inasmuch as the waste minimization issue has arisen within that context. The OTA, however, would apply its preferred concept (i.e., "waste reduction") to all waste streams irrespective of the program under which they are regulated, and indeed, irrespective of whether they are currently regulated at all. In the OTA's view, a comprehensive multimedia approach to pollution prevention should not be artificially constrained by program boundaries when technical criteria for making distinctions are difficult to muster (OTA 1986: 11; OTA 1987: 25).

Waste Minimization and the Limits of Command and Control

One point on which there does appear to be consensus is that traditional command-and-control regulatory approaches employing standards are unlikely to be of much value in directing waste minimization (OTA 1986: 130; EPA 1986a: 94). Specification standards, which set requirements for particular types of equipment, clearly would not work. Establishing them would require that the government get involved in the details of production decisions in millions of individual firms employing many thousands of different processes, a prospect that should daunt even the most ardent advocate of regulation. Moreover, most observers seem to agree that significant reductions in the amounts of waste generated are likely to require innovative changes in processes and, in some cases, products, which are difficult to specify a priori.

Performance standards -- setting limits on the amount of waste produced per unit of output -- are somewhat more plausible, but also are likely to fail to accomplish much at reasonable cost. As with specification standards, regulators would face a large and diverse universe of production processes and firms. It is hard to imagine EPA having the resources needed to determine what levels of waste *were* appropriate for each of them, particularly in the context of innovation in response to price and regulatory changes. All of these problems would be exacerbated by the fact that many waste streams are complex mixtures of different substances, making it difficult to develop workable definitions of waste reduction. In addition, although performance standards are less likely than specification standards to freeze technology, it is difficult to write appropriate performance standards when technological change is an important factor. Writing standards that can be met with current technology may provide insufficient incentive for innovation, but tighter standards based on the performance of new, perhaps as yet unspecified, technologies run the risk of being infeasible or excessively expensive if the hoped-for improvements fail to perform as expected.

As a result of these problems and others, most operative and proposed policies have eschewed command-and-control in favor of persuasion, information, technical assistance, or limited financial incentives. So far, it is unclear that existing waste-minimization policies have had any measurable effect. RCRA requires that generators file annual reports on their waste-minimization activities. When filing manifests for shipping hazardous wastes, they also must sign the following statement:

> I have a program in place to reduce the volume and toxicity of waste generated to the degree I have determined to be economically practicable, and I have selected the method of treatment, storage, or disposal currently available to me which minimizes the present and future threat to human health and the environment.

The meaning of this statement, particularly with its qualifying clause of "economically practicable," is unclear. As a result, the "requirement" that all generators have waste minimization programs has only the force of moral suasion.

Programs to encourage waste minimization are more active at the state level. Some states now offer small grants or tax breaks to firms undertaking source reduction. Others offer technical assistance or work with trade associations to encourage the diffusion of information on waste-reduction techniques. Such efforts, however, remain minuscule compared to the resources devoted to writing standards for and issuing permits to disposal facilities.

Economic incentives may offer useful alternatives or supplements to more traditional forms of regulation in seeking to reduce hazardous wastes. Economists have long advocated incentives for their efficiency in allocating resources when regulated parties vary widely in their costs of achieving desired goals. Such flexibility is particularly im-

---

[4] 50 **Federal Register** 28744 (July 15, 1985).

portant for waste minimization, because the most cost-effective approaches are likely to vary so widely across industries and firms.

An Overview of this Report

This report examines some of the possible ways in which the regulation of hazardous wastes might be improved by more explicitly integrating economic incentives into the regulatory system. Although our emphasis is the effect of incentives on waste minimization, we do not limit our analysis to reductions in waste, because we think that it is important to remember that waste minimization is not an end in itself, but rather one of several interacting means of reducing the risks of hazardous waste. The desirability of waste minimization should not be evaluated in isolation from the rest of the system.

In addition, our analysis goes beyond incentives that are targeted explicitly at reducing waste. The amounts and types of waste minimization that occur depend not only on policies targeted directly on such activities, but also on the incentives created by other policies, particularly those that regulate disposal methods. The most efficient incentives for reducing wastes (and risks) may be indirect, with little or no explicit tie to waste minimization.

Chapter 2 lays out our basic analytic framework, and uses it to examine the effect of existing indirect incentives for waste minimization. It shows that firms will have inadequate incentives to reduce wastes if disposal is risky and those risks are not internalized, as was probably the case until relatively recently. Stringent disposal standards and liability rules, however, have lowered risks and raised disposal costs, so that the gains from additional waste reduction are likely to be relatively small, if firms comply with the regulations.

In Chapter 3, we examine the problem of incomplete compliance with RCRA rules. Some types of waste are outside the RCRA framework, while in other cases firms fail to obey the rules, either out of ignorance or by conscious choice. Many (perhaps most) firms nominally within the regulatory system fail to meet at least some requirements. Firms outside the regulatory system, operating in the "black market," are potentially a more serious problem, but their numbers are difficult to gauge.

We apply our framework to the problem of noncompliance in Chapter 4, first showing how tightening standards, while it may reduce risk from firms that comply and also reduce the wastes they generate, can increase risk by driving more firms out of compliance. Enforcement is the obvious traditional answer to noncompliance, but ordinary regulatory enforcement targeted at firms known to the regulator can have perverse effects by driving them into even more hazardous black-market disposal. To avoid that problem, law enforcement efforts also must be directed against the black market. To the extent that enforcement efforts are successful, they also increase waste-minimization efforts by raising the effective cost of disposal.

Enforcement attempts to encourage safe disposal by raising the expected costs of the alternatives. In contrast, safe-disposal subsidies, as discussed in Chapter 5, act by lowering the cost of disposal in compliance with RCRA rules. Such subsidies may be particularly desirable when it is difficult to identify noncompliers because they create incentives for firms to identify themselves. Disposal subsidies, however, lower the cost of generating waste, and thus weaken incentives for waste minimization. In theory, subsidies for waste minimization can offset these problems, but they are almost certain to be very difficult to implement. Most of the waste-minimization subsidies that have been considered thus far are indirect and incomplete, and suffer from major targeting inefficiencies. Moreover, as has long been known in the economics literature, subsidies to reduce negative externalities pose a variety of problems, including distorted product prices.

As we analyze in Chapter 6, in some cases pairing an input tax with a safe-disposal subsidy may combine the best of both types of incentive instruments -- encouraging waste minimization through the tax while the subsidy provides an incentive for firms to choose legal disposal methods. Despite the general lack of enthusiasm in the political system for incentives, such tax-subsidy schemes flourish under the rubric of deposit-refund systems.

Chapter 7 presents as a case study the problem of used lubricating oil, a large volume waste stream that has troubled policy makers and regulators for many years. We show how a deposit-refund system could be used to reduce the amount of used oil that is dumped or disposed in environmentally harmful ways. We provide crude estimates of the net benefits of such an approach, and also analyze the impact of transactions costs and other factors.

Finally, in Chapter 8 we briefly suggest areas for future research.

Chapter 2

INDIRECT INCENTIVES FOR WASTE MINIMIZATION

Waste minimization may play a key role in efforts to reduce the health and environmental risks posed by toxic substances. In many instances, reducing the amount of waste will be both cheaper and environmentally safer than generating and disposing of it. Waste minimization, however, is not an end in itself, but rather one of several means for reducing risk. Moreover, progress in the reduction of hazardous wastes is affected by many policies that are not explicitly targeted on it, in particular by regulations that affect the cost of disposal. Thus, we should not examine waste minimization in isolation, but rather as part of a broader set of interacting opportunities for reducing risk.

In this chapter, we develop a framework for examining those interactions, and then evaluate the extent to which existing regulatory, legal, and market forces provide appropriate incentives for waste minimization. We begin with a simple model of a firm's decision in which risk is determined by the volume and toxicity of waste generated and by how it is treated and disposed once created. This model allows us to explore how existing regulation of treatment and disposal indirectly affects waste minimization decisions, and to show how incentives for waste minimization may be suboptimal because firms do not pay for the risks imposed by their wastes.

If disposal risks are tightly regulated and disposal is costly, however, the potential gains from inducing firms to engage in additional waste minimization appear to be small. These potential gains look even smaller when we extend the model from the firm to the level of the market. In the final section, we examine the extent to which existing regulations and tort liability have already motivated firms to internalize damages.

Waste-Minimization at the Level of the Firm

The Basic Model

We begin with a basic model of the choices.facing a firm with respect to waste minimization and disposal of wastes. Figure 2-1 presents a very simple flow chart, starting with inputs to the firm's production process, which yields both the final product and wastes. The amount and composition of the wastes can be affected by the firm's choice of inputs and its production process. Finally, the firm also must choose how to treat and dispose of the wastes that are created.[1] In describing the model, we proceed backwards, starting with disposal.

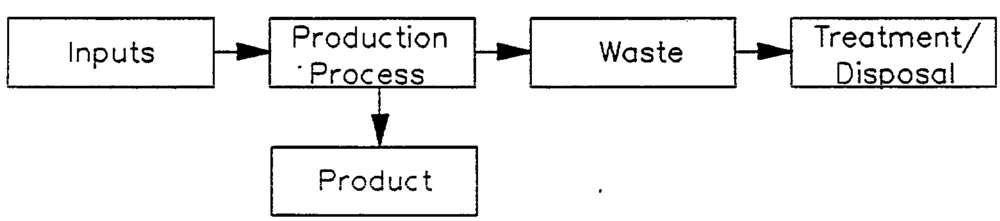


Figure 2-1. Flow Diagram of Waste Generation and Disposal within the Firm

[1]"Treatment" may encompass everything from extensive actions that render the waste harmless to simply pouring it into a drum. For convenience, we shall refer to "treatment and disposal" simply as "disposal."

Disposal costs and risks. The risk posed by waste depends not only on its volume and toxicity when generated, but also on how it is treated and disposed. Let r be the risk posed by a unit of waste, where r is affected both by treatment and the method of disposal. For carcinogens, for example, r might be defined in terms of predicted cancer cases per unit of waste. We assume that for a given combination of waste types, treatment, and disposal, r does not vary with w, the amount of waste per unit of output; i.e., the incremental risk posed by another unit of waste is invariant with respect to the amount of waste. This assumption is consistent with the standard regulatory presumption that the dose-response function for carcinogens is linear at low doses. Even if the dose-response function is not linear, it is likely to be a reasonable approximation for an individual firm's wastes if they are deposited at a site with the wastes from many other firms; in that case, the firm's wastes are small relative to the whole, so that marginal damage will not vary much with that firm's wastes.[2]

The level of risk per unit of waste can be affected in a wide variety of ways; some wastes, for example, can be treated chemically to sharply reduce their hazard potential. The risk also depends critically on the method of disposal; switching the disposal of a liquid waste from an unlined lagoon to sealed fiberglass containers in a double-lined

[2] If nonlinearities are significant, a full accounting of them would require taking account of the cumulative nature of wastes. In such cases, marginal risk is likely to be a function of cumulative wastes deposited at a site (and at other sites near enough to affect the same populations), rather than on just the flow of waste into treatment and disposal.

landfill, for example, is likely to reduce risk. Proper incineration may reduce risk still further.

We denote the cost of achieving a risk level of r as D(r). Thus, -D' (r) is the marginal cost of reducing risk. We assume that D(r) is U-shaped; even in the absence of any intervention, cost-minimizing behavior by the firm leads to a finite risk level, r_N (where the subscript "N" is mnemonic for "non-regulated"). Reducing risk below that level, however, increases disposal costs:

$$
-D'(r) \begin{cases} < 0, \text{ for } r > r_N, \\ = 0, \text{ for } r = r_N, \\ > 0, \text{ for } r < r_N, \text{ and} \end{cases} \tag{2-1}
$$

$$
D''(r) > 0.
$$

If the disposal facility is not regulated and is not liable for the risk imposed, it will set $r = r_N$, the cost-minimizing risk level. Note that although this is the point at which the marginal cost of risk reduction is zero, the cost of disposal, $D(r_N)$, is likely to be positive; even if firms bear no liability for damages and disposal is unregulated, disposing of most wastes entails some cost.[3]

Note also that while we assume that the marginal cost of risk reduction rises as r is reduced over the relevant range ($r \leq r_N$), the unit cost of disposal does not vary with volume; i.e., for any given type of waste and risk level, the incremental cost of disposing of an-

[3] One important exception would be wastes that have a positive net value in recycling or other uses.

other unit of waste does not vary. This assumption is valid if the market supply curve of disposal services is perfectly elastic, or if the individual firm's demand for disposal services is small relative to the market as a whole.[4]

Cost of waste minimization. Let w be the amount of hazardous waste produced per unit of output, where w is defined to incorporate both volume and degree of hazard. We assume constant returns to scale in the production of the good and its associated waste; for any given technology, doubling output doubles costs and generates twice as much waste. The cost of producing a unit of output with an associated waste of w is M(w), the cost of the least-cost combination of inputs (including the design of the production process) for that level of waste. This cost function, however, does not incorporate any disposal costs or damages caused by the waste that are external to the firm.

We assume that M(w) also is U-shaped; i.e., at high levels of waste, firms save money (even without considering disposal costs or external damages) by reducing wastes. If firms are cost minimizers and disposal does not have a negative price, however, the marginal cost of waste minimization per unit of output, -M' (w), will be positive over the relevant range. We make the further reasonable assumption that the

[4] For the market as a whole, the supply curve is likely to slope upwards; higher prices are needed to elicit additional supply. We deal with this case later in the chapter. For individual firms, the marginal cost of disposal may fall with volume; many disposal firms charge less per unit for large quantities of waste than for small ones. However, so long as the marginal cost of disposal does not fall more rapidly than the marginal cost of risk reduction (-D' (r),) the marginal efficiency condition still gives a global optimum.

marginal cost of waste reduction increases as waste-reduction efforts become more intense (i.e., as w is driven towards zero):[5]

(2-2) $-M'(w) \geq 0$, and
$M''(w) > 0.$

If disposal is competitive, the price that a firm faces for disposal will be equal to its cost.[6] Thus, the firm's cost of producing a unit of output if disposal is unregulated is:

(2-3) $C = M(w) + wD(r_N)$

To find the optimal amount of waste minimization, we differentiate Equation 2-3 with respect to w:

(2-4) $\partial C/\partial w = 0 = M'(w) + D(r_N),$

which yields:

(2-5) $-M'(w) = D(r_N).$

[5] This assumption is inconsistent with the view that some enthusiasts of waste minimization seem to hold -- that firms routinely pass up significant potential cost savings from waste minimization. Although we do not claim that real firms have perfect knowledge and constantly optimize to reduce costs, we also think it unreasonable to think that firms consistently, over long periods of time, choose production processes that are more costly than necessary, as measured from the firms' perspective.

[6] Later in this chapter, we discuss what happens if the providers of disposal services exercise market power, and hence set the price above marginal cost.

Thus, the firm minimizes waste to the point at which the, marginal cost of reducing waste is equal to the cost of disposing of a unit of waste. We denote w_N as the level of waste at which this condition is achieved.[7]

It is important to recall that w incorporates the degree of hazard as well as volume. Although for convenience we generally shall refer to w as a quantity, reductions in w could mean reductions in hazard as well as reductions in volume; in the simple case in which w is proportional to volume and toxicity, for example, a four-fold reduction in w could reflect reducing volume or toxicity by a factor of four, cutting both in half, or some other combination of actions. Changes in w could come about through changes in inputs (e.g., switching to a less toxic solvent), changes in the production process (e.g., increasing the efficiency of a conversion process so that more of a feedstock ends up as product and less is generated as waste), or improved pre-disposal management practices (e.g., pretreatment or neutralization).

For the moment, we assume that the only significant risk occurs once the wastes have been treated and disposed; i.e., we do not consider risks in the production, transportation, and pretreatment processes. The risk posed by producing a unit of output is then wr, the amount of waste times the risk per unit of waste.

<u>Social costs</u>. To derive an expression for social cost, we need to express the risks in the same units as the costs. Let λ be the cost of

[7] Note that w_N is not the level of waste at which $M'(w) = 0$, because the cost of disposal in the absence of regulation is unlikely to be zero.

a unit of risk. For example, in the case of a carcinogen, λ might be defined as the dollar value assigned to preventing a case of cancer. Assigning such a value obviously raises a host of complex and troubling issues that have been addressed at length, though without complete resolution, in the literature.[8] Fortunately, for our purposes here, how most of those issues are resolved is irrelevant; all that we require is acceptance that there is some finite value to reducing risk that can be assigned, either explicitly or implicitly. We also assume, for reasons argued elsewhere (Nichols and Zeckhauser 1986), that the value of risk reduction is proportional to the amount of risk reduced; e.g., preventing two cases of cancer has twice the value of preventing one case.

Using the notation developed above, the social *cost* of producing a unit of output is given by:

$$S = M(w) + wD(r) + \lambda wr \tag{2-6}$$

The first term on the right-hand side is simply the firm's cost of producing a unit of output given that it generates w units of waste per unit of output. The second term is the cost of disposal, which is the amount of waste times the unit *cost* of disposal, with the latter a function of the risk level. The final term is residual external damage, which is the product of three factors: the risk per unit of waste (r), times the amount of waste per unit of output (w), times the dollar value of risk (λ).

[8] See, e.g., Bailey (1980), and the summary by Nichols (1984: 135).

Optimal disposal and waste minimization. To find the optimal solution, we take the partial derivatives with respect to the two control variables, w and r, and set the results equal to zero:

(2-7) $$\partial S/\partial r = 0 = wD'(r) + \lambda w, \text{ or}$$
$$-D'(r) = \lambda.$$

(2-8) $$\partial S/\partial w = 0 = M'(w) + D(r) + \lambda r, \text{ or}$$
$$-M'(w) = D(r) + \lambda r.$$

Equation 2-7 says that the optimal disposal efforts reduce risk (r) to the level at which the marginal cost of risk reduction is equal to its marginal value; note that because we have assumed that damages are proportional to the amount of waste, the optimal level of disposal risk is independent of the amount of waste generated. Figure 2-2 illustrates Equation 2-7. The marginal cost of risk reduction rises as the risk level is reduced, starting from a value of zero at $r = r_N$; the optimum, r^*, is the risk level at which the marginal cost curve intersects the marginal damage curve, the horizontal line at a height of λ. Reducing r from r_N (the value that firms would choose in the absence of regulation or internalizing the risks) to r^* yields a net benefit per unit of waste equal to the area of the darkly-shaded triangle, which is

[9]This result may strike many readers as improbable; after all, the nature of the waste will affect the optimal type and degree of treatment and disposal. Our model does not violate that fact, however, as it does not say that the optimal disposal of a unit of waste is independent of the nature of the waste, only that the price of disposing of any particular kind of waste is fixed over the relevant range for a firm.

the difference between the value of the reduction in risk (the sum of the two shaded areas) and the increase in disposal costs (the lightly-shaded triangle).

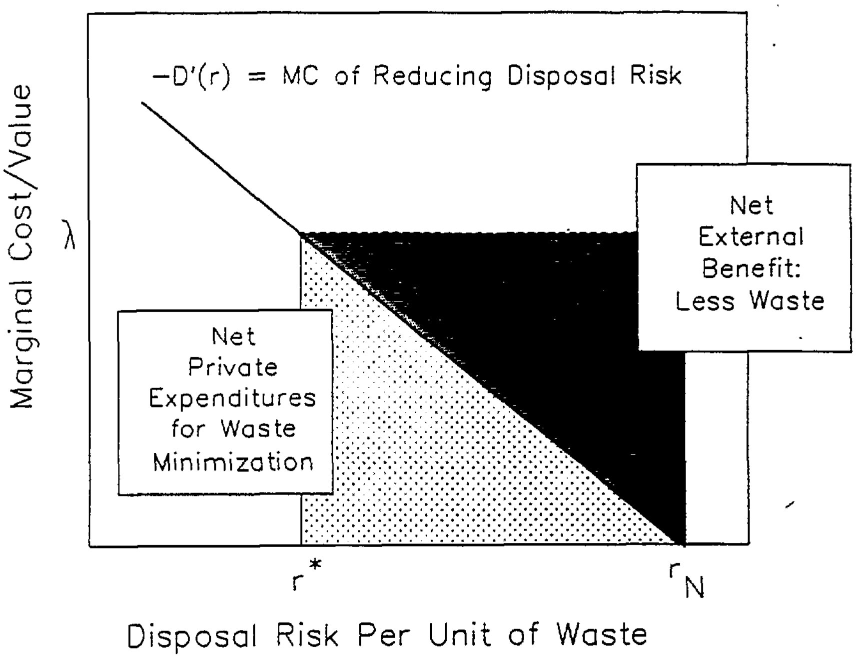


Figure 2-2. Optimal Level of Disposal Risk

Equation 2-8 says that waste minimization should proceed to the point at which the marginal cost of additional effort is just equal to the cost of disposal plus the residual damage caused by a unit of waste (given the current disposal effort). Note that the optimal level of waste minimization depends on the level of disposal through two counter-acting mechanisms; as disposal is made safer (r is reduced), disposal costs [D(r)] rise, which increases the desirability of waste minimization. Reducing r, however, reduces the residual external damage caused by a unit of waste, which reduces the optimal level of waste minimization.

Figure 2-3 illustrates the situation; for simplicity, we have used the realization of the firm's cost function, C, instead of the full functional specification in Equation 2-3. In Panel (a), if disposal is unregulated, the firm chooses a waste level of w_N, the point at which the marginal cost of waste minimization [-M'(w)] is equal to the cost of unregulated disposal $[D(r_N)]$, which we now denote as C_N. Given that level of disposal risk, however, the marginal social value of minimizing waste is $S_N = C_N + \lambda r_N$, the highest horizontal line, and the socially optimal amount of waste per unit of output is w_1. Thus, if disposal is cheap but risky, the firm will engage in relatively little waste minimization, but a comparatively high level of effort will be justified in terms of social net benefits. If we had no control over disposal, but could regulate the amount of waste minimization, the optimal level of waste to allow would be w_1. At least in theory, that level could be achieved either by imposing a standard at that level or by charging the firm a fee of λr_N for each unit of waste disposed. The net benefit per unit of output of reducing waste from w_N to w_1 would be the area of the shaded triangle in Panel (a), *c* + *d* + *e,* which is the decline in external damages less the cost of the reduction in waste.

Now consider the opposite extreme; we could regulate disposal to reduce risks to their optimal level, r^*, but have no control over waste minimization. Reducing the risk from r_N to r^* reduces the net social cost of a unit of waste from

(2-9) $$S_N = C_N + \lambda r_N,$$

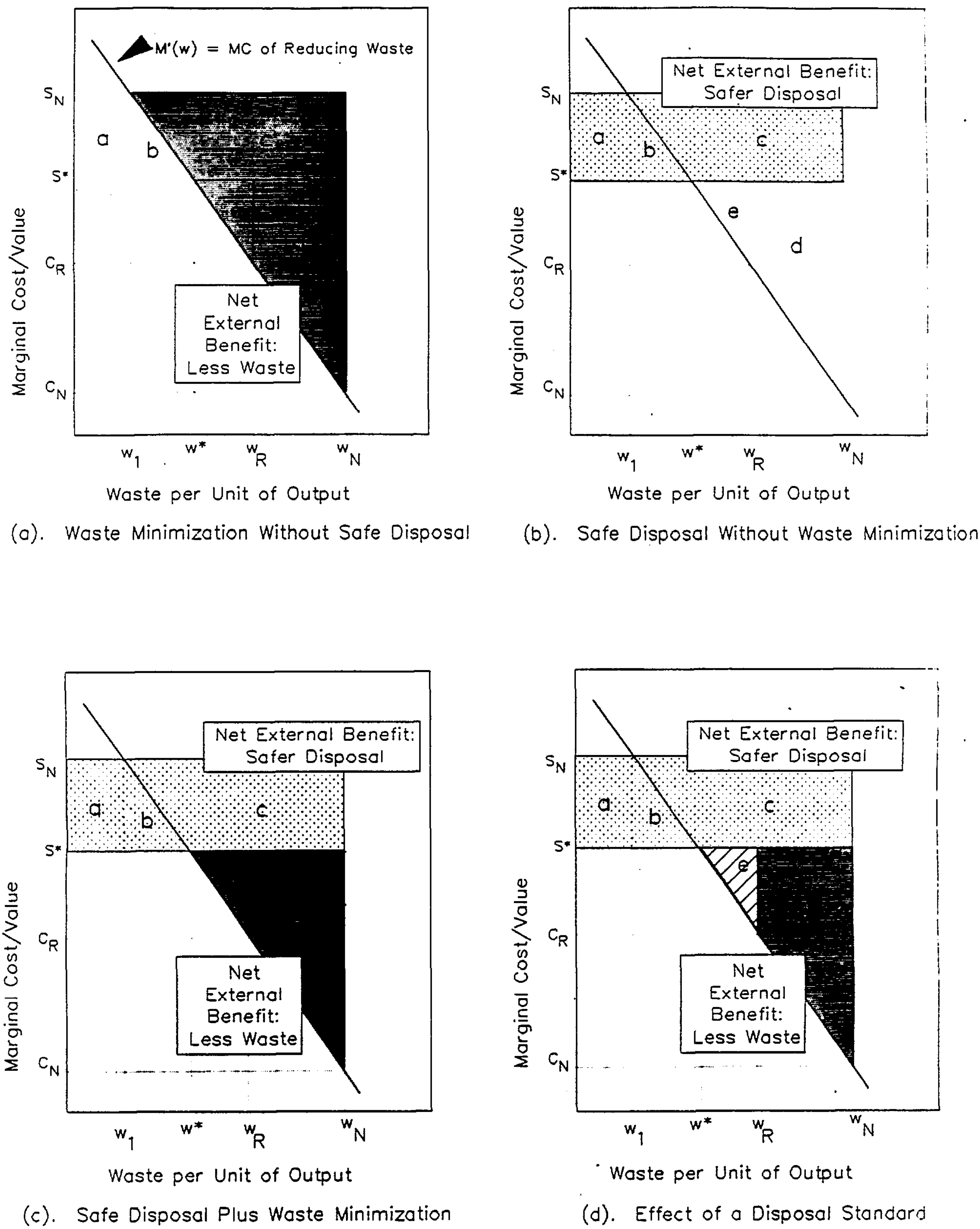


Figure 2-3. The Effects of Waste Minimization and Lower Disposal Risk

to

$$(2\text{-}10) \qquad S^* - D(r^*) + \lambda r^* = C_R + \lambda r^*,$$

where C_R is the cost per unit of waste of regulated disposal at the risk level r^*. This reduction is equal to the area of the darkly-shaded triangle in Figure 2-2, and represents the difference between reduced damages and higher disposal costs. If the amount of waste generated per unit of output remained at w_N, the social cost per unit of output would be reduced by the area of the shaded rectangle *(a + b + c)* in Panel (b) of Figure 2-3, which is the reduction in the social cost of a unit of waste times the amount of waste generated per unit of output.

Full optimality requires coordinating disposal and waste minimization, which is illustrated in Panel (c) of Figure 2-3. If disposal risk is reduced to r^* through direct regulation or other means, the optimal level of waste is shown as w^*, the level at which the marginal cost of waste minimization is just equal to the marginal social cost of another unit of waste given that disposal risks have been optimized. Note that w^* is less than w_N, the unregulated amount of waste, but higher than w_1, the optimal amount of waste at the original disposal risk. Thus, safer disposal reduces the social value of waste minimization.[10]

[10] If disposal regulations are pushed beyond the point at which marginal cost equals marginal benefit, additional tightening of disposal regulations will increase the total social cost of a unit of waste (because the incremental increase in disposal costs will exceed the incremental reduction in damages), thus increasing the value of waste reduction. The gap between social and private costs of waste disposal, however, will continue.to shrink as disposal regulations are made more stringent.

Once disposal risks have been lowered, reducing waste from w_N to w^* provides a net benefit per unit of output equal to the area of the darkly-shaded triangle *(d + e),* in addition to the benefit achieved by reducing the risk of disposal. Thus, the net benefit of optimizing both disposal risks and waste generation is equal to the sum of the two shaded areas *(a + b + c + d + e)* in Panel (c) of Figure 2-3.[11]

This optimum might appear to require simultaneous, coordinated regulation of both disposal and waste minimization. A charge levied on disposal, however, can achieve full efficiency by providing the appropriate indirect incentives for waste minimization.

Suppose the regulator levies a tax or fee of t per "standardized" unit of waste disposed, where the standardization factor is r, the risk per unit of waste. For example, if two lots of waste have the same volume, but one poses twice the risk as the other (because of differences in disposal methods or because of differences in the degree of hazard), the riskier one would be counted as twice as many standardized units.[12] Faced with this charge, the firm's cost minimization problem is:

$$\text{(2-11)} \qquad \text{Min } C = M(w) + wD(r) + twr.$$

If $t = \lambda$, the values of w and r that minimize Equation 2-11 will be the same ones that minimize social cost in Equation 2-6. The intuition be-

[11]Attributing the net benefits to safer disposal or reduced waste generation is, of course, somewhat arbitrary, as the part of area c lying above area *d* could be allocated to either.

[12]Equivalently, the fee could be adjusted to reflect the risk, and then levied on w alone.

hind this result is straightforward. The fee provides an incentive for firms to select less risky disposal methods because it increases with the riskiness of the disposal method selected. Reducing the risk raises the cost of disposal, and hence the price that generators must pay. In addition, because the fee must be paid for any residual risks, firms also are forced to internalize the remaining external damages of waste generation. As a result, they face the full social cost of generating waste, and thus have the appropriate incentive to reduce it. Moreover, regulators can set the optimal fee without knowing the cost functions for either waste minimization, $M(\cdot)$, or disposal, $D(\cdot)$. They need know "only" the marginal damage, λ.[13]

Disposal standard. Whatever the theoretical merits of such a charge, in practice the regulation of the risks of hazardous waste disposal relies almost entirely on standards; To what extent will such standards also provide appropriate incentives for waste minimization?

As shown earlier in Equation 2-7, the optimal level of disposal risk is independent of the level of waste minimization. Suppose that the regulator imposes a disposal standard of r^*. The firm must pay for disposal, but no requirements are placed on waste minimization and no charge is imposed for residual risk. Thus, the firm's cost minimization problem is:

---

[13] This conclusion requires at least two qualifications. First, marginal damage is extraordinarily difficult to estimate, because of uncertainties about both the risk levels and how to value their reduction. Efficient standards, however, are equally dependent on the reliability of the damage estimates, so that uncertainties about marginal damage do not provide a basis for choosing between price- and quantity-based schemes. Second, the optimal charge rate will not be independent of the cost functions if our assumption that damage is proportional to both w and r does not hold.

$$\text{(2-12)} \qquad \text{Min } C = M(w) + wD(r^*).$$

Differentiating with respect to the firm's only decision variable, w, yields:

$$\text{(2-13)} \qquad -M'(w) = D(r^*).$$

As in Equation 2-5, the firm reduces waste to the level at which the marginal cost of reducing waste is equal to its disposal cost. Compared to the unregulated equilibrium, however, two important factors have changed. First, disposal costs have risen, so that the firm. has a stronger incentive to reduce wastes; in Panel (d) of Figure 2-3, it reduces its waste per unit of output from w_N to w_R, the point at which Equation 2-13 is satisfied. Second, the net social cost of waste has declined, so that the socially optimal level of waste per unit of output has increased, from w_1 to w^*. As a result, the divergence between the private and the socially optimal levels of waste minimization is reduced substantially, though not eliminated altogether. A comparison with the social optimality condition given in Equation 2-8 and illustrated in Panel (c) of Figure 2-3 shows that the firm will not go far enough in reducing waste; Equation 2-13 does not include the residual damage term, λr.[14]

---

14. It is tempting to suggest that the regulator tighten the standard beyond r^*, so the firm's cost would be raised to S^*. The problem with that prescription is that unlike the fee or tax, which is a transfer, the extra control cost is a real cost. In addition, note that no matter how tight the standard (short of zero risk), there is still some residual damage that the firm is not paying for, so that this problem remains even if the standard is "too tight."

Relative to the unregulated outcome, the optimal disposal standard yields net benefits per unit of output equal to the sum of the two shaded areas *(a + b + c + d)* in Panel (d) of Figure 2-3. Relative to the optimal disposal charge, it falls short by the area of the diagonally-shaded triangle *e*. The size of that triangle is given by:

$$\text{(2-14)} \qquad \Delta S = [w_R - w^*][D(r^*) + \lambda r^*] - \int_{w^*}^{w_R} -M'(w)dw.$$

If we use a linear approximation for -M'(w), this expression may be rewritten as:

$$\text{(2-15)} \qquad \Delta S \approx [w_R - w^*][\lambda r^*]/2.$$

The marginal cost of minimizing waste, -M'(w), may be thought of as an inverse demand curve for the right to generate waste; i.e., it gives the marginal price that the firm would be willing to pay to be able to generate that much waste per unit of output. Let ϵ_w be the (absolute value of the) own-price elasticity of demand for generating waste at the price $D(r^*)$ and the quantity w_R. Along the "demand curve" [-M'(w)], the change in quantity $(w_R - w^*)$ is associated with a "price" change of λr^*. Thus, by the definition of elasticity,

$$\text{(2-16)} \qquad \epsilon_w = \frac{w_R - w^*}{\lambda r^*} \cdot \frac{D(r^*)}{w_R}, \quad \text{or}$$

(2-17) $$w_R - w^* = \epsilon_w[w_R/D(r^*)]\lambda r.$$

We then can rewrite Equation 2-15 as:

(2-18) $$\Delta S/w_R = \epsilon_w[\lambda r^*]^2/[2D(r^*)].$$

The left-hand side of Equation 2-18 is the potential welfare gain per unit of waste generated under the standard. This potential welfare gain is proportional to the elasticity of waste generation with respect to the firm's cost of disposal (ϵ_w); proportional to the square of residual damages (λr^*); and inversely proportional to disposal costs [$D(r^*)$].

If we define $\delta = \lambda r^*/D(r^*)$ as the residual damage per unit of waste as a fraction of disposal costs, the welfare change may be written on a proportional basis as:

(2-19) $$\Delta S/w_R = \epsilon_w\delta^2 D(r^*)/2.$$

The left-hand side is the welfare change per unit of waste generated under the standard. The right-hand side tells us that this change is proportional to the elasticity of demand for waste generation (ϵ_w); to the square of residual damage as a fraction of disposal cost (δ); and to the cost of disposal [$D(r^*)$]. Not surprisingly, the greater the ease with which firms can reduce wastes and the greater are the residual external risks, the larger is the potential efficiency gain from supplementing a disposal standard with instruments designed to encourage additional waste minimization.

Figure 2-4 plots this measure of the potential welfare gain as a function of δ for several different values of ϵ_w, where the vertical axis measures the potential gain as a fraction of disposal cost. Because the gain is proportional to the square of residual damage, it is particularly sensitive to that parameter; cutting residual damage in half reduces the potential gain by a factor of four. As noted before, the potential gain is directly proportional to ϵ_w, which measures the sensitivity of waste generation to the firm's cost of disposal. Thus, for example, for any given value of δ, the potential welfare gain for a firm with $\epsilon_w = 1$ will be twice as large as for a firm with $\epsilon_w = 0.5$.

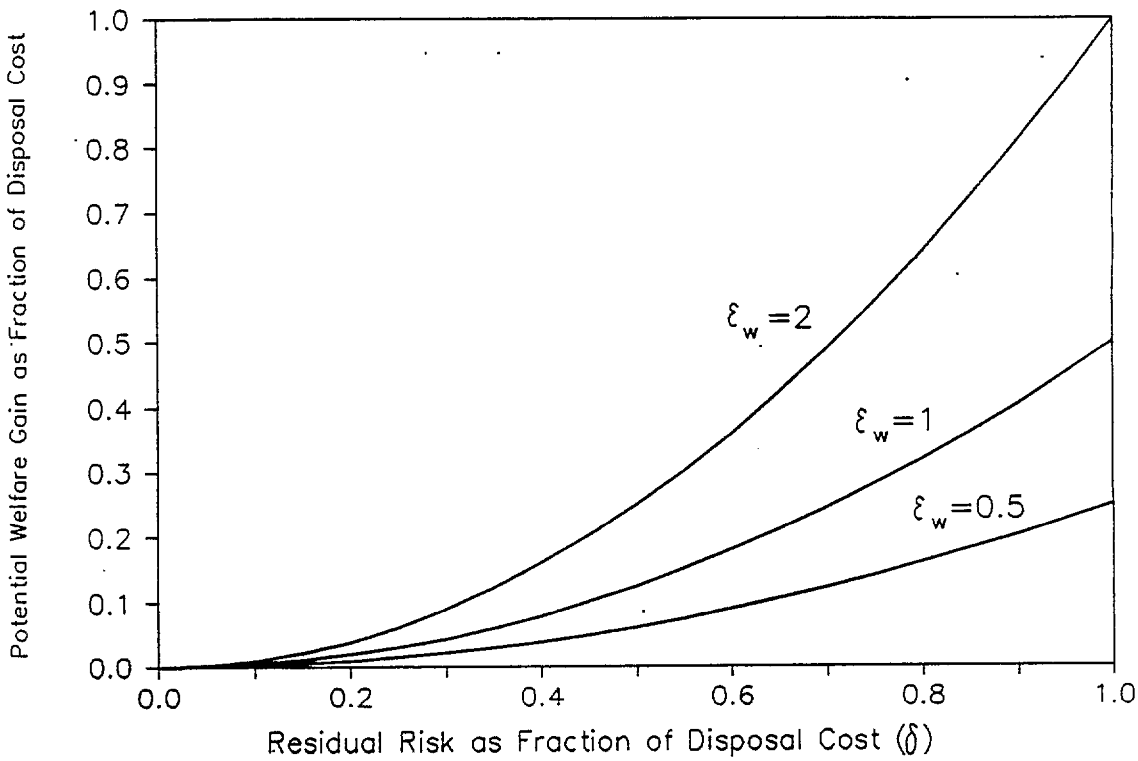


Figure 2-4. Potential Welfare Gain from Optimal Waste Minimization

Figure 2-4 indicates that for the potential welfare gain from additional waste minimization to be large relative to disposal costs, the damages caused by residual risk must be relatively large and waste generation must be quite sensitive to disposal costs. For example, even if the residual damage is as large as the disposal cost (i.e., δ = 1) and a 1 percent increase in disposal costs causes a 2 percent reduction in wastes (i.e., ϵ_w = 2), the potential welfare gain is still no larger than the cost of disposal.

In the absence of disposal regulations, the gains from causing damages to be reflected in waste minimization decisions may be substantial. With relatively low disposal costs and high risks, values of δ far in excess of unity are likely to occur. If stringent disposal standards are implemented, however, the potential gains from additional regulation of waste minimization are likely to be quite modest, because such standards will drive down risk and drive up disposal costs, both of which shrink the gap between the private and social costs of disposal. In general, this means that the benefits from additional waste minimization are likely to be small for wastes currently disposed in compliance with RCRA rules. However, the gains may be substantial if wastes are disposed outside the regulated system in ways that are inexpensive and risky.

Empirical Estimates Based on the Firm-Level Model

Empirical estimates of the potential efficiency gain from additional waste minimization require estimates of disposal costs, residual risks, and the sensitivity of waste generation to disposal costs. In this section, we develop crude estimates of these parameters and apply them to the model.

Disposal costs. The cost of hazardous waste disposal has risen dramatically in recent years. The California Department of Health Services, for example, estimated that the typical cost of land disposal in that state rose from $41 per ton in 1983 to $185 per ton in 1986 (California Task Force 1986b: 28), an increase of more than 350 percent in only three years. Even if we adjust for general inflation (using the GNP deflator for general business), the increase still amounted to 311 percent, or a compounded real annual rate of more than 60 percent.

Sharp increases in the price of disposal have not been restricted to California. Monsanto, a very large waste generator, reports that during the decade ending 1986, its cost per ton of disposing of non-halogenated solvents increased more than an order of magnitude, to $1,280 (Massachusetts Department of Environmental Management 1987); even in real terms, the compound average annual rate of increase was more than 20 percent. Figure 2-5 presents prices charged by Clean Harbors, a Massachusetts treatment and disposal firm lacking its own disposal facilities; the first bar in each pair shows the 1978 price (converted to 1987 dollars using the GNP Price Deflator for Domestic Business), while the second shows the price per ton disposed in 1987; the real-price increases ranged from 75 percent (6.4 percent per year) for halogenated solvents to over 1000 percent (31.0 percent per year) for waste oil. These cost increases reflect higher prices for particular disposal methods (e.g., landfill or incineration); changes in the allowable methods of disposal for some wastes (e.g., some wastes that previously were disposed in landfills now must be directed to more costly methods of treatment or disposal): and heightened concerns about liability.

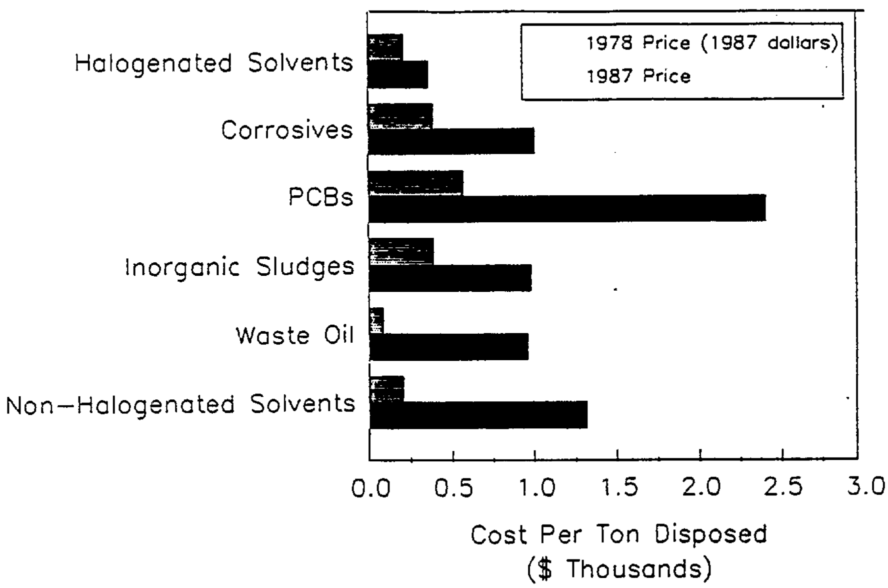


Figure 2-5. Comparison of Real Disposal Costs for Clean Harbors, 1978 and 1987 (1987 dollars)

Regulations promulgated to implement RCRA probably account for most of the increased cost of disposal nationally. Under RCRA, treatment, storage, and disposal facilities (TSDFs) must comply with complex and expensive new rules for facility design, management, and financial responsibility, all of which drive up costs. These costs are particularly great for facilities that must meet corrective action requirements for their existing disposal cells before they can obtain a Part B permit to accept additional wastes.

The initial impact of the RCRA rules was blunted by the routine granting of interim status to existing facilities. Impatient with the EPA's slow progress in implementing the full requirements, however, Congress eliminated interim status as of November 1985. After that date, TSDFs were not supposed to remain open unless they met certain conditions. In particular, they had to have applied for Part B permits, and they had to demonstrate that they met requirements for financial responsibility and groundwater monitoring.

These changes had a severe impact on the number of legally available facilities, despite efforts by the EPA to interpret the requirements flexibly.[15] Table 2-1 compares the numbers of facilities available prior to November 1985 to the numbers available as of April 1987. In less than two years, the number of storage/treatment facilities fell by a factor of three, and the number of land disposal facilities dropped even more sharply, by a factor of four. The supply is substantially tighter for firms that must rely on commercially available facilities. As of January 1987, there were a total of only 164 commercially available facilities nationwide, 22 states had no commercially available facilities, and 35 states had no RCRA-permitted, commercially available landfills (New York State Legislative Commission 1987: Table 7). Although the decline in the availability of facilities is not necessarily undesirable (one of Congress's major goals was to force the closure of inadequate facilities), clearly it has reduced disposal capacity and thus contributed to price increases.

The 1984 RCRA Amendments (HSWA) have driven up prices through changes in demand as well as supply. For example, the HSWA eliminated the small quantity generator (SQG) exemption. Although this change

---

[15] EPA recognized early in 1985 that many TSDFs would be unable to meet the financial responsibility requirements because of the collapse of the liability insurance market. Thus, the Agency issued enforcement guidance on April 12, 1985 that applicants able to demonstrate a "good faith" effort to obtain insurance would be allowed to remain open. Subsequently, EPA promulgated an interim final rule that allowed the use of a corporate guarantee of a parent firm in lieu of insurance (51 _FR_ 25351, July 11, 1986). _See_ Tenusak and Bailey (1987: 16).

Table 2-1. Changes in Number of Regulated Disposal Facilities, 1985-87

RCRA Status	Storage/ Treatment	Incinerators	Land Disposal
Prior to November 1985			
Total Regulated Facilities[a]	3,978	287	1,487
April 1987			
Issued Permit	427	42	39
Applied for permit	913	181	333
Total	1,340	223	372
Percentage change, 1985-87	-66.3	-22.3	-75.0

Note: [a] Includes interim status facilities closed as of November 8, 1985.

Source: "RCRA Permitting Status Update," Hazardous Waste Consultant (Lakewood Colo.: McCoy & Associates), July/August 1987, pp. 2-27.

sharply increased the number of generators subject to RCRA requirements, it probably will not have much impact on the amount of waste processed by RCRA-covered facilities, because SQGs appear to account for only about 1 percent of the waste generated (Abt Associates 1985).

HSWA also broadened.the range of wastes considered hazardous under RCRA. The statute directed the EPA to (1) list certain additional waste streams as hazardous; (2) tighten its extraction procedure as a predictor of leaching potential (the "EP Test");[16] (3) identify additional characteristics of hazardous waste, including toxicity; and (4) to list,

[16] EPA has proposed to modify the EP test to increase the number of toxicants considered in defining a waste stream as hazardous, and adopt the more sensitive Toxicity Characteristic Leaching Procedure (TCLP). See 51 FR 21648 (June 13, 1986). Both of these changes would result in broader definitions of hazardous wastes.

in cooperation with the Agency for Toxic Substances and Disease Registry and the National Toxicology Program, wastes as hazardous solely because of the presence of carcinogenic, mutagenic, or teratogenic constituents at "levels in excess of levels which endanger health" [RCRA Section 3001(b)(1)]. These changes increase demand for RCRA-approved disposal.

The HSWA also increased the effective costs of disposal by establishing deadlines for the banning of land disposal for various waste streams, forcing the generators of such streams to find alternative (and more expensive) disposal methods or to reduce the amounts of waste generated.[17] The land disposal bans are likely to drive up the cost of these alternative disposal methods because of increased demand and limited capacity.[18] These impacts will grow over time, as additional waste streams are banned according to the schedules set out in the statute.

Residual risks. The cost of legally disposing of hazardous waste has risen sharply and is likely to continue to increase for the indefinite future. Thus it is clear that the incentives for waste minimization are substantial and that the gap between social and private disposal costs has narrowed sharply over the past decade because of increases in disposal costs. The gap also appears to have narrowed be-

[17] RCRA Sec. 3014(d) gives EPA a sharply constrained ability to exempt a waste stream from a ban. EPA must essentially guarantee zero release from the site for as long as the waste remains hazardous. Bans on land disposal began prior to HSWA. California, for example, began banning wastes containing specific concentrations of toxic metals and PCBs in June 1983 (ICF 1985: II-6).

[18] These bans limit legal demand for land disposal, and thus may moderate price increases for that method of disposal. For most waste streams, however, the net effect is likely to be a price increase.

cause of reductions in risk and because firms now must internalize more of whatever risks remain.

It is impossible to estimate with any confidence the risks posed by the disposal of wastes in RCRA-approved facilities. As with any effort to quantify toxic risks, there are huge uncertainties in estimating dose-response functions based on limited animal and epidemiological data. Compared to many sources of air or water pollution, risk assessments for hazardous wastes are further handicapped by particular difficulties in predicting exposure and by the fact that most waste streams are complex mixtures of substances in varying proportions.

In addition to these inherent problems, our ability to estimate the risks from RCRA-approved facilities is further hampered by the fact that the EPA has not devoted much effort to careful risk assessment in connection with the RCRA or CERCLA programs, focusing instead on technology-based standards. The limited evidence available, however, suggests that the risks from wastes treated or disposed in conformance with RCRA rules are likely to be fairly small.

Recently, the EPA completed an agency-wide study to assess the comparative risks of the various environmental hazards that it regulates (EPA 1987). The effort relied on a combination of quantitative estimates and the judgments of senior Agency officials. The end result was a ranking of program areas along several dimensions of risk (human health, ecological effects, and welfare), assuming the implementation and enforcement of existing regulations. The RCRA program did not rank high on any of these dimensions.[19] The group's upper-bound estimate for

[19] It ranked 13 of 29 on cancer risks, 11 of 23 on welfare effects, 6 of 6 on ecological impacts, and 16 of 29 on non-cancer health risks.

cancer risks from RCRA-type wastes was 100 cases per year. If we apply that estimate to a base of 250 million tons of waste disposed per year, it translates to only 4 x 10^{-7} cancer cases per ton disposed.[20]

The risk estimates cited above are incomplete and highly uncertain. We believe, however, that they are far more likely to over- rather than underestimate actual risks. Because the exposure levels in virtually all cases are low, the relevant primary health risk is cancer. The EPA acknowledges that its procedures for estimating dose-response functions for carcinogens generate "plausible upper bounds" rather than realistic estimates of risk, and many observers argue that they are biased upwards by a substantial amount, possibly by several orders of magnitude, (Nichols and Zeckhauser 1986). As a result, it seems likely that the actual residual risks are even lower than the nominal target levels set by the EPA.

Rough estimates of potential gains from waste minimization. The data available allow us to make some crude upper-bound estimates of the potential welfare gains using Equation 2-19. Suppose that we take $200 as the private cost of disposing of a ton of waste, based on the Cali-

[20] The risk estimate reflects the stock of wastes, while the 250 million tons per year is a flow. If, however, the risk estimate of 100 cases per year reflects the steady-state stock of waste resulting from a constant annual flow, then it is appropriate to take the ratio of the two numbers to get an estimate of the average risk posed by a ton over the indefinite future. Note, however, that much of the risk associated with disposing of a ton of waste will occur well into the future, and that this method ignores that fact; i.e., it does not discount future risks. In addition, the estimate assumed that in the absence of RCRA regulations, contamination would not be discovered and that people would continue to drink contaminated water, which seems highly unlikely in light of the public pressure to test water and to find alternative sources if even minute concentrations of carcinogens are found.

fornia figure cited earlier for land disposal. For residual risk, let us use the figure of 4 x 10^{-7} cancer cases per ton, derived above from the EPA's comparative risk study. If we value each cancer case at $1 million, then residual damages are (4 x 10^{-7})($1 x 10^{6}) = $0.40 per ton, and δ = $0.40/$200 = 0.002, or 0.2 percent. Using Equation 2-18, the potential gain is then $\epsilon_w(0.002)^2(200)/2 = \$0.0004\epsilon_w$, or less than one-tenth of a cent per ton of waste, even if $\epsilon_w = 2$.

If we value each cancer case at $10 million (or inflate the risk per ton by a factor of 10 to account for non-cancer risks), the residual damage is still only $4 per ton and δ = $4/$200 - 0.02, or 2 percent. In that case, the potential welfare gain is $\epsilon_w(0.02)^2(200)/2 = \$0.04\epsilon_w$ per ton, which still is less than a dime per ton if $\epsilon_w = 2$.

These estimates suggest that the potential welfare gains from additional waste minimization, beyond that which firms will find economical under current disposal prices, are likely to be minimal, at least for wastes posing average risks. For wastes posing much higher risks, the gains might be substantially larger, although disposal costs under RCRA also are likely to be higher than average.

To place some rough upper bounds on these potential gains with high-risk wastes, consider an extreme case. Benzene is a widely used industrial chemical and a major ingredient in some solvents. It also is a human carcinogen, having been found to cause leukemia among workers exposed to it. We do not have data on the risks posed by benzene when it is disposed in various ways. We do, however, have the EPA's estimates of the risks posed by benzene emitted from maleic anhydride plants, which work out to less than 8 x 10^{-5} cases of cancer per ton of benzene emitted.[21] Presumably RCRA-approved disposal methods for wastes containing benzene pose far lower risks, because wastes are unlikely to be pure benzene and, more important, it is hard to imagine an approved disposal method as risky as evaporating the waste into the ambient air. Thus, we suspect this risk factor is orders of magnitude too high. If, as a rough cut, we again use a value of $1 million per case of cancer avoided, then the damage per ton of benzene would be $(8 \times 10^{-5})(1 \times 10^{6})$ = $80/MT. From Figure 2-5, the cost of disposing of non-halogenated solvents was $1280 per ton in 1986. Taking the ratio of those two numbers gives us a value for δ = 80/1280 - 0.0625. The potential welfare gain is then $\epsilon_w(0.0625)^2(1280)/2 = \$2.50\epsilon_w$, or $2.50 per ton for $\epsilon_w = 1$ and only $5 per ton for ϵ_w - 2.

Extending the Model to the Market Level

Our basic model focuses on decisions by individual generators operating in a competitive market. If the market for disposal is perfectly competitive and the supply of disposal services is perfectly elastic, all of the results derived for individual firms scale up to the market as a whole; the welfare measures per unit of output are simply multiplied by the number of units produced. Many disposal markets, however,

---

[21] Based on various EPA documents, Nichols (1984: ch. 9) estimates that the rule proposed in 1980 for maleic anhydride plants would have reduced benzene emissions by 5,059 MT and exposure by 1.1 million parts per billion per person per year (ppb-person-years). Using EPA's risk factor of 3.4×10^{-7} cases per ppb-person-years, that would imply an average risk per metric ton of:

$(3.4 \times 10^{-7}$ cases/ppb-person-years$)(1.1 \times 10^{6}$ ppb-person-years$)/5{,}059$ MT $= 7.4 \times 10^{-5}$ cases/MT.

may not fit these conditions, particularly in the short run. Although there do not appear to be any inherent technological reasons why disposal should not be competitive (with close-to-constant costs), regulatory hurdles and extraordinary siting difficulties make the supply of disposal less than perfectly elastic and also may contribute to the wielding of market power by suppliers of disposal services. Relative to a perfectly competitive disposal market with constant costs, either condition reduces the inefficiency caused by a failure of disposal regulations to impose the full social cost of disposal on generators.

The Impact of a Rising Market Supply Curve for Disposal Services

Our expressions for the potential welfare gains from additional waste minimization have assumed that the marginal cost of disposing of a unit of waste depends only on the risk level -- e.g., on the level of treatment and on the precautions taken to reduce leaks from disposal -- and not on the total amount of waste to be disposed. For the individual generator that is small relative to the market, this assumption seems perfectly reasonable; it is a price taker, so that variations in the quantity that it generates have no impact on its disposal costs. Once we scale up to the market as a whole, however, the assumption becomes less reasonable, particularly in the short run. The key reason to expect the supply of disposal services to be less than perfectly elastic is that the supply of disposal sites is limited.

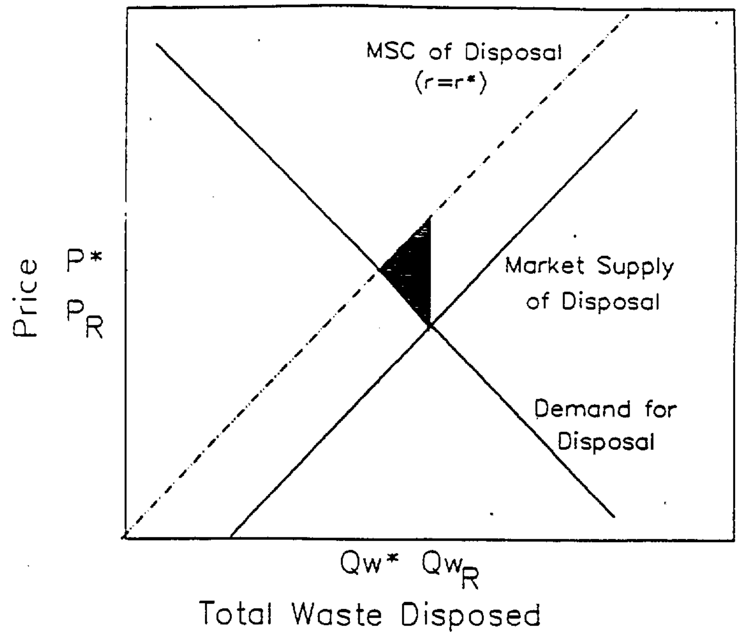


Figure 2-6. Waste Minimization with a Rising Supply Curve for Disposal Services

If the supply of disposal services is less than perfectly elastic, then the price of disposal will be a function not only of the risk level, but also of the total quantity disposed. Figure 2-6 illustrates the situation; it is similar to Figure 2-3, but the horizontal axis represents total waste, Qw, where Q is the quantity of output, rather than waste per unit of output. The market supply curve of disposal services reflects the stringency of disposal regulations. Unlike the supply curve facing an individual firm, however, it is likely to be upward sloping; at any given disposal-risk level, the price of disposal must rise to elicit additional supply. With a disposal standard that sets r = r*, but does not internalize the external damages, the equilibrium occurs at a disposal price of P_R, and the total quantity of waste is Qw_R.

The true social cost of waste, however, is higher than price along the market supply curve, again because of residual damages; the marginal social cost is simply the market supply curve shifted up by λr^*. If suppliers (or demanders) of disposal services internalized those residual damages, perhaps through a tax or tort liability, the quantity disposed would fall to Qw^*; i.e., generators would engage in additional waste minimization. The failure to achieve this optimal level of waste minimization causes an efficiency loss equal to the area of the shaded triangle. If, as before, we express that loss as a fraction of disposal costs and use linear approximations for the supply and demand curves, the algebraic expression for this efficiency loss is:

$$\Delta S/Qw_R \approx (\delta^2/2)\left\{\frac{\eta\epsilon_w}{\eta + \epsilon_w}\right\}P_R, \qquad (2\text{-}20)$$

where η is the elasticity of supply, Q is the level of output, P_R is the market price for disposal, and the other variables are as defined earlier in connection with Equation 2-19. Compared to the individual-firm case, the welfare loss is multiplied by the factor $\eta/(\eta + \epsilon_w)$; e.g., if the supply and demand elasticities are equal, the welfare loss is half as great as before. If supply is inelastic, the welfare loss is smaller yet; in the limit, if the supply of disposal is perfectly inelastic, there is no welfare loss at all.

In those cases in which the supply of disposal services is inelastic, supplementing disposal standards with taxes or tort liability to internalize residual damages will have little impact on allocative efficiency. If the demand for disposal is relatively elastic, such taxes will not increase the price paid by generators very much, so they will have little additional incentive to minimize wastes. The primary impact will be to lower the net price received by disposal facilities and to transfer rents from their owners to the government (in the case of taxes) or lawyers and the victims of the damages (in the case of tort liability). If demand is inelastic, the price of disposal will rise more, but the higher price will have little effect on waste minimization; a low elasticity of demand for disposal services reflects a limited ability of generators to reduce wastes.

Imperfect Competition

In many parts of the country, the difficulty of siting new disposal capacity is likely to confer substantial market power on firms with existing facilities. In some cases, such firms may have an effective monopoly on legal disposal. To the extent that disposal firms do wield market power, they will charge a price in excess of their own private marginal costs, and the price charged for disposal already may be in excess of the socially efficient level; imperfect competition in the supply of disposal services thus can lead to too much waste minimization, rather than too little.

Figure 2-7 illustrates this situation for a monopolist. A monopolist sets the quantity such that marginal revenue is equal to marginal cost. In Figure 2-7, demand is sufficiently inelastic that the monopolist's price is higher than the optimal price, including external damages. In such cases, imposing a tax on waste disposal would reduce net benefits by reinforcing already excessive incentives for waste minimization and driving the quantity of waste even further below its efficient level.

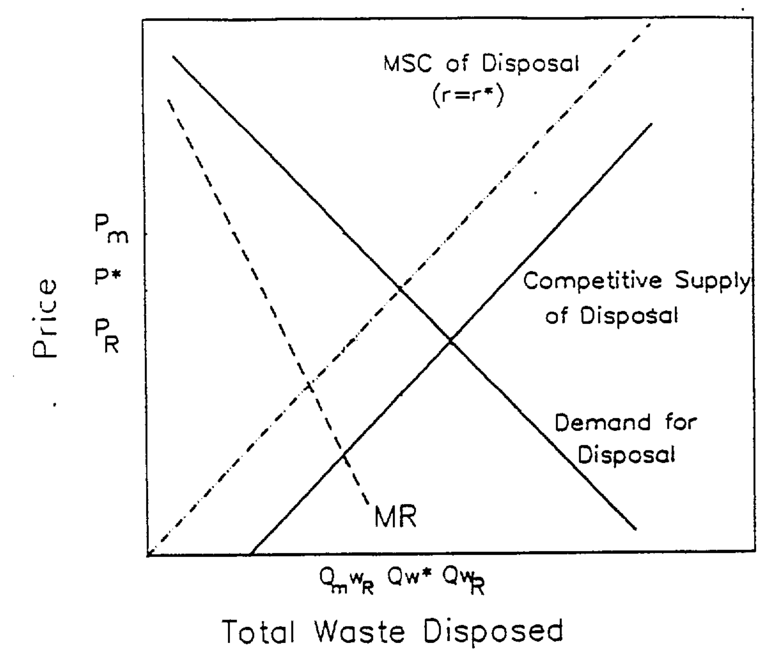


Figure 2-7. The Impact of a Monopoly in Disposal Services

In some instances, of course, the monopoly price may be below the true social cost; in such instances, a waste tax or liability can improve efficiency, though less so than in a competitive market. In those cases in which the industry marginal-cost curve is perfectly inelastic over the relevant range, the presence or absence of monopoly will be irrelevant; the quantity of wastes disposed will be invariant regardless of market structure and regardless of any taxes or liabilities imposed.

Existing Mechanisms for Internalizing Damages

Our calculations of potential gains from additional waste minimization have assumed that residual damages are not reflected in disposal prices and that firms do not bear any of those damages. In practice, however, several mechanisms act to cause generators to internalize at least a portion of residual damages. Taxes levied on disposal provide some internalization on an *ex ante* basis. Liability for cleanup and some times of damages should wastes pose a risk in the future provides an *ex post* form of internalization.

Taxes and Fees on Disposal

The EPA recently reported that 30 states impose taxes or fees on the disposal of hazardous wastes (Versar 1986a: Table 7-2). Although the motivation for these waste-end taxes and fees is to raise revenues to fund state hazardous waste programs, they do have the effect of internalizing at least some of the residual risk. California, for example, imposes fees that vary with waste characteristics. In 1984, they ranged from $2.68 per ton for mining waste to $59.40 for the most hazardous category; as percentages of commercial disposal prices, the fees varied from 8 to 20 percent (ICF 1985: Sec. 2.2.3). Our earlier calculations suggest that such rates are likely to be sufficient to fully internalize residual damages, at least as measured by likely health risks, for RCRA-approved disposal methods.[22]

[22] California's fee structure is based on relative pre-disposal risks, not the relative residual external damage subsequent to approved disposal. For the highest of these fees to be equivalent to an optimal tax, the residual external damage posed by approved disposal would have to be about 100 times greater than the EPA's upper-bound estimate of RCRA disposal risks, or cancer prevention would have to be valued at $100 million per case. We suspect that tax rates of this magnitude fully internalize residual damages, and then some.

Liability

The incentive effects of liability are more difficult to quantify, but probably are substantially more significant than those of existing waste-end taxes. Under CERCLA, virtually every person and entity engaged in the generation, transportation, treatment, disposal, or other handling of hazardous waste bears strict, joint, and several liability for the cost of cleaning up hazardous waste sites that pose risks. For some sites, these clean up costs have run to the tens of millions of dollars. Although the focus of CERCLA has been on correcting the results of mismanagement prior to RCRA, CERCLA liability nonetheless applies to RCRA sites, and thus serves as a mechanism for internalizing residual damages. Moreover, RCRA rules governing the detection and remediation of leaks (including monitoring after a site has closed) increase the probability that any future problems will result in liability for the firms involved.

In addition to cleanup costs, CERCLA also makes firms liable for damages to natural resources, such as the closing a fishery or a swimming area. Firms are not liable under CERCLA for damages to human health, although injured individuals may sue successfully in some cases under state statutes or common law.[23]

To the extent that TSDFs bear the expected cost of this liability, it is reflected in disposal costs; i.e., the price of disposal already

---

[23]Common law rules may allow suits for increased risk and fear of cancer, even in cases in which no physical damage can be documented. See, e.g., Sterling v, Velsicol, (W.D. Tenn. 1986), 647 F. Supp. 303, 17 Environmental Law Reporter 20081; and Siegel and Salvesen (1987) (arguing in favor of a rule that would allow such suits if one's probability of contracting cancer is increased by at least 10 percent).

incorporates at least some residual damages. Generator liability does not show up in the price of disposal, but does increase its effective cost. Although current disposal practices presumably are far safer and less likely to result in cleanup actions than the past behavior responsible for today's litigation, it is clear that liability is a major concern in the decisions that firms make regarding disposal and waste minimization. The seriousness with which market participants take this liability threat is indicated by the high demand for insurance, which unfortunately is generally not commercially available (Connolly 1987).

At least some generators impose internal fees on their operating units to account for potential liability. As of 1986, for example, Davidson Rubber, a manufacturer of plastic and urethane moldings for automobiles, assessed production departments $188 per drum (or over $800 per ton) of hazardous waste generated, apparently based on costs borne by the firm in cleaning up a CERCLA site.[24] More recently, Digital Equipment Corporation has estimated that, because of potential liability, the full cost of hazardous waste generation is about twice as high as the cash cost of RCRA disposal. Efforts are underway to make production departments internalize these costs, rather than carrying them in overhead.[25] Waste minimization is a particularly attractive

[24] Oral remarks by Robin Frank, Davidson Rubber, at "Hazardous Waste Disposal Management Conference" (Executive Enterprises), Washington D.C. (June 17, 1986).

[25] Oral remarks of James K. Rogers, Manager of Corporate Energy and Environmental Systems, Digital Equipment Corporation, at "Regional Environmental Regulation Conference" (Executive Enterprises), Boston Massachusetts (May 23, 1988).

method of reducing potential liability, because not generating waste in the first place is the only sure way to avoid future liability for cleanup costs.

Liability is likely to be of greatest concern to larger companies for two reasons. First, the possibility of bankruptcy is less effective in placing a ceiling on potential losses for large firms than for small ones. Second, under joint and several liability, a generator can end up paying for the entire cleanup of a site, even if it contributed only a small fraction of the waste, if the TSDF and other generators are judgment-proof because they have gone out of business or have insufficient assets; larger generators are more likely to remain in business and to have the "deep pockets" necessary to pay cleanup costs. As a result, for at least some firms, the expected liability cost of disposal may be well above the expected damages that its wastes impose. In such cases, the firm will have an excessive incentive to minimize wastes.

Conclusion

In the absence of government regulation or liability for disposal risks, not only is disposal excessively risky, but firms also have seriously deficient incentives to reduce the generation hazardous wastes because the private cost of disposal will fall far short of its social cost. Over the past decade, however, the disposal of hazardous waste has become much more stringently regulated. Standards regulating disposal substantially narrow the gap between private and social costs of waste generation by driving up private costs and by reducing external costs (by reducing risk).

At the same time, CERCLA, and other changes in liability rules have forced generators to internalize a much greater fraction of the risks their wastes impose. As a result, we find it hard to argue that generators face incentives for waste minimization that are significantly below their optimal levels, and that waste minimization requires special incentives or regulations.

If we think of waste minimization as part of a larger class of approaches to environmental problems based on "prevention" rather than end-of-pipe treatment, substantially stronger incentives appear to exist for hazardous wastes than for other types of environmental problems, such as air and water pollution. The typical air or water pollution standard provides little or no incentive to do better than the standard requires; there is no financial reward, for example, for the utility that modifies its boilers to produce less sulfur than permitted under the Clean Air Act. In contrast, because a generator must pay to dispose of each unit of hazardous waste, it faces strong, continuing incentives to minimize these wastes, incentives that persist right down to the point at which the firm generates no waste at all.

The incentives for waste minimization also are stronger than prevention-based approaches in other areas because the liability system plays so much larger a role in hazardous waste than with other environmental problems. The chemical company whose wastes later require cleanup, despite full compliance with disposal regulations, is very likely to bear the cost. In contrast, if that same company emits the same substances into the air in compliance with air-pollution regulations and those emissions cause adverse health effects, the probability of a successful suit against the company is practically nil because of the tremendous difficulty in proving causation.

We suspect that the reaction of many readers to this line of argument will be, "If the incentives are so strong, why do firms continue to generate such large quantities of waste?" We have several responses. First, it appears that firms are engaging in far more waste minimization than in the past; although reliable data do not exist to allow one to track aggregate amounts of waste with any confidence, there is considerable anecdotal evidence that many firms are devoting a great deal of attention to waste minimization.[26]

Second, in evaluating the effect of today's incentives, it is essential to recognize that firms cannot change their behavior instantaneously in response to price changes When prices change radically, it takes time for new technologies to evolve and for firms to learn of their availability. Government *may* be able to speed up these processes by supporting research and by helping to disseminate information. The evaluation of such strategies, however, is beyond the scope of this study.

Third, if new, lower-waste technologies are embodied in long-lived capital equipment, we would expect many firms to find it most cost-effective to delay switching to the new technology until their existing capital had reached the end of its useful economic life (which will, of course, come sooner with changes in relative prices). The sharp rise in energy prices in the 1970s provides an instructive analogy; early estimates of the price elasticities of demand for energy tended to be

[26] See, e.g., CEM (1985); CEM (1986); CEM (1987).

fairly low, and many observers bemoaned the inability or unwillingness of firms (and individuals) to reduce their energy consumption solely in response to higher prices. Over time, however, energy demand responded much more to higher prices, indicating that long-run elasticities were much higher than those in the short-run.[27] Similarly, we feel confident that even if disposal prices stopped rising, a great deal of additional waste minimization would occur in future years as a result of previous price increases.

Finally, we think it that it is impossible to say how much waste minimization is optimal in the abstract, without reference to the costs and risks of alternative approaches. If less waste minimization seems to be occurring than some observers think is appropriate, it may be a signal that waste minimization is more expensive than expected, rather than proof that incentives are incorrect or that firms are incapable of identifying and adopting cost-saving innovations.

Our skepticism about the need for regulatory programs targeted directly at waste minimization should not be misinterpreted as saying that waste minimization does not have an important role to play in reducing the risks posed by hazardous wastes. Rather, it says that firms already may have sufficient incentives to minimize wastes, at least to the extent that they comply with existing legal requirements. This last qualification is an important one, one that we explore at

[27]The results of several studies are summarized in Fisher (1981: 117-118). A sampling of these figures reveals the point asserted in the text. Paired estimates of short- and long-run electricity-demand elasticities for U.S. industry include (-0.14; -1.20) and (-0.06, -0.52). Estimated residential energy-demand elasticities display much greater differences. Estimates range from -0.12 to -0.63 for the short-run; -0.42 to -1.70 for the long-run.

length in subsequent chapters, for if firms do not comply with disposal regulations, there may be a strong rationale for greater efforts directed at waste minimization or for trying to develop ways of improving compliance.

Chapter 3:

"WASTE MINIMIZATION" THROUGH NONCOMPLIANCE

Disposal of wastes in conformance with RCRA rules has grown increasingly expensive. These rules, in concert with potential CERCLA liability, provide powerful indirect incentives for hazardous waste minimization. Compliance with RCRA rules also appears to reduce risks to low levels. As a result, the potential gains from additional waste minimization would appear to be minimal.

Our results in Chapter 2 were robust with respect to several important parameters, such as the unit cost of disposal, the level of residual risk, and the implicit value of life-saving. However, the analysis rested upon a critical assumption -- that firms actually comply with the rules.

Overview of Noncompliance

In this chapter we suggest that choosing not to comply offers an effective (albeit clandestine) way to minimize the amount and cost of RCRA-approved waste disposal. This characterization of noncompliance as an alternative form of "waste minimization" is not flippant; it simply recognizes that unless regulatory coverage is sufficiently inclusive and extraordinarily well-policed, evasion and noncompliance become financially attractive methods of appearing to achieve the letter (but certainly not the spirit) of public policy objectives. The parameter that regulators must look toward for evidence of waste minimization is a reduction in the amount and degree of hazard of manifested wastes. Unfortunately, declines in this statistic may instead demonstrate regulatory evasion and noncompliance, and it is quite difficult to discern which activity is actually taking place.

Motives for Noncompliance

Firms choose whether or not to comply with governmental dictates for many reasons. Compliance may result simply from tallying up private benefits and costs. For some enterprises, noncompliance may be easy to detect or penalties so substantial that it is unquestionably an inferior financial choice. Sanctions can be quite tangible, such as fines or potential CERCLA liabilities, or intangible, such as the risk of damaging one's personal or corporate reputation. For large firms with substantial capital, the prospect of unlikely but enormously expensive CERCLA cleanups may motivate both a high degree of voluntary compliance and extensive efforts to minimize future hazardous waste generation.

For other firms, however, it may be cheaper not to comply no matter what intangible costs are involved or how carefully they are weighed against benefits. The savings from regulatory evasion constitute their own reward: noncompliance reduces costs, thereby imparting an advantage over honest competitors and putting increased pressure on the latter to cheat as well. For industries in which noncompliance is the norm rather than the exception, ethical firms will be driven out of business or un-, able to enter because of their inability to compete.[1]

[1] Attempting to enter based on lower costs obtained from some other source besides noncompliance may be hazardous as well. For a riveting story of corruption and the establishment of noneconomic barriers to entry in the New Jersey waste carting business, see Block and Scarpitti (1985).

Small quantity generators (SQGs) may be especially inclined not to comply because they face both unusually high compliance costs and systematically smaller expected penalties for noncompliance.[2] The marginal cost of RCRA disposal declines sharply over a wide range of waste generation levels because there are substantial economies of scale in transport, chemical analysis, record keeping, as well as many of the approved disposal technologies.[3] Expected CERCLA liabilities may also be correlated with a firm's reachable assets, because regulatory enforcement tends to be directed toward large generators where the return to enforcement effort is the highest. Moreover, bankruptcy provides a ceiling on absolute liability that small firms are more likely to reach.[4] Each of these factors clearly weakens SQCs' incentives to comply.

Other identifiable classes of generators also may be particularly susceptible to noncompliance. It may even be the norm, for example, in competitive industries where proper hazardous waste disposal would consume a large fraction of earnings and thus force many firms out of business.

---

[2]In general, SQCs are firms that generate less than 1,000 kg of hazardous waste per calendar month. Firms that produce less than 100 kg are sometimes called very small quantity generators, or VSQCs.

[3]*See*, e.g., Schwartz et al. (1985).

[4]Expected liability also may be systematically higher for firms that comply. Compliance does not extinguish liability, and may instead enhance the government's ability to recover cleanup costs by providing a paper trail from a problem facility back to the generator. Not only is detection more difficult for wastes disposed of illicitly, but the task of tracing wastes back to their source can be both costly and problematic.

Noncompliance and the Hazardous Waste Policy Literature

Surprisingly little has been written about the problem of noncompliance in the many reports and analyses of hazardous waste minimization. A few sources offer anecdotal evidence of illegality, but typically the problem is assumed away. A common practice has been to emphasize technological factors and cost-effectiveness calculations using compliance as the relevant performance baseline.

Generally, noncompliance has been viewed as an enforcement problem rather than an opportunity for applying economic incentives. According to the conventional wisdom, it is best controlled by committing more resources to enforcement and imposing increasingly severe civil and criminal penalties. However, there are no credible estimates in the literature concerning how much noncompliance actually occurs or how much additional enforcement would be necessary to control it. Typically, noncompliance and illicit disposal receive scant attention. A few examples from this literature illustrate the point.

Prior to the establishment of its current hazardous waste minimization programs, the California Department of Health Services commissioned a report to consider whether additional economic incentives were needed (ICF 1985). In this report, the description of the existing hazardous waste problem failed to mention either casual noncompliance or illegal dumping (ICF 1985: Sec. 2.1.) Instead, the report emphasized policy alternatives designed to achieve additional waste minimization among those generators that already comply with regulations (ICF 1985: Sec 4.1). But as we indicated in Chapter 2, the potential social gains from such efforts seem likely to be small, even if they are obtained in an efficient manner.

California Governor Deukmejian subsequently empanelled a blue ribbon task force to develop a consensus strategy for dealing with problems posed by hazardous wastes. This task force produced an estimate of the State's hazardous waste generation that precisely equalled the state's estimate of the amount of wastes legally disposed, thus implicitly assuming away any problem with noncompliance or illegal disposal.[5] According to the task force, illegal disposal was an enforcement problem that resulted from the shortage of legal landfill capacity rather than any fundamental pattern of economic behavior:

> The same factors working for change in current land disposal practices could well prompt increases in illegal and unsafe dumping of hazardous wastes. Possible deterrents to this are stronger law enforcement and ensuring the availability of adequate opportunities for proper waste disposal. (California Task Force 1986a: 68.)

According to this view, hazardous waste generators can be divided into "good guys" and "bad guys." As long as there is adequate legal disposal capacity, irrespective of its cost, the "good guys" will refrain from dumping their wastes illegally (California Task Force 1986b: 30). Dealing with the "bad guys" is an enforcement matter that calls for better training of local law enforcement personnel and more funds to be devoted to enforcement (California Task Force 1986b: 160-161).

---

[5]The data reproduced in the task force's report is suspect for a number of other reasons, as well. In the report, California's total hazardous waste generation is estimated at 2,067,000 tons for 1984 (California Task Force 1986b: 28). Prior analyses placed hazardous waste generation in California between 15,000,000 (OTA 1983) and 17,284,000 metric tons (CBO 1985). The task force made no attempt to reconcile these inconsistent figures.

One particularly popular federal analysis of waste minimization is the report recently published by the Congressional Office of Technology Assessment (OTA 1986). Despite passionate concern for the risks posed by hazardous waste and sensible respect for the difficulties of establishing a program to regulate waste minimization directly, the problem of noncompliance receives very little attention. Instead, OTA emphasizes the need to make a transition from regulated hazardous waste management to absolute reductions in both the amount and toxicity of wastes generated. The OTA did not address the question whether more risk-reduction benefits could be obtained from this effort than from decreasing the proportion of hazardous wastes that are disposed illegally or improperly.

HSWA Section 8002(r) instructed the EPA to report to the Congress on progress toward waste minimization and to identify additional statutory and regulatory initiatives thought necessary or useful in this endeavor. In its report, the EPA generally treats noncompliance as an enforcement issue best combated through intensified inspection and monitoring of hazardous waste facilities. Illegal disposal is discussed only in the context of small quantity generators and firms for which regulatory compliance might be prohibitively expensive:

> The extent of current illegal disposal activities is a matter of speculation. . . As the cost of managing their hazardous wastes increases, many economically distressed firms may see illegal disposal as the only way to continue operating." (EPA 1987: 41).

Casual noncompliance and illegal dumping are both assumed to be occasional aberrations from the regulated hazardous waste management system rather than systematic cost-minimizing responses.

Several studies suggest that noncompliance is more systematic and serious than the conventional wisdom would appear to suggest. In a study of small quantity generators in California that relied on interviews with government officials, trade association executives, and waste treatment, recycling and disposal firm personnel, Schwartz et al. (1987) concluded that illicit disposal was actually far more widespread than had been acknowledged in the literature. Because illegal operatives face minimal threats from enforcement, the report recommends the development of low-cost alternatives to illegality (Schwartz et al. 1987: 20).

More recently, Hammitt and Reuter (1988) have conducted an exploratory study of illegal disposal *per se.* The report focuses on firm and industry characteristics that might be used to better target regulatory enforcement resources, recognizing that high compliance costs lead economically-motivated actors to at least consider if not adopt a range of noncomplying behaviors. Enforcement is conceptualized in the report as an optimization problem in which the marginal social benefits from enforcement can be equated with the marginal social costs. The authors make no claim that the current level of enforcement is necessarily suboptimal, inasmuch as data concerning the efficacy of enforcement are unavailable, but instead offer recommendations for improving the allocation of enforcement resources so as to increase the effectiveness of the existing enforcement effort.

Both of these reports suggest that the existing regulatory machinery may suffer from a great deal more noncompliance than has been generally admitted. This means that regulatory initiatives premised upon full compliance may well be ill-advised. In the remainder of this chapter, we offer a glimpse into some of the many ways that waste generators may legally or illegally circumvent or evade RCRA and thus fundamentally alter both the optimal regulatory design and the nature of any additional economic incentives that might be warranted.

Firms Legally Outside the RCRA System

RCRA governs a very broad range of residuals from economic activities. Indeed, much of RCRA's complexity stems from this very attempt at comprehensiveness. Nevertheless, certain types of wastes and waste generators are exempted from coverage by statute or administrative rule, or subjected to less than full regulatory control. Moreover, there is also substantial overlap between RCRA and other environmental statutes, such as the Clean Water Act (CWA), Clean Air Act (CCA), the Safe Drinking Water Act (SDWA), and the Toxic Substances Control Act (TSCA). In this section we discuss just a few of the many ways that firms can legally avoid RCRA's full force.

Statutory and Administrative Exclusions

RCRA does not apply to certain waste streams because they are not "solid wastes" as defined by statute or regulation, such as domestic sewage and industrial discharges permitted under the National Pollutant Discharge Elimination System (NPDES). Other wastes are considered "solid wastes" but are exempted from RCRA, including household wastes, irrigation return flows, mining wastes, utility fly ash, and muds, from oil and gas drilling. In addition, the EPA has granted exemptions to a variety of specialized industrial processes. In 1980 the EPA administratively established an exclusion for wastes generated by SQGs. Thus, from the outset the universe of regulated parties has been necessarily incomplete.

Small quantity generators (SQGs). Large quantity generators (LQGs), firms that generate more than 1,000 kg of hazardous waste per calendar month, were required to notify the EPA of that fact during 1980. SQGs were initially exempted from this requirement. As of 1982, the EPA had identified approximately 14,000 LQGs which were to be subject to RCRA Subtitle C.

Prior to establishing the SQG exemption, the EPA believed that there were about 695,000 SQGs producing about one percent of the total waste volume (Abt Associates 1985: 31). Thus, the EPA believed that because of their numbers, including SQGs in the regulatory system would overwhelm the Agency's monitoring and enforcement capabilities while producing but a tiny increase in potential risk reduction.[6]

These concerns seem well-founded. According to the small quantity generator survey, which was in progress when Congress deliberated on HSWA, the SQG problem is probably smaller than earlier data had suggested. The survey estimated that only 0.4 percent of the total waste volume was generated by approximately 630,000 SQGs. The remaining 99.6 percent of the waste volume was produced by the 14,000 or so firms already subject to full Subtitle C regulation (Abt Associates 1985: 29).

[6]Some economists argue that such regulatory "tiering" can be justified on efficiency grounds, even in the absence of monitoring and enforcement costs, by virtue of the high fixed costs of regulatory compliance. See, e.g., Brock and Evans (1985a).

Congressional concerns about the environmental significance of SQG wastes, even when disposed contrary to RCRA rules, have thus not been substantiated.[7]

Despite this apparent asymmetry, Congress nevertheless instructed the EPA to commence regulating SQGs, which the Agency did on July 15, 1985 (50 FR 28743). These regulations established three classes of SQGs, with relaxed reporting requirements relative to the full RCRA system, but similar rules concerning the ultimate disposition of hazardous wastes. Qualification for SQG status occurs monthly, so many firms may be coming in and out of the system at irregular intervals. Thus, the size of the EPA's regulated community is constantly in flux. Self-certification of SQG status, although ultimately subject to EPA inspection and monitoring, means that the EPA cannot be sure which firms are subject to its regulatory requirements and which are not.

The strength of RCRA as an incentive for waste minimization varies somewhat depending on whether a firm is potentially eligible for the relaxed SQG rules. Clearly, SQG status confers cost advantage over full RCRA coverage, so firms near the threshold will be motivated to stay below it if at all possible. In the short-run, the SQG threshold limits these firms' production volume, because marginal cost will rise dramatically if the threshold is crossed. In the long-run, of course, firms can make source reduction investments that raise the output level corresponding to the SQG threshold.

---

[7]An excellent example of Congressional concern is found in Harris, Want and Ward (1987: ch. 10). The available data confirm that illegal or environmentally improper disposal methods dominate among SQGs (GAO 1983a; Abt Associates 1985; Schwartz et al. 1985). However, completely eliminating hazardous wastes generated by SQGs would reduce the known quantity of wastes by less than one percent.

Because wastes are counted at the plant level, another way to expand production while remaining below the SQG threshold is simply to build another **plant**.[8] If multiplant expansion is cheaper than waste minimization, or creates significant additional benefits apart from reducing hazardous waste disposal costs, then the SQG threshold will reduce the optimal plant size without having much affect on the amount of waste generated.[9]

Overlapping coverage with other environmental statutes. Environmental regulation has traditionally emphasized end-of-pipe pollution control strategies focused on a single medium such as air or water. Hazardous waste regulation has evolved from a similar root -- statutes and programs emphasizing the management of solid waste **landfills**.[10]

[8]**A** plant generating less than 1000 kg per calendar month qualifies as an SQG even if it is one of many identical plants owned by the same firm.

[9]**There** is a continuing debate concerning the effect of environmental regulations on optimal plant size, and the implications of such an effect upon industry and firm structure. E.g., Pashigian (1984) argues that high compliance costs have driven small firms out of business and increased optimal plant size. In contrast, Evans (1986) disputes Pashigjan’s evidence as inconclusive, and argues based on other data that regulatory “tiering” and exemptions -- such as the EPA’s original SQG exclusion -- have reduced or eliminated economies of scale in compliance with environmental regulations.

[10]**An** important vestige of this history is that "hazardous wastes" are defined differently by the RCRA statute and by the EPA’s implementing regulations. In general, hazardous wastes appear to be a subset of “solid wastes," but the definitions are somewhat murky. "Solid waste” includes materials that are not "solid" at all, but liquid or gaseous, but definitely wastes and unquestionably hazardous. Domestic sewage is exempt from RCRA, however, and thus is not a "solid waste" under RCRA. Wastes that are beneficially recycled were exempt from RCRA under the EPA's pre-1985 definition, but since that date have become “solid wastes”. For an incisive look at this bewildering situation, see Garelick (1987).

However, the problems addressed by RCRA in its current formulation require a more comprehensive approach. Hazardous wastes can appear in many different forms, and focusing entirely on a single environmental medium (e.g., land) could allow risks to be simply transferred to other media where regulatory controls perhaps are less effective (e.g., air or water).

Difficulties managing this complex regulatory program, in which residuals can be transferred across environmental media to exploit relative weaknesses in regulatory control, offer an additional rationale for encouraging waste minimization *per se.* The question, of course, is not whether inter-media transfers should be discouraged, but rather what instruments seem most capable of achieving this objective.

Wastes discharged subject to NPDES permits or sent to POTWs. According to the most recent hazardous waste generation survey performed for the Chemical Manufacturers Association (CMA), more than 210 million tons of hazardous waste generated by survey respondents in 1985 were wastewaters discharged subject to a NPDES permit, sent to a publicly-owned sewage treatment facility (POTW), or disposed by underground injection. This represents more than 90 percent of the waste generated by these firms,[11] and about 70 percent of all hazardous waste believed to

[11] See CMA (1987: 26). A total of 77 CXA member companies in SIC 2800 covering 681 plants responded to the survey (CMA 1987: 19). Although this includes 36 of the top 50 and 21 of the top 25 chemical firms (ranked by sales), the 210 million ton figure is necessarily incomplete because it represents the total from responding firms rather than an industry-wide. estimate derived from the survey.

be generated.[12]

Despite the relatively large volumes of waste involved, RCRA's application to NPDES and POTW discharges is at best uncertain because they are regulated primarily under the Clean Water Act (CWA).[13] Thus, RCRA's effectiveness is to a significant extent controlled by regulators elsewhere in the EPA who are operating subject to different statutory authorities.

Concerns about the amount of hazardous waste discharged through the NPDES and POTW systems have been raised by various sources. In 1983, the General Accounting Office reported a high rate of noncompliance with existing NPDES permits, and suggested that the EPA's administration of the program was ineffective (GAO 1983b). Similar problems have been alleged to exist with respect to municipal sewage treatment (Drayton 1984: 40). In HSWA, Congress ordered the EPA to study the domestic sewage exemption and report back by February 1986, and issue necessary regulations by October 1987. In its report, the EPA concluded that existing regulatory programs were sufficient to control these risks provided that the rules were enforced, and recommended against rescinding the domestic sewage exemption (EPA 1986c).

[12] Abt Associates (1985: 29) estimated total hazardous waste generation at 265 million metric tons, or 291 million English tons. The CMA survey uses English tons.

[13] Underground injection is regulated under the Safe Drinking Water Act (SDWA) according to the EPA's Underground Injection Control (UIC) permit system. However, Congress used HSWA to explicitly extend RCRA to underground injection (RCRA Sec. 7010), and banned the underground injection of hazardous wastes as of May 8, 1985. Moreover, the HSWA established new authority under Subtitle I ("eye") to regulate underground storage tanks (USTs) so as to prevent them from leaking and thereby becoming LUSTs.

If RCRA and CERCLA do provide strong incentives to reduce reliance on land-based hazardous waste treatment and disposal options and thereby stimulate waste minimization, it is reasonable to expect that some wastes will be shifted to these water-based disposal options if it is technologically feasible to do so and difficult for regulators to prevent or detect. Waste minimization thus takes on added significance in such cases because prevention is the only sure way to avoid inter-media transfer.

Wastes disposed by air emission. RCRA's regulatory objectives also overlap with certain parts of the Clean Air Act (CAA). In particular, CAA Section 112 requires the EPA to promulgate standards for hazardous air pollutants. So far, the Agency has issued only a handful of standards (Haigh, Harrison and Nichols 1984), and its capacity to enforce them is unclear. Given the extent to which high disposal costs and potential liability discourage traditional land-based treatment and disposal, the incentive to vent volatile toxics to the air has clearly increased. Like the case of wastewater discussed above, actions that look like waste minimization may be cross-media, transfers instead.

Firms Illegally Outside the Regulatory System

In the previous section we have identified three major ways in which firms legally may generate or dispose of wastes outside of RCRA rules. In this section we discuss illegal variations on the theme of noncompliance, focusing primarily on relatively stark examples.

Obviously, firms and individuals that generate, transport, or dispose of hazardous wastes, but are unknown to either the EPA or the relevant state authorities, will not be much affected by regulatory incentives for waste minimization. The regulatory program is relevant to them only insofar as it devotes resources to detecting illegal activity and imposes significant civil and/or criminal penalties.

The Economics of Illegal Disposal

The demand for illegal hazardous waste disposal depends primarily on the asymmetry between disposal costs in the legal and black markets. As we indicated in Chapter 2, the cost of legal waste disposal is high because of the technological requirements of RCRA and the potential liabilities of Superfund. The costs of illegal disposal, of course, are much lower. Technology is simple and requires no special skills or expertise, and the importance of criminal sanctions is diluted considerably by the infrequency with which they are applied.[14] The conventional wisdom associates illegal dumping with fly-by-night operators and SQGs (Harris, Want and Ward 1987: 129; SQG Hearings 1983), but apparently many environmental attorneys find little reason to presume such a correlation (Ward 1983). In any event, Superfund liability may be a smaller risk for firms engaged in illegal disposal. The task of detecting contamination seems likely to be difficult without probable cause for looking for it. Even when dumping is discovered, tracing wastes from this destination to their source is quite difficult barring incompetence

[14]Reuter (1984: 36) suggests that the combination of low technology, minor scale economies, low entrepreneurial status, homogeneity in the product, and local markets all favor the development of racketeer-controlled cartels. These features unquestionably characterize illegal hazardous waste disposal.

by the dumper or inside **information.**[15]

As the regulatory system becomes more stringent in its requirements, legal disposal costs continue to rise but the costs of illegal dumping remain essentially unchanged. If demand is at all responsive, tighter regulation must lead to more illegality. Similarly, changes in CERCLA liability rules, settlement procedures, and site cleanup objectives that increase the expected cost of Superfund, also widen the gap and therefore stimulate more illegal disposal. If illegal disposal is predominantly motivated by economics, then controlling it requires that this gap be narrowed.

Traditional Law Enforcement Remedies

The usual response to criminal behavior is through law enforcement. That is, government seeks to reduce illegality by raising its cost relative to legal behavior. In effect, law enforcement approaches attempt to shift inward the supply curve for illegality, either by increasing the size of the penalty or raising the probability that is will be imposed.

Typical of such law enforcement efforts is the EPA's Criminal Enforcement Division, which was established within the Office of Enforcement and Compliance Monitoring in 1982. Until 1984, however, the EPA's

---

[15]**It** is important to separate detection from attribution because some dumpers apparently desire the authorities to discover their actions, at least after the fact. An example of this behavior can be found in the abandonment in vacant lots or along roadsides of (apparently) intact drums of hazardous waste. According to Hammitt and Reuter (1988), the incidence of such abandonments appears to be increasing, at least within Los Angeles County. The number of cases handled by the Sanitation Bureau of the City of Los Angeles has risen from nine in all of 1985 to 27 during just the first half of 1987.

criminal investigators lacked full law enforcement powers necessary to execute search warrants, tap into state criminal data bases and law enforcement radio networks, make arrests, and carry firearms. They had to rely upon the U.S. Marshal's Service, the FBI, and state and local law enforcement agencies whenever such powers were required.[16] Initially, the Criminal Enforcement Division hired 23 experienced criminal investigators and has continued to grow,[17] but the number of investigators is small in comparison to other federal agencies whose law enforcement responsibilities are also somewhat tangential to their principal organizational missions.[18] Budget constraints have prevented the EPA's National Enforcement Investigations Center (NEIC) from investigating about three-fourths of the credible allegations of criminal conduct received

[16] For an extensive discussion of the issue of EPA law enforcement, see Law Enforcement Hearings (1984). In 1984, EPA criminal investigators were deputized as Special Marshals by the Attorney General.

[17] Harris, Want and Ward (1987: 248) reports 35 criminal investigators at the EPA as of 1986. Note that these investigators are responsible for enforcing the EPA's full slate of environmental laws, not just cases involving RCRA.

[18] For example, in FY 1983, the Department of Justice fielded approximately 11,381 sworn law enforcement officers (FBI [7,500], Drug Enforcement Administration [2,261], and U.S. Marshals Service [1,620]). Other agencies whose duties are primarily law enforcement but are not part of DOJ include the Coast Guard (28,087), the Secret Service (2,729), the Customs Service (2,175), and the Capitol Police (1,222). Other agencies whose primary responsibilities are, like the EPA, not law enforcement but nevertheless employ sworn officers include the Postal Service (4,205); the Internal Revenue Service (3,315), the General Services Administration (2,366), the Forest Service (402), and the Federal Aviation Administration (127). Data for the Department of the Interior, (i.e., the National Park Service, the National Park Police, and the Fish and Wildlife Service) and the Department of Housing and Urban Development (i.e., the Federal Housing Administration) were unavailable. See Law Enforcement Hearings (1984: 123-127).

(GAO 1985).[19]

Hammitt and Reuter (1988) report that within Los Angeles County, the total number of personnel devoted to hazardous waste law enforcement is about 25, about three-fourths of them sworn law enforcement officers. The California Department of Health Services fields ten criminal investigators statewide. Pennsylvania's Toxic Waste Investigation Program houses three attorneys and nine investigators, and no additional local enforcement personnel. The Massachusetts Department of Environmental Quality Engineering (DEQE) employs about 30 investigators and inspectors, but responsibility for criminal enforcement rests with the six environmental police officers employed by the Division of Environmental Law Enforcement. Although this unit is supervised by the Attorney General's office, it is funded through the Department of Fish and Wildlife. Thus, hazardous waste law enforcement must compete with hunting, fishing, and boating programs for its budget. Officers work traditional day shifts; overtime is severely limited and they lack the communications equipment necessary to operate between sunset and sunrise or communicate with other law enforcement agencies, such as the state police.[20]

Effects of law enforcement. Estimating the actual impact of criminal sanctions is difficult for many reasons, not the least of which is that the baseline level of illegal activity is unknown and probably impossible to determine. Nevertheless, we can suggest concerns based on

[19]Goldman, Hulme and Johnson (1986) contends that the effectiveness of the EPA's National Environmental Investigations Center (NEIC) as a deterrent is limited because its operations are not visible.

[20]Information from personal communications with Massachusetts Environmental Law Enforcement officers.

the structure of existing incentives. First, criminal sanctions must be discounted according to the combined probabilities of detection, apprehension, and successful prosecution. Site access, which at times has been troublesome for regulatory enforcement personnel, is problematic in the criminal context. Apprehension may involve timely response as well as painstaking case preparation. Prosecutors vary in their concern for environmental crimes, and they always have many competing demands from outside agencies solicitous of their services.

Second, all sanctions are not equally effective. The threat of extraordinarily large fines, for example, deters only those firms and individuals who possess sufficient resources to pay them.[21] Illegal dumpers, however, seem unlikely to be so heavily capitalized. Moreover, as criminal sanctions become more severe, the probability of their actual imposition tends to decline. Evidentiary standards and due process requirements escalate and plea-bargaining becomes more attractive to the prosecution as a means of reducing administrative costs.[22] Judges who encounter a broad range of criminal cases may view RCRA's high statutory penalties as an aberration from normal criminal justice practice and

[21]RCRA Section 3008(d) identifies criminal violations. Maximum penalties for first time offenses are $50,000 per day of violation and five years' imprisonment. These penalties double for second and subsequent convictions.

[22]RCRA Section 3008(e) provides maximum penalties of 15 years imprisonment and a $1 million fine for "knowing endangerment." According to Riesel (1985), only one indictment had been sought under this provision as of 1985, and even this single action remained unresolved because the defendant was a fugitive.

refuse to impose them even against defendants committing the most egregious hazardous waste-related crimes.[23]

Third, the returns to law enforcement inevitably decline at the margin. Resources tend to be allocated to the easiest cases first because they are the least expensive to prosecute.[24] Difficult cases consume more resources and offer lower expected returns. Dumpers exacerbate this problem by becoming increasingly circumspect as the risk of punishment rises. The deputization of EPA investigators, combined with the anticipated use of informants and undercover operations, intensifies the risks faced by illegal operatives, who may respond by imposing new costs on the enforcers.[25]

Finally, the waste disposal industry In general has long suffered from a notorious reputation for racketeer influence or control, phenomena that seem plausibly related to illegal activities. In some regions of the country, this opprobrium has been richly deserved. In the Greater New York area, for example, firms that initially provided hazardous waste disposal services were begotten of the more commonplace

---

[23]For a contrary example involving a relatively stiff sentence, see Environmental Forum (1983).

[24]Obviously, it is also vitally important to win. Pursuing the most difficult cases first risks losing, and the signal received by criminals could be worse than if there was no enforcement at all.

[25]EPA investigators testifying in favor of full law enforcement authority cited numerous instances,in which enforcement actions were impeded or prevented by threats of violence against them. Clearly, the EPA's criminal investigations cannot be very aggressive in the absence of law enforcement authority. However, the risk of violence faced by investigators may not necessarily decline once investigators have been deputized: investigations that used to be thwarted by mere threats may now require more persuasive approaches. See Law Enforcement Hearings (1984).

trade of solid waste disposal, or "carting." According to both popular and scholarly accounts, organized crime has dominated the carting business for decades, and apparently is immune to control through either regulatory or law enforcement efforts.[26] Perhaps more importantly, the very fact that hazardous waste disposal evokes an unsavory taste discourages some responsible firms from entering the business. As Reuter (1984: 36) writes with respect to the New York carting trade, prosecutions enhance the notoriety surrounding the industry, which further enhances the powers of the racketeers and dissuades honest firms from getting involved. In short, to the extent that organized crime is part of the illegal disposal equation, law enforcement will probably be no more effective against illegal hazardous waste disposal than it has been in other areas.

Evidence of Noncompliance

Given the hazards of empirical research in illicit markets, it is not surprising that the available evidence of the extent of illegal hazardous waste activities is fragmented and anecdotal. Nevertheless, con-

[26]Reuter (1984) contends that the New York City Department of Consumer Affairs, which has had regulatory responsibility for carting since 1956, has inadvertently helped rather than hindered efforts to maintain mob control. Block and Scarpitti (1985) allege that on numerous occasions public officials and regulatory authorities have *intentionally* impeded law enforcement operations directed against mob-controlled illegal hazardous waste dumping.

terns about illegal disposal have arisen in many contexts.[27]

Obviously, both the EPA and the Department of Justice have felt that illegal hazardous waste disposal constituted a problem worth pursuing. In concert with the Environmental Crimes Unit within the Land and Natural Resources Division of the Department of Justice, the EPA's Criminal Enforcement Division produced 40 indictments and 29 convictions in FY 1983. Absent any benchmark for comparison, we cannot tell whether these numbers are large or small, nor if they represent success or failure.[28] However, investigators have testified that these prosecutions represent only a small fraction of ongoing illegal disposal activity. Judgments such as this must be interpreted with care, of course, because criminal investigators have a professional stake in discovering large numbers.

Much of the anecdotal evidence of illegal disposal has occurred in the context of SQGs. Indeed, Congressional action to eliminate the SQG exemption was primarily motivated by the widespread belief that it was resulting in significant environmental harm (SQG Hearings 1983; GAO 1983a). According to the EPA's SQG survey, public sewers and solid waste landfills are the most popular hazardous waste disposal methods

---

[27] *See*, e.g., Epstein, Brown and Pope (1982) (a popular expose of hazardous waste issues); Drayton (1984) (an environmentalist critique of the EPA's regulatory performance); Block and Scarpitti (1985) (alleging extensive organized crime infiltration in the hazardous waste transport and disposal business); Schwartz et al. (1987) (reporting interviews suggesting widespread illegal disposal among small quantity generators); and Hammitt and Reuter (1988) (describing criminal enforcement practices in selected jurisdictions).

[28] *See* Law Enforcement Hearings (1984: 93), Riesel (1985: 10066). Neither source reveals whether the remaining 11 indictments were still pending or had resulting in acquittals -- a crucial distinction for making even a cursory evaluation of enforcement efficacy.

(Abt Associates 1985: 41). More recent evidence from California -- a state that did not adopt the federal SQG exemption -- suggests that a high rate of SQG noncompliance still **exists.**[29]

SQGs are not alone in practicing illegal disposal, of course. For example, on November 9, 1987, the EPA announced a consent agreement with Texas Eastern Transmission Corp. in which the firm agreed to pay cleanup costs expected to reach $400 million plus a $15 million fine. The natural gas pipeline company, which earned $1.3 billion in 1986, allegedly disposed PCB-contaminated oil in 89 illegal pits located in 14 states from Texas to New Jersey. According to EPA investigators, this case is but the initial result of a wider probe into PCB dumping by the interstate gas pipeline industry (Boston Globe 1987: 3). If enforcement resources are directed first towards higher-valued cases, then the Texas Eastern case may not be an isolated instance, but rather the easiest of several potential cases to prosecute. This would imply that illegal disposal by even very large firms is far more extensive than previously **believed.**[30]

---

[29] **See** Schwartz et al. (1987: 15). This study cites results from two regional government surveys which are revealing. In a Southern California survey, as much as 20 percent of liquid hazardous wastes generated by SQGs were believed to be disposed illegally. A San Francisco Bay Area SQG survey reported that 57 percent of the respondents practiced some form of illegal disposal. Perhaps most ominously, 34 percent of the respondents were unwilling to pay anything for a legal alternative, and another 18 percent were not willing to pay more than $25 per month. The study also reports claims made by the District Attorney for Santa Clara County (San Jose area) that more than half of the 2,000 auto repair facilities in the county dispose of their hazardous wastes illegally.

[30] **An** alternative interpretation is that the EPA selected the Texas Eastern case for prosecution because of the size of the firm involved and the deterrent effect that would result from making an example of it.

As this case demonstrates, traditional law enforcement methods certainly have their place in detecting and punishing egregious regulatory violations. They are not a panacea, however. Given the number of potential violators and the seemingly unlimited ways in which illegal hazardous waste disposal can be conducted, criminal enforcement seems unlikely to "solve" the illegal disposal problem. As long as it is driven by economic incentives, law enforcement methods are limited in their ability to overcome viable illegal markets.

Noncompliance Among Firms Inside the Regulatory System

Regulatory noncompliance can occur at each stage of the hazardous waste management system. Generators may break the rules by improperly identifying, handling, or packaging wastes for subsequent treatment or disposal, or by designating an inappropriate destination. Transporters may fail to properly maintain vehicles and records, ensure driver qualifications and training, or follow correct operating procedures. TSDF operators may dispose of wastes lacking proper documentation, or otherwise violate the terms of their RCRA permits. For any party seeking to monitor these actors' performances-- private or governmental -- the ability to detect regulatory noncompliance is imperfect at best. It is also quite expensive because of the complexity of the regulatory requirements and the mounds of evidence comprising the audit trail.

The Uniform Manifest is the foundation of this informational system. In theory, maintaining a comprehensive "cradle-to-grave" inventory of hazardous waste prevents their improper disposal. The waste generator identifies the waste to be shipped, documents its composition and concentration, and directs the transporter exactly where it is supposed to go. The transporter then verifies the accuracy of the information provided by the generator and follows the generator's instructions precisely. Upon delivery, the TSDF operator confirms that the waste shipment is as advertised, disposes of it as instructed by the generator and returns a copy of the manifest to confirm that the job has been completed.

Unfortunately, the manifest system suffers from two critical problems that have yet to be resolved. First, despite the intentions of its inventors the accuracy of data recorded on hazardous waste manifests is not independently confirmed. Second, manifest data are typically not collected or maintained in ways that are conducive to effective regulatory oversight.[31]

Erroneous Manifest Information

It is expensive to verify the accuracy of the manifest every time a shipment of waste changes hands. Amount, composition, and concentration cannot be discerned by simple inspection, and conclusive proof may require extensive chemical analysis that is either costly or time-consuming or both. Thus, transporters and TSDF operators have incentives to rely upon generators for most of this information, and the independent verification that was expected to occur does not happen in practice. At best, information supplied by the generator is transmitted

[31] A third problem, mentioned only implicitly in the previous section, is that the manifest system cannot document wastes that never enter the RCRA management system. This has led at least one observer to characterize the manifest approach as merely a "hearse-to-grave" rather than "cradle-to-grave" system.

correctly through the system.

Besides the lack of independence, opportunities abound for manifest data to become willfully or unwittingly corrupted. As the number of participants and the amount of paperwork rise, the likelihood of errors inevitably increases as well. These factors suggest a significant capacity to conceal intentional misrepresentation amidst inadvertence.

Inadequate oversight of manifests and reports. Every person or firm that handles a given shipment of hazardous waste is supposed to retain a copy of the manifest for at least three years. Many states that operate EPA-authorized RCRA programs insist on receiving a copy as well. This creates a veritable avalanche of paper. In Massachusetts, for example, more than 7,400 manifest reports are submitted to the state each week -- a volume that severely taxes the state's regulatory resources, prevents timely data entry, and makes analysis problematic.[32]

Generators are required to file exception reports if they fail to receive a copy of the manifest confirming receipt by the TSDF within 45 days after shipment. However, neither the EPA nor the states have any way to ensure that these exception reports are actually submitted.

The other significant informational requirement is that generators are required to submit biennial reports to the EPA (or the authorized state program) on March 1 of each even-numbered year covering hazardous waste management activities occurring during the preceding odd-numbered

[32] **See** Massachusetts Department of Environmental Management (1987: 4), noting several serious problems that plague the state's computerized manifest system. Massachusetts officials concede that "data currently available and presented [in their report] do not accurately reflect the true quantity of manifested waste." Confidentiality protections limit the ability of outside researchers to gain access to the data.

year. In theory, these reports give regulators a complete summary of hazardous waste flows for every other year. Experience has shown, however, that these reports have little value because of the gaps in reporting and the absence of a standardized reporting format (Massachusetts Department of Environmental Management 1987: 4).

So far, regulators have been generally unwilling, or unable to utilize this information base. The conventional wisdom is that the information contained in these reports is suspect. Interstate variations in reporting requirements, formats, and data structures have effectively frustrated comparison and analysis. Problems such as double-counting, incomplete reporting, and failure to even submit the reports are apparently extensive (Massachusetts Department of Environmental Management 1987: 4). Once a credible case has been developed to support the allegation of a violation, data from manifests and biennial reports may offer useful corroboration, but they have little value as instruments for discovering noncompliance.

Generator noncompliance. Because transporters and TSDF operators rely upon generators for manifest information, generators have considerable latitude in deciding how to classify waste and how much of it to report. Even if transport and disposal costs are fixed, potential future liability creates an incentive to misrepresent waste types and understate volumes. Transporters and facility operators are unlikely to be able to detect small deviations, and even large discrepancies may go unnoticed if detection requires sophisticated equipment or methods

The exception reports referred to earlier make generators responsible for keeping track of their wastes. However, it is unclear whether a low rate of exception reports implies a high level of compliance or a widespread failure to report exceptions.

Perhaps the most significant opportunity for generator noncompliance arises from on-site waste management subject to a valid RCRA permit. Manifests are not required for on-site disposal (although biennial reports are), but there is no opportunity -- much less incentive -- for independent confirmation of data accuracy. Given the recent emphasis on waste minimization, the regulator is hard-pressed to discern socially beneficial reductions in waste generation from creative bookkeeping.

Transporter noncompliance. RCRA regulates transporters almost as extensively as TSDF operators; one reason for this degree of control is that the practice of "midnight dumping" originated with unscrupulous haulers who typically disposed of hazardous wastes along road sides, in vacant lots, and in abandoned buildings. Transporters need EPA identification numbers but do not need permits, and haulers of SQG wastes are excluded from coverage. In general, hazardous waste transporters must comply with regulations promulgated by the Department of Transportation (DOT) under the Hazardous Materials Transportation Act (HMTA), maintain records of source and delivery points for all wastes they handle in accordance with the RCRA Uniform Manifest system, and transport wastes only to RCRA-approved facilities.[33]

---

[33]Transporters must also comply with RCRA generator requirements when, for example, they clean hazardous sludge from tank cars, rail cars, ship holds, and pipelines, because the act of removing wastes from these enclosures constitutes "generation" under RCRA regulations. When multiple parties are involved in emptying, removing, or cleaning vehicles or vessels, all become hazardous waste generators, and will be held jointly and severally liable for violations of RCRA rules. If a transporter-generator stores, treats, or disposes of wastes removed from cleaning, it becomes a TSDF operator as well. Separate EPA identifica-

The manifest system is the critical feature of RCRA transporter regulations. Deviations from the procedures and destination required in the manifest are prohibited; a transporter cannot legally redirect wastes, even when circumstances prevent the transporter from making the delivery prescribed by the manifest.

In theory, transporter behavior is monitored by generators at one end, and by TSDF operators at the other. Generators cognizant of potential liability will police their side of the transaction, and refuse to do business with transporters that fail to abide by the rules. Similarly, TSDFs will oversee the destination side of the exchange and refuse to accept wastes that are improperly labeled, packaged or manifested, or arrive in unsuitable vehicles. Thus, as long as the manifest system is followed carefully, transporters will have strong incentives to comply with RCRA and HMTA rules.

An additional incentive for transporters. to comply is the threat of entry, primarily in the form of vertical integration. Transport technology is not particularly complex, and joint, and several liability makes it risky for generators to rely upon independent haulers that lack substantial reachable assets. For similar reasons, TSDF owners may also develop their own transport capacity.

TSDF noncompliance. Wastes may be disposed at permitted RCRA facilities that are out of compliance with RCRA requirements, and nei-

---

(continued)
tion numbers are required for each role and each site at which any regulated activity takes place.

Two important areas in which transporter regulation is considerably weaker than that applied to TSDFs is that transporters do not need permits or have to meet similar financial responsibility requirements -- unless, of course, they act as generators as described above.

ther the generator nor the transporter need be remotely aware of it. There are three general reasons why TSDF noncompliance arises.

The most obvious form of noncompliance is willful violation of the regulatory system, and this is often assumed to be the most serious problem. This may be correct in specific cases but need not be true in general. For example, willful noncompliance with costly requirements that offer trivial environmental benefits actually enhances efficiency. Similarly, some regulatory requirements may be counterproductive, even when viewed from the most narrow environmental perspective, and willful noncompliance with such rules enhances environmental quality (and perhaps efficiency as well).

The second way noncompliance may occur involves upset conditions. Sometimes, normal start-up and maintenance tasks cause these events, and the condition is automatically resolved once normal operations resume. Ironically, the mere testing of safety equipment and emergency procedures may create additional nontrivial upset hazards.[34] Besides technological imperatives, upset conditions can be caused by nature or human error.

Finally, noncompliance may be stochastic, the inevitable result of the sheer complexity of the regulations, the technical character of the performance criteria, or even the instruments used to measure performance. Facility permits are extensive, technical, and site-specific documents that are amended as circumstances warrant. Some compliance criteria (particularly those dealing with ground water monitoring) tend

[34] See Nichols and Wildavsky (1987) for a discussion of these risks in the context of nuclear power plants.

to be moving targets for which requirements for affirmative compliance may never exist. It is thus a difficult task for a responsible facility operator to ensure continued compliance, thus making it hard to discern willful from inadvertent violations.

Ironically, willful violations are probably the most difficult to detect, with upset conditions not far behind. Intentional acts can be scheduled when inspections are not expected or shielded from the inspector's view. Upset conditions are unpredictable but generally infrequent, which makes them unlikely to occur during an inspection. Thus, inspections are most likely to discover stochastic variations in facility operations because of the large number of independent criteria upon which a violation might be founded.

Generators that stay in the RCRA system may face considerable liability risks from TSDFs that are regularly or even occasionally out of compliance. Uncertainty surrounding future liability for current waste disposal thus raises the expected cost of hazardous waste disposal above the actual fees paid. However, the public-good aspects of TSDF monitoring discourage generators from doing it. Even if free-riders could be eliminated, generator efforts to improve TSDF compliance may not be cost-effective because regulatory compliance does not extinguish liability. In any case, uncertainty surrounding TSDF compliance further stimulates interest in source reduction.

Evidence of Noncompliance

The available data indicate that a large number of permitted facilities are regularly out of compliance. As of October 1986, 770 of the 1,655 (47 percent) land disposal facilities in the EPA's Strategic Planning and Management System had been found to be in significant non-compliance (SNC), and the number of new SNC facilities identified was rising faster than the Agency could reduce its backlog through enforcement actions.[35] An examination of enforcement actions in New York City during FY 1984 revealed that 90 percent of the facilities listed in the EPA's HWDMS data base had never been inspected. In addition, half of the violations observed were Class I violations, 90 percent of which had not been corrected six or more months after citation (Goldman, Hulme and Johnson 1986: 243, 262).

Problems with RCRA enforcement have been evident since the outset of the program. Enforcement resources were increased by HSWA to approximately $26 million in FY 1987, enough to fund more than 400 FTE employees (CRS 1987: 41). However, RCRA enforcement had been cut from about $13 million in FY 1981 to $5 million in FY 1983, making the recent increase appear larger than it really was. In FY 1984, the EPA fielded only 176.5 full-time-equivalent inspectors across its ten regional offices, and they each performed an average of just 5.55 inspections during the fiscal year (Goldman, Hulme and Johnson 1986: Table 10-1).

Enforcement efforts throughout the EPA's regulatory programs have been criticized as both ineffective and inefficiently targeted. Some believe that the EPA's long-standing emphasis on achieving *initial* com-

[35]**See** CRS (1987: 43-44); and Goldman, Hulme and Johnson (1986: Table 10-2). SNC applies to any facility that has Class I violations of ground water, closure, post-closure, or financial responsibility requirements, or which poses "a substantial likelihood of exposure to hazardous waste or has caused actual exposure, has realized an economic benefit as a result of non-compliance, or is a chronic or recalcitrant violator."

pliance (i.e., the installation of state-of-the-art technologies) has been misplaced, thereby giving short shrift to the task of ensuring *continuing* compliance (i.e., the dynamic achievement of regulatory objectives).[36] Given the long time periods of concern surrounding hazardous waste disposal, continuing compliance seems particularly important if public health and the environment are to be protected from hazardous wastes.

Conclusion

Our discussion makes quite evident that a nontrivial amount of hazardous waste escapes regulatory control under RCRA. Some of this waste is excluded by statute or administrative discretion. Much of it is supposed to be controlled through programs authorized by other environmental statutes, such as the Clean Water Act. The remainder escapes regulatory control, both because enforcement is inevitably imperfect and resources committed to enforcement inevitably seem inadequate for the task. Since a comprehensive risk assessment has not been conducted, we cannot determine whether these gaps in regulatory control portend significant environmental risks. We can say, however, that if rising hazardous waste disposal costs and potential liabilities are stimulating waste minimization efforts, then they also make it attractive to stay out (or get out) of RCRA. Because of the targeting difficulties involved, additional incentives ostensibly intended for waste minimization may exacerbate these phenomena if they fail to reduce the financial advantages obtained from noncompliance.

---

[36] __See__, e.g., Russell, Harrington and Vaughan (1986).

Moreover, opportunities for illegal disposal seem to be as plentiful as ever. The EPA's establishment of a criminal enforcement program is ample testimony that illegal disposal continues to occur. As long as it is relatively inexpensive, illegal disposal will remain an attractive way to achieve "waste minimization." This is but another reason why any additional incentive programs should be designed to narrow the gap between legal and illegal disposal costs.

Chapter 4:

THE IMPACTS OF NONCOMPLIANCE AND ENFORCEMENT ON RISK AND WASTE MINIMIZATION

In Chapter 2 we showed how the optimal level of waste minimization depends on the interaction of disposal costs and external damages. Rising RCRA disposal costs make waste minimization more attractive to the firm because it enables the firm to avoid these costs. But the very rules that make disposal more expensive also reduce the residual damages that result from disposal, making waste minimization less valuable to society at large.

In Chapter 3 we outlined a variety of ways in which firms may legally or illegally fail to comply with regulatory standards. The possibility that firms may choose not to comply makes the tradeoff between rising disposal costs and waste minimization all the more important. The strong incentives for waste minimization provided by RCRA and CERCLA will not have nearly as large an effect on such firms so strategies for encouraging waste minimization must take noncompliance into account.

We now weave together these two threads of the analysis, extending the framework from Chapter 2 to include the effects of noncompliance. The most obvious response to the problem of noncompliance is enforcement, the effects of which we examine in the second half of the chapter.

Alternative Models of Compliance

The environmental policy literature is replete with analyses of the merits of alternative regulatory instruments applied across a wide range of pollution problems.[1] However, with few exceptions this literature embraces the assumption that firms comply with the dictates of the regulatory agency, whether it be in terms of meeting design or performance standards, or accurately reporting emission volumes for the purpose of implementing an incentive-based system. Only recently have the implications of noncompliance been explored in much depth, most notably in a major study of environmental enforcement by Russell, Harrington and Vaughan (1986). In that study, the existing literature is classified according to what assumptions have been made on the following four issues:

1. Will the firm cheat if it is in its self-interest to do so?
2. Can the firm control its discharge levels exactly?
3. Does the regulator monitor firm behavior?
4. If the regulator does monitor, can it do so without error?

The first and third issues refer to explicit assumptions with respect to firm and regulator behavior. The second and fourth items have to do with whether the underlying processes of pollution control and regulatory enforcement are deterministic or stochastic.

Most models in environmental economics make very restrictive assumptions about these factors: firms have perfect control and will not cheat, and thus monitoring issues are irrelevant. This is precisely the approach we used in Chapter 2, where we showed that firms would reduce waste generation until the marginal cost of doing so equalled the unit

[1] See, e.g., Kneese and Schultze (1975); Baumol and Oates (1979); Schelling (1983); Nichols (1984); Bohm and Russell (1985); and Tietenberg (1985).

cost of waste disposal. Firms were presumed not to cheat, the regulator did not need to monitor firm behavior, and neither party's actions were complicated by stochastic processes.

In models of noncompliance, at least one of these assumptions is relaxed. For example, Viscusi and Zeckhauser (1979) assume firms do not comply if it is not in their self-interest. Firms have full control of the relevant output measure (in the Viscusi-Zeckhauser paper, product quality), and the regulator monitors their behavior without error if an inspection occurs. Thus, uncertainty surrounds only whether a particular firm will be inspected. Viscusi and Zeckhauser show that the optimal standard under incomplete enforcement is obtained where the marginal benefits of regulatory compliance equal the opportunity cost of benefits foregone due to noncompliance. Thus, the optimal standard is generally weaker (and never stronger) than if full compliance can be ensured.

Similar approaches have been used by Harford (1978) and Storey and McCabe (1980). Both of these papers focus on how the presence of noncompliance affects the choice between standards and charges. In addition, Harford's paper analyzes the merits of subsidizing pollution control costs as a means of overcoming the noncompliance problem. A recent paper by Sullivan (1987) attempts to determine the optimal levels of enforcement effort and disposal subsidies in a regime where firms either comply or engage in illegal disposal. Sullivan takes the existing regulatory standards as given, so his model cannot be used to evaluates the effects of alternative control levels on compliance rates.

In studies by Downing and Watson (1973; 1974; 1975), Watson and Downing (1976), and Vaughan and Russell (1983), one or both of the deterministic process assumptions are relaxed. The emphasis of these works has been directed toward either firms' optimal control strategies (Downing and Watson) or the regulator's optimal monitoring scheme (Vaughan and Russell). In a paper by Linder and McBride (1984), both processes are stochastic. The book-length study by Russell, Harrington and Vaughan (1986) also allows for stochasticity in both measurement and control and suggests a range of appropriate enforcement strategies.

With respect to hazardous waste management (and especially minimization), these process assumptions seem particularly important. First, consider the difficulties facing regulators. Monitoring is especially difficult for several reasons. The agency may have incomplete information concerning both the identity of firms subject to its regulatory authority, and the amount and type of wastes they currently generate. Knowing of the firm's existence does not necessarily translate into knowing where to look for evidence, because waste management activities can occur far from the production lines that generate the wastes. Furthermore, the measurement task itself is greatly complicated by the number of substances involved, the myriad methods by which they could be combined, the complex testing methods needed to verify and quantify the presence of many of them, and the variety of ways in which these materials can be managed. Aggravating these monitoring problems is the large number of firms that are plausibly subject to regulatory authority, particularly since Congress rescinded the EPA's SQC exemption in 1984.

Second, from the waste handler's perspective, compliance is at best an elusive target. The full gamut of RCRA regulations is stunningly complex, so much so that a cottage industry has materialized offering expert help just to interpret the rules. The EPA has responded to the confusion and complexity by publishing guidance documents intended to clarify the requirements, but in many cases these documents themselves are difficult to understand, providing still more grist for the RCRA consulting mill.[2] Regulatory ambiguity is exacerbated by uncertainty about the pace of technological change, particularly because of its "ratchet" effect on standards, and the direction and timing of future agency actions. The highly prescriptive character of HSWA heightens concerns that the Congress will make major changes in its next RCRA reauthorization, changes that could destroy the value of compliance decisions made now.

Assumptions concerning firm behavior are similarly important in hazardous waste regulation. In Chapter 3 we classified noncompliance as willful, aberrant (i.e., due to upset conditions), or stochastic (i.e., due to imperfect measurement of performance), and said that regulators are often hard-pressed to distinguish among them -- clearly a conundrum for both the efficacy and perceived legitimacy of regulators' enforcement programs. All three types of noncompliance seem to be copious with respect to hazardous wastes. Noncompliance may be manifest in many ways, from simply refusing to pay any attention to the regulations, to mistaken compliance with the wrong rules, to intentional compliance with the wrong rules, to outright illegality. In addition to measurement error, stochastic noncompliance may arise simply due to inconsistencies in

[2] See, e.g., comments by Garelick (1987) on the EPA's guidance document concerning recycling.

regulatory interpretation, the intensity and frequency of inspection, and the nature of the facility being inspected.

One phenomenon that previous noncompliance models seem to have neglected is the possibility that regulatees might respond by abandoning the regulated system for the black market. A good reason for this past neglect is that there are few analogs of the black market for hazardous waste disposal in environmental programs; noncompliance is frequently a problem in other areas, but it rarely leads firms to take actions that increase risks. In the hazardous waste area, however, incidents of illicit disposal contributed significantly to public (and Congressional) perceptions of the need for strict regulation, and still occur with sufficient frequency to arouse concern.[3]

A Framework for Noncompliance and Enforcement

Our analysis of the effects of noncompliance begins with a modified version of the model developed by Viscusi and Zeckhauser (1979). In that model, firm and agency decisions are deterministic; firms know what it costs to comply and know when they are in compliance, and the regulator can measure compliance perfectly. Limited enforcement resources, however, make it impossible to monitor all firms and there are caps on the penalties that may be imposed. As a result, the expected penalties for noncompliance may not be sufficient to get all firms to comply. After setting up the basic framework, we introduce the possibility of noncompliance taking the form of high-risk, "black

[3] <u>See</u>, e.g., Epstein, Brown and Pope (1982); Law Enforcement Hearings (1983); and Block and Scarpitti (1985).

market" disposal. We then look in more detail at the effects of different regulatory approaches -- including various forms of enforcement, taxes, and subsidies.

The Basic Model

In Figure 4-1, the curve D(r) represents the total direct cost to the firm of disposing of a unit of waste a$ a function of the unit risk remaining after disposal. It is simply the total-cost analog to the marginal cost curve illustrated in Figure 2-2, -D'(r). In the absence of regulation, the firm will minimize costs at the risk level r_N, which yields a cost of C_N - $D(r_N)$.

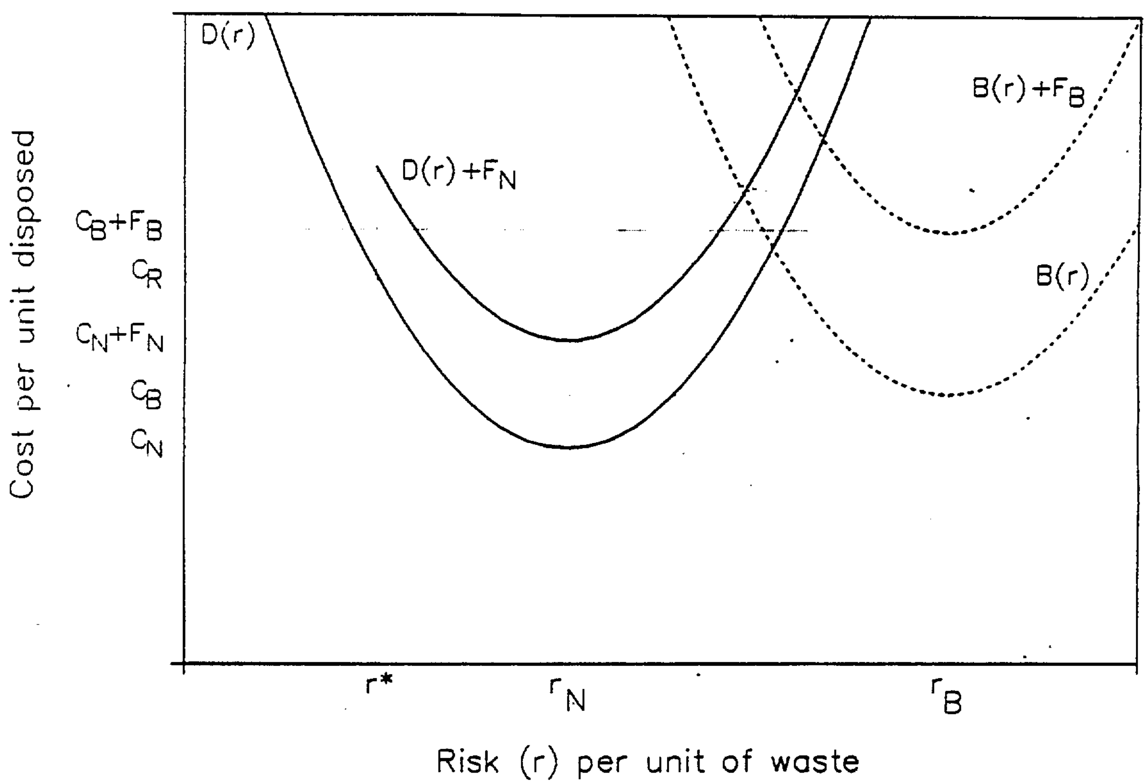


Figure 4-1. The Representative Firm’s Waste Disposal Decision

Suppose, as in Chapter 2, that the regulator establishes standards mandating r* as the maximum allowable, unit risk, where $r^* < r_N$. If the firm complies with these standards, it minimizes costs at $C_R = D(r^*)$. In Chapter 2, we assumed that the firm would comply, despite the fact that $C_R > C_N$.

Failing to obey the standard exposes the firm to noncompliance penalties, potential liability for remedial or corrective actions, and intangible costs such as damage to reputation. But these costs are not automatic: the probability that they will be imposed depends on, among other things, the size of the enforcement effort. For any fixed level of enforcement, the threat of penalties resulting from noncompliance raises the firm's expected total costs by an amount that we define as F_N.[4] Thus, for risk levels greater than r^*, the firm's total cost function shifts upward to $D(r) + F_N$. Under the presumed optimal standard, the full total cost curve is therefore discontinuous, with a vertical shift at r^*. This curve is minimized at either r^*, in which case the firm complies, or r_N, in which case the firm does not.[5] For the firm to comply with the standard,

(4-1) $$C_R < C_N + F_N \text{, or}$$

[4]F_N is a function of the level of sanctions and the combined probabilities of detection, apprehension, and punishment. In a more general analysis of enforcement strategies, these components would be disaggregated. Our purpose here is limited to analyzing the overall effect of enforcement rather than the effects of its constituent parts.

[5]In keeping with the Viscusi-Zeckhauser model, we have assumed that the expected value of the fine, F_N, is constant with respect to the difference between actual unit risk and the unit risk embodied in the regulatory standard. This need not be the case. For example, penalties might increase with the severity of the violation. In that case, the expected cost of noncompliance would be minimized at a risk level lower than r_N; firms that did not comply would still engage in some risk reduction simply to reduce the expected cost of sanctions. Thus, lost risk reduction opportunities from noncompliance would be smaller under this kind of penalty structure. Unless such penalties are sufficiently large, however, these firms still will not comply with the standard.

$$(4\text{-}2) \qquad C_R - C_N < FN .$$

The left side of Equation 4-2 equals the vertical distance between C_R and CN in Figure 4-1; the right side is the vertical shift in the cost curve if the firm is out of compliance. As we have drawn the figure, this condition is not satisfied; i.e., the firm's minimum-cost strategy is noncompliance.

Viscusi and Zeckhauser emphasize the fact that tightening the standard can reduce overall achievement of the regulator's goal, in this case risk reduction. As the standard becomes tighter, those firms that remain in compliance will achieve greater safety, but the higher costs of tighter standards also will drive more firms into noncompliance, thus undoing some of the benefits obtained by the standard. In their paper, they show how the optimal standard depends on the distribution of firms' costs; that result applies to hazardous wastes as well, but we do not explore it here.[6]

The Black Market

In keeping with the Viscusi-Zeckhauser model (and most other analyses of compliance), thus far we have assumed that if firms fail to comply with a standard, they will simply continue with whatever disposal method they used prior to regulation. The worst that can happen with a standard and its associated enforcement effort is that will be ineffec-

[6]The issue arises again in the case study presented in Chapter 7, in which we derive the optimal subsidy for a deposit-refund system applied to used lubricating oil.

tive; failure to comply means business as usual. In the case of hazardous waste, however, firms may have other, even riskier, disposal options available through a "black market" that arises to evade detection. If conventional regulatory enforcement drives some firms into the black market, it may have the perverse effect of increasing risk.

Suppose that in addition to the regulated market, the firm has illegal disposal options reflected in the curve B(r) in Figure 4-1, where the $B(\cdot)$ notation indicates that the black market involves fundamental differences in technology and market characteristics.[7] Illegal disposal costs are minimized at r_B, which corresponds to a disposal cost of $C_B = B(r_B)$. In the figure, we have shown $C_B > C_N$; i.e., in the unregulated situation, "ordinary" noncompliance is both cheaper and less risky than black-market disposal. Conventional enforcement, however, can make the black market viable by driving the cost of ordinary noncompliance above that of illegal disposal:[8]

[7] An awkward semantic issue arises here. Strictly speaking, ordinary noncompliance and black market disposal are both illegal in the sense that they violate either laws or regulations and can be punishable through various sanctions. In this framework, waste management practices that may have been considered acceptable in the past, but are now regarded as inadequate or improper, constitute "ordinary noncompliance" because they arise within the context of the firm's pre-existing waste management technology and a regulated marketplace. We shall reserve the terms "illegal" and "illicit" for those disposal activities so egregious that no pretense of propriety could conceivably accompany them. They occur in what we shall term the "black market," an environment regulated not by the institutions that accompany and legitimate market exchange, but by the encumbering presence of law enforcement, and to some extent, the needs and desires of the participants to regulate themselves.

[8] If C_B initially is less than C_N, then the firm would have been in the black market from the outset, and regulatory enforcement would have no effect. If $C_B > C_R$, then the black market is not viable and regulatory enforcement cannot induce firms to switch into it.

$$C_N + F_N > C_B \quad (4\text{-}3)$$

This is the case shown in Figure 4-1; conventional enforcement pushes the firm into the black market, rather than into compliance.

Enforcement resources also may be devoted to deterring black-market disposal. We define F_B as the expected value of penalties for black-market disposal. Although there may be some overlap between the two types of enforcement, in general the approaches are likely to employ different methods.

Law enforcement will shift the cost of black-market disposal upward to $B(r) + F_B$. If both regulatory and law enforcement programs are in place, a firm initially in the regulated market (but not in compliance) will switch to the black market only if its cost is less than the minimum of ordinary noncompliance and compliance:

$$C_B + F_B < \min\left\{\begin{matrix} C_R \\ C_N + F_N \end{matrix}\right\}. \quad (4\text{-}4)$$

As we have drawn Figure 4-1, this condition is not satisfied; the penalty for black-market disposal is large enough to make it more costly to the firm than compliance with the standard, and ordinary noncompliance is the least-costly option of all.

Note that with the possibility of high-risk, black market disposal, Viscusi and Zeckhauser's cautions about the potential adverse effects of tighter standards are reinforced; not only will such standards create incentives for noncompliance, but endeavoring to enforce them may make matters worse by driving at least some firms into the black market.

The Effects of Waste-End Taxes

Incomplete compliance also raises questions about the advisability of waste-end taxes. In Chapter 2 we showed that if compliance is assured, the regulator can motivate the firm to select the socially optimal level of waste generation and method of disposal by imposing a tax equal to residual external damage. If noncompliance is feasible, however, some firms that initially did comply would respond to the tax by dropping out of the system, thereby creating new welfare losses. If a viable black market exists, these welfare losses may be very large.

We make the reasonable assumption that firms in the black market will evade the tax; if regulators are unable to identify cases of blatant evasion, they certainly will be unable to levy a tax that requires monitoring of the waste output of such firms. Thus, in terms of Figure 4-1, a waste-end tax has no effect on the right-hand cost curve, the one associated with the black market.

Firms engaged in ordinary noncompliance may or may not be able to evade the tax. If they cannot evade it, then the firm will choose to comply if:

$$(5) \qquad C_R + T_D < \min\begin{Bmatrix} C_O + F_N + T_D \\ C_B + F_B \end{Bmatrix},$$

where T_D is the tax per unit of waste. In this case, the waste-end tax would not affect the tradeoff between compliance and ordinary noncompliance; in terms of Figure 4-1, the total cost curves for the regular market would shift upwards by equal amounts at all risk levels.[9] The tax, however, does make the black market more attractive, and thus can cause firms to shift from either compliance or ordinary noncompliance to the black market. The waste-end tax does succeed in causing regulated market firms to cut back on their waste generation, but the gains from these reductions may well be offset from substantially higher disposal risks for those firms that switch to the black market.

An alternative assumption is that firms practicing ordinary noncompliance also can evade the waste-end tax; after all, the tax cannot be levied at the generator level without revealing the firm's noncompliance, and levying the tax at the TSDF level will primarily capture approved disposal methods. The effects of the tax become more ambiguous, because now the firm will comply only if:

$$C_R + T_D < \min\begin{Bmatrix} C_O + F_N \\ C_B + F_B \end{Bmatrix} . \tag{6}$$

In this case the waste-end tax clearly lowers the costs of both forms of noncompliance relative to compliance, and thus provides an incentive for firms to stop complying irrespective of which form of noncompliance is least expensive. It does not provide an incentive for firms to switch from ordinary noncompliance to the black market; conversely, however, it

[9] This assumes a simple form of waste-end tax that does not vary with the risk level. A more sophisticated approach would be to vary the tax with the risk; that would require additional monitoring, however, and would not cope any more effectively with the black market problem.

provides no incentive to minimize waste generation for firms that engage in ordinary noncompliance.

In sum, a waste-end tax that would achieve the socially optimal result in the absence of noncompliance becomes potentially counter-productive once compliance is not assured. Moreover, the larger the tax the more likely it will be that a firm will find its incentive to comply has been nullified. Evaluating the aggregate effects of a waste-end tax requires that these results be summed across the distribution of firms' switch points, about which we have no information. Nevertheless, it seems quite plausible that some firms will have noncompliance costs that reside within the sensitive zone. If tax rates are high relative to disposal costs, then this zone may be quite large, and thus contain a significant number of firms.[10]

Impacts of Enforcement on Disposal Risk and Waste Minimization

In this section, we use the graphical approach developed in Chapter 2 to analyze in more detail the effects of enforcement on the firm's choice of disposal options and its level of waste minimization. We look first at conventional regulatory enforcement, targeted primarily at firms that are part of the regulated system, but which may not be in compliance with RCRA rules. We then turn our attention to "law enforcement, " which is aimed at firms in the black market.

---

[10]Waste-end taxes in California, for example, ranged from $2 to $150/ton in 1986 Hammitt and Reuter 1988; 8).

The Effects of Regulatory Enforcement

The four panels of Figure 4-2 illustrate the effects of regulatory enforcement on waste minimization for a firm engaged in ordinary noncompliance.[11] C_N is the firm's initial disposal cost and w_N is its initial level of waste generation per unit of output.

Suppose that black-market disposal is more costly than compliance, and thus illegal disposal is not a viable alternative. Regulatory enforcement raises the expected cost of noncompliance. If the enforcement effort is small, however, expected penalties may be insufficient to raise noncompliance cost above C_R -- the critical point for the firm to switch. As a result, the risk per unit of waste disposed will be as high as ever. Enforcement, however, will induce some additional waste minimization, with net social benefits less than or equal to the shaded area in Panel (a).

If the enforcement effort is increased so that the expected cost of noncompliance exceeds C_R, then the firm will indeed switch to compliance. This reduces the social cost of the hazardous wastes generated per unit of output from S_N to S^*, and yields net social benefits equal to the lightly shaded rectangle in Panel (b). Switching to compliance pushes unit disposal costs to C_R, so the firm will also reduce waste generation to w_R per unit of output. Thus, the net social benefits from waste minimization equal the darkly shaded area in Panel (b).[12]

[11] Recall that, by assumption, regulatory enforcement has no effect on firms in the black market.

[12] Implicitly, we have treated the intersecting area as a benefit associated with a reduction in disposal risk. It could, however, just as easily be attributed to waste minimization.

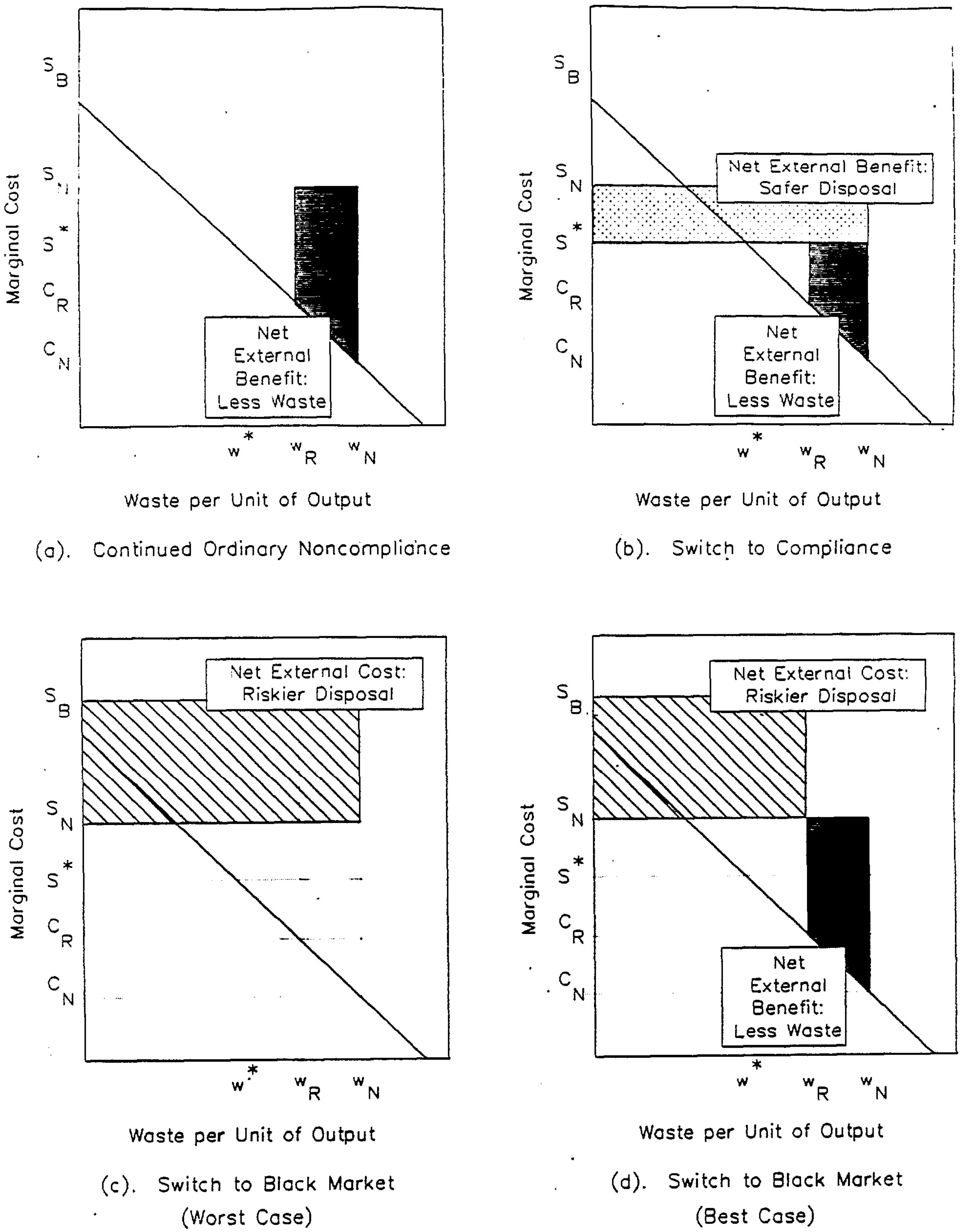


Figure 4-2. Impact of Enforcement on Ordinary Noncompliers

If the black market is viable, then the salutary results shown above may be seriously attenuated, if not reversed. If the expected cost of black-market disposal lies below that of legal disposal, then as the expected penalty for ordinary noncompliance increases, the firm will switch to the black market rather than to compliance. In the "worst" case, black-market disposal initially is almost as cheap as ordinary noncompliance; the slightest increase in regulatory enforcement induces a switch to the black market, and no waste-minimization benefits arise. Furthermore, the switch to illegal disposal creates new social costs from increased disposal risks equal to the diagonally shaded rectangle in Panel (c).

In the "best" case, the cost to the firm of black-market disposal is almost as high as that of legal disposal. Only a vigorous regulatory enforcement program that seeks to pressure the firm to switch to compliance will instead cause the firm to switch to black-market disposal. It will, however, induce some waste minimization. As a result, it yields waste-minimization benefits equal to the darkly shaded area in Panel (d). New social costs from increased disposal risks still arise, an amount equal to the diagonally-striped rectangle in Panel (d).

Note that these effects are reinforcing. Potential waste-minimization benefits get larger and potential disposal-risk costs get smaller as the switch point approaches C_R. Thus, if the sum of waste-minimization benefits and disposal-risk losses is negative when the switch occurs near C_R, then the sum will also be negative for all lower switch points as well.

The Effects of Law Enforcement

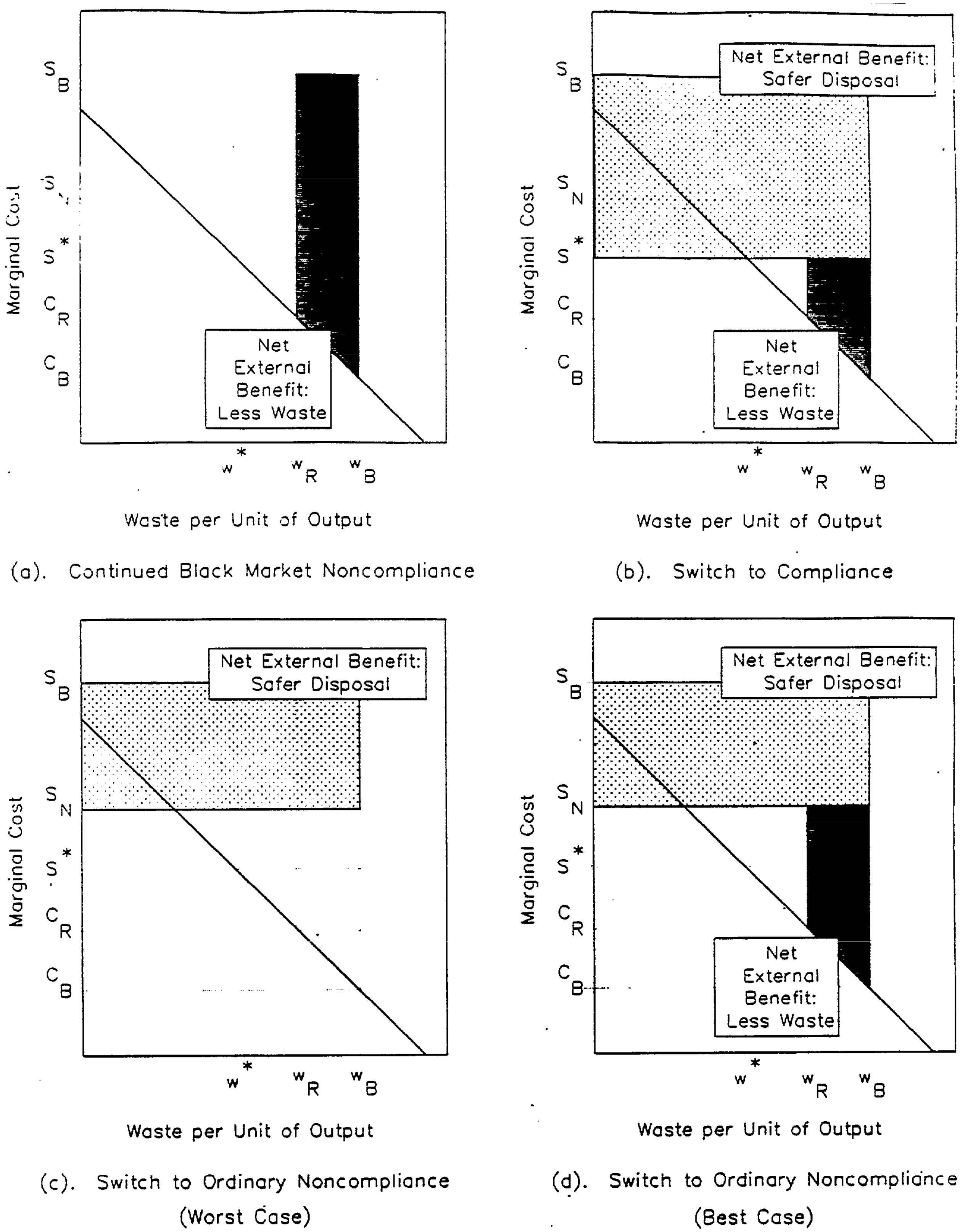
Now consider law enforcement targeted on black market disposal. The effects are illustrated in Figure 4-3, which duplicates the basic panels from Figure 4-2, except that the firm's initial cost, C_B, is the cost of black-market disposal.

As with regulatory enforcement, even an effort that is too small to induce any change in disposal method can yield waste-minimization benefits; the maximum possible benefit from waste minimization alone is the darkly-shaded area of Panel (a). Note that waste minimization yields a larger net social benefit with black-market noncompliers than with ordinary noncompliers because of the higher social cost of black market disposal.

If ordinary noncompliance is more expensive than compliance, then the firm will come into compliance if the expected penalty for black market disposal raises the cost of continued illegality above C_R. The social cost per unit of output is then reduced from S_B to S^*, yielding net benefits equal to the lightly shaded rectangle in Panel (b). In addition, because compliance entails higher disposal costs, the firm cuts waste generation to w_R and creates additional social benefits equal to the area of the darkly shaded trapezoid.[13]

---

[13]The net benefits defined by the intersection of these areas can be attributed to either waste minimization or to the reduced risk of disposal.



(a). Continued Black Market Noncompliance

(b). Switch to Compliance

(c). Switch to Ordinary Noncompliance (Worst Case)

(d). Switch to Ordinary Noncompliance (Best Case)

Figure 4-3. Impact of Enforcement on Black Market Noncompliers

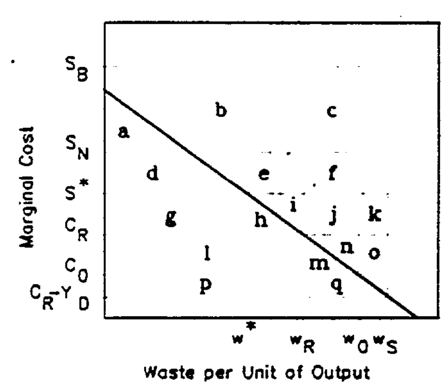
However, if the expected cost of ordinary noncompliance is slightly less than than that of compliance, then tightening law enforcement will drive the firm out of the black market and into ordinary noncompliance rather than compliance. This may seem undesirable, but the social benefits that arise may be substantial. By switching to ordinary noncompliance, the firm lowers the social *cost* of each unit produced from S_B to S_N, yielding net benefits equal to the area of the lightly shaded rectangle in Panel (c). If the cost of ordinary noncompliance is just barely higher than that of black-market disposal, these will be the only benefits reaped. The switch, however, also may yield additional waste-minimization benefits as large as the area of the darkly-shaded trapezoid in Panel (d), if the cost to the firm of ordinary noncompliance is almost as great as that of compliance.

Comparison of Enforcement Strategies

We have summarized the potential effects of both types of enforcement efforts in Table 4-1; the graph below the table is structured to provide a guide to the areas identified in Figures 4-2 and 4-3. In both cases, enforcement is intended to make noncompliance more expensive than compliance. If enforcement works as intended, benefits result from less risk per unit of waste disposed and from a reduction in the amount of waste generated. Again in both cases, if the enforcement program is not large enough to change the firm's disposal method, no reductions are achieved in risk per unit of waste, but some gains are likely to result from waste minimization.

Table 4-1. Net Benefits of Alternative Regulatory Enforcement Programs

Change in Disposal Method	Waste Minimization Net Benefits	Disposal Risk Net Benefits
REGULATORY ENFORCEMENT (impact on firms initially in ordinary noncompliance)		
Continued Noncompliance	≤f+j+n	0
Switch to Compliance	+j+n	+d+e+f
Switch to Black Market		
Best outcome	+f+j+n	-a-b
Worst outcome	0	-a-b-c
LAW ENFORCEMENT (impact on firms initially in black market)		
Continued Black Market Disposal	≤c+f+j+n	0
Switch to Compliance	+j+n	+a+b+c+d+e+f
Switch to Ordinary Noncompliance		
Best outcome	+f+j+n	+a+b+c
Worst outcome	0	+a+b+c



Guide to Table 4-1

Note: In cases where the attribution of net benefits is ambiguous, they have been classified as resulting from reductions in disposal risk.

The key difference between the two types of enforcement lies in what happens if firms switch to another form of noncompliance rather than to compliance. For regulatory enforcement, switching to another form of noncompliance implies entry into the black market, with accompanying losses in social welfare due to increased disposal risks. By contrast, for law enforcement, changing the method of noncompliance implies exit from the black market, which means that potentially significant reductions in residual damage are obtained even if compliance does not occur.

Note also that some additional waste minimization is likely to take place irrespective of any shift in disposal method. At first glance this may seem to be a tidy result, but the impression is deceptive. Should the regulator observe waste minimization occurring subsequent to an increase in regulatory enforcement, for example, it does not necessarily mean that firms are responding to the increased threat of sanctions by complying with regulatory standards. Rather, since the waste minimization effect is independent of the firm's choice of disposal method, many firms may be responding by withdrawing from the regulated waste management system and selecting illegal disposal options instead.

Summary

The possibility of noncompliance complicates the tasks facing both regulators and policy analysts. Strategies such as waste-end taxes that appear highly desirable if compliance is assured may be counterproductive if firms can stop complying. Similarly, tightening standards may reduce overall safety by reducing the number of firms in compliance.

The obvious prescription for noncompliance is stepped-up enforcement efforts. Conventional enforcement, however, may increase risk by driving noncomplying firms to more dangerous, but less easily detected, forms of disposal in the "black market." Enforcement directed specifically at black-market disposal does not run that risk.

This analysis might appear to suggest that law enforcement targeted on black-market disposal is always superior to regulatory enforcement aimed at ordinary noncompliers. In general, however, the optimal enforcement strategy is likely to involve a mix of both approaches, and the relative emphasis will depend on a variety of factors. These factors include:

1. **Relative risks.** The relative levels of S_B, S_N, and S^* matter greatly. The larger the gap between S_B and S_N and the smaller the difference between S_N and S^*, the more important it is to curtail black market disposal and the less important it is that firms fully comply with disposal regulations. Under such conditions, law enforcement becomes relatively attractive.

2. **Relative enforcement costs.** *Ceteris paribus,* the cheaper an enforcement method is, the more attractive it will be relative to its alternative.

3. **The actual and potential distributions of firms across various categories.** The more firms there are in the black market as opposed to ordinary noncompliance, the more attractive law enforcement will be. Even if the black market is initially thinly populated, law enforcement will be preferred if intensified regulatory enforcement would drive many more firms into illegality.

It is also important to remember that enforcement of any stripe can be an expensive commodity. Not only does it consume scarce governmental resources that could profitably be spent elsewhere, but reliance upon enforcement also imposes additional costs on society. Some of these costs are obvious: resources devoted to enforcement are real social costs and not just transfers. Other costs are more subtle, such as the investment firms make to devise new ways of evading the enforcers. These adaptive responses diminish the effectiveness of enforcement, make any given level of efficacy more expensive, and reduce the value to society of the very actions enforcement seeks to motivate.

In theory, the problems of noncompliance could be eliminated by raising the expected penalties for both ordinary noncompliers and for those operating in the black market. Raising the probability of apprehension, however, is costly, requiring more enforcement resources. Alternatively, the penalties for noncompliance could be increased; as numerous papers have shown, even if the probability of apprehension is very low, deterrence can be achieved if the penalties are high enough (Becker 1968). In practice, however, the possibility of bankruptcy places a limit on effective penalties, and in most cases the political system places even lower effective limits.

Chapter 5:

SUBSIDIES TO ENCOURAGE COMPLIANCE AND WASTE MINIMIZATION

Enforcement programs seek to promote compliance by raising the cost of noncompliance. An alternative is to reduce the cost of safe disposal or, perhaps waste minimization, through subsidies. In this chapter, we explore the merits of such subsidies. The first section evaluates subsidies narrowly in terms of their impacts on the choice of disposal method and on the amount of waste generated per unit of output. Viewed from that perspective, subsidies, particularly those targeted at waste minimization, look quite favorable. In the next section, however, we point out several serious drawbacks, some of which apply to subsidies generically, but others of which are more particular to waste-minimization subsidies.

The Basic Analytics of Subsidies

A wide variety of subsidies might be considered for reducing the risks associated with hazardous wastes. Countless variations are possible given the many different activities that could be subsidized and the different measures that might be used to determine the amounts paid to firms. In this section, we abstract from these possibilities to focus on two simple forms of subsidies: those targeted on safe disposal and those aimed at waste-minimization.

Safe Disposal Subsidies

Consider first a subsidy for safe disposal. Let the subsidy be a fixed amount per unit of waste disposed, with the rate set at Y_D for each unit disposed at a risk of r^* or less, where, as before, r^* is the risk level achieved under the disposal standard. Referring back to Figure 4-1, instead of shifting the cost of disposal, D(r), up for riskier methods (as regulatory enforcement does), such a subsidy would shift it down for safer methods (to the left of r^*). If the firm complies, costs are minimized at r^*, so the firm will comply if:

$$C_R - Y_D < C_O = \min\left\{\begin{matrix} C_N \\ C_B \end{matrix}\right\}. \tag{5-1}$$

Obviously, the disposal subsidy increases the probability that the firm will choose safe disposal in compliance with RCRA rules over either ordinary noncompliance or the black market.[1]

The effects of a safe-disposal subsidy on waste minimization and social costs are illustrated in Figure 5-1. Consider first a firm that is in compliance with the disposal regulations; as shown in Panel (a). The effects in this case are unambiguously negative: the firm already complied, so there are no risk-reduction benefits, and the subsidy lowers the cost of disposal, thus increasing the firm's waste per unit of output from w_R to w_Y. The net social loss from the subsidy is the area of the shaded trapezoid in Panel (a).

[1]In Equation 5-1 we ignore the expected costs of enforcement. This simplifies the exposition, reduces clutter in the diagrams, and has no effect on the results.

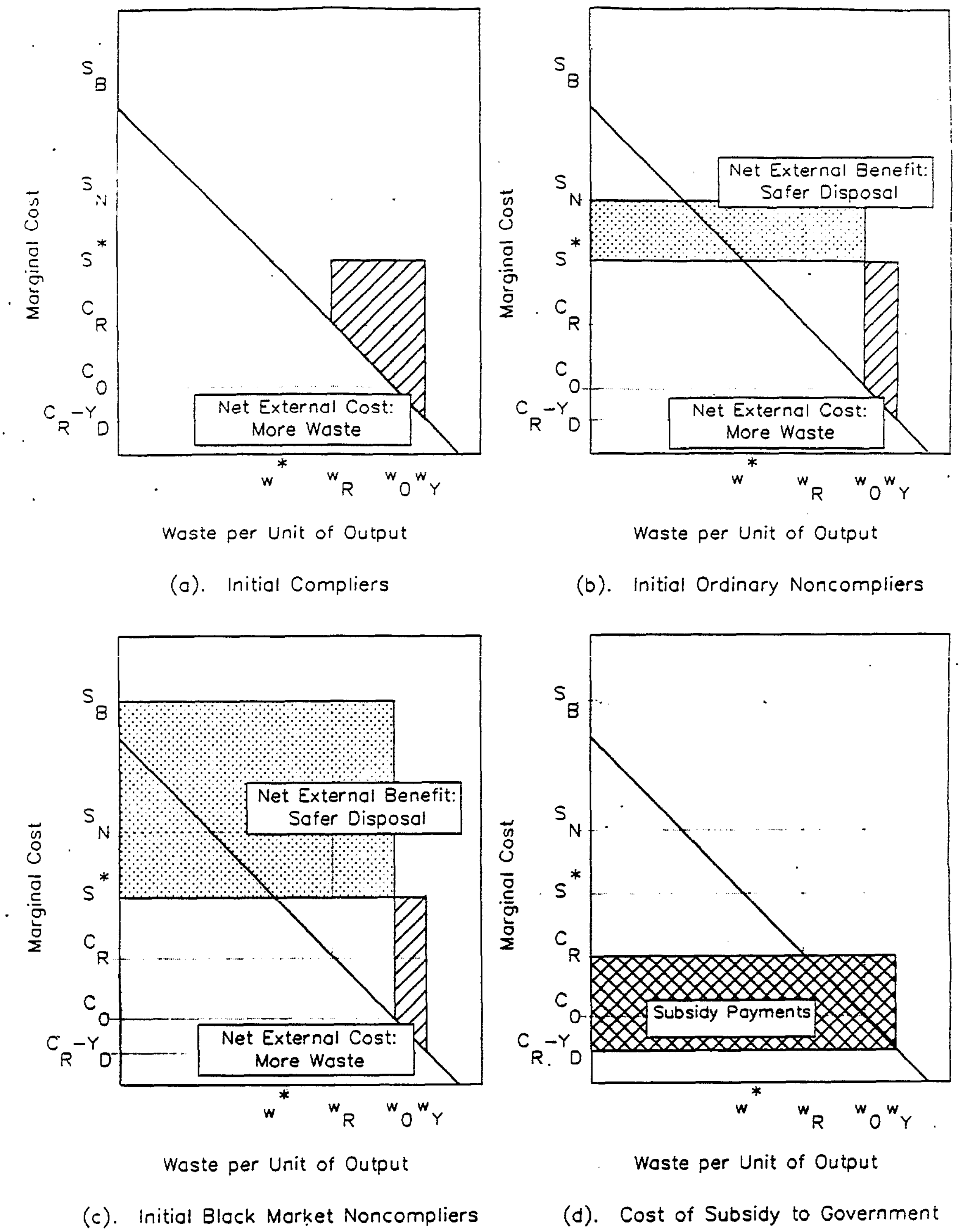


Figure 5-1. The Effects of a Safe-Disposal Subsidy

Now consider the effects of the subsidy on firms initially out of compliance. If the subsidy is too small, so that $C_R - Y_D > C_O$, nothing happens; the firm continues as before, and no costs or benefits are incurred. If, however, the subsidy is sufficient to induce compliance, the social cost of disposal falls. Panel (b) shows the effects for a firm initially in ordinary noncompliance. The lightly-shaded rectangle shows the net benefit of safer disposal. By lowering the cost of disposal, however, the subsidy also induces additional waste generation, causing a net loss equal to the diagonally-striped area. As drawn, the disposal-risk benefits outweigh the losses from increased waste generation, but there is no guarantee that the net result will be positive.[2]

Panel (c) of Figure 5-1 shows the effects for a firm that switches from black-market disposal to compliance as a result of the subsidy. The effects are the same as in the previous panel, except that the net benefits related to safe disposal are larger because the reduction in unit social costs is greater. As a result, it is far more likely that the net effect of the subsidy will be positive.

These changes in social costs are net figures, and give all costs and benefits equal weight. As discussed in more detail later in this chapter, however, the government's budgetary cost is likely to be of special interest, because funding that cost will require raising taxes.

---

[2] If the subsidy were optimized, so that the net cost of safe disposal were just equal to C_O, there would not be any additional waste generated. Tailoring the subsidy so precisely, however, would be a near-impossibility, especially for a subsidy that would apply to many different firms, each with its own cost structure. Thus, for most firms, any given subsidy will be either too large or too small, and only by chance will it be exactly right.

(which generally will have inefficiencies of their own), reducing expenditures on other programs, or increasing the deficit. The cross-hatched area in Panel (d) shows the government's cost for firms that participate in the subsidy program. Note that the size of the subsidy payment does not depend on the firm's initial behavior, only on the fact that it complies in the end, and on the amount of waste that it safely disposes.

Waste-Minimization Subsidies

The disposal subsidy increases waste generation because it increases the cost of waste minimization relative to disposal. In the process of reducing one wedge between relative private and social costs, it increases another. One solution to this problem would be to subsidize waste minimization, so that its private price relative to safe disposal were closer to its relative social cost.

In theory, a firm's eligibility for a waste-minimization subsidy might be independent of its disposal methods. In practice, however, waste minimization subsidies are almost certain to require on-site measurement and substantial interaction between the regulator and participating generators. It seems highly implausible that firms out of compliance with RCRA rules would be deemed eligible under such circumstances. Thus, we assume that any program to subsidize waste minimization would be restricted to firms in compliance with disposal rules; for firms out of compliance, switching to compliance would be a prerequisite for participation.

The appropriate marginal incentive for a firm in compliance with disposal regulations is provided by setting the waste-minimization subsidy, Y_M, equal to the external benefit from preventing the generation of a unit of waste; i.e., $Y_M = S^* - C_R$, which we define as E^*.[3] For a firm initially in compliance with disposal rules, it makes sense to participate in the waste-minimization subsidy program as long as the base amount of waste is not too much lower than the level of waste generated by the firm prior to the subsidy (w_R). Panel (a) of Figure 5-2 illustrates the results. The subsidy raises the marginal opportunity cost of generating waste by Y_M, to S^*. Thus, the firm reduces waste per unit of output to w^*, resulting in a net social benefit equal to the shaded triangle in Panel (a); this is the same quantity identified in Chapter 2 as the net benefit of forcing firms to internalize the external damage remaining after compliance with the optimal standard. If the subsidy is paid for all reductions below w_R, the shaded triangle also shows the net gain to the firm (the subsidy payment minus the net cost of reducing wastes).

The net social benefit from the waste-minimization subsidy is likely to be substantially larger for a firm initially out of compliance than for a firm that initially complies -- provided, of course, that the firm participates in the subsidy program. However, initial noncompliers are less likely to participate because the value of the waste minimization subsidy must be balanced against the cost of first coming into compliance with disposal regulations.

[3]Note that E^* is equal to λr^*, the expression for residual external damages used in Chapter 2. We use E^* because the risk per unit of waste safely disposed (r^*) and the value of life-saving (λ) are not analyzed separately in this discussion.

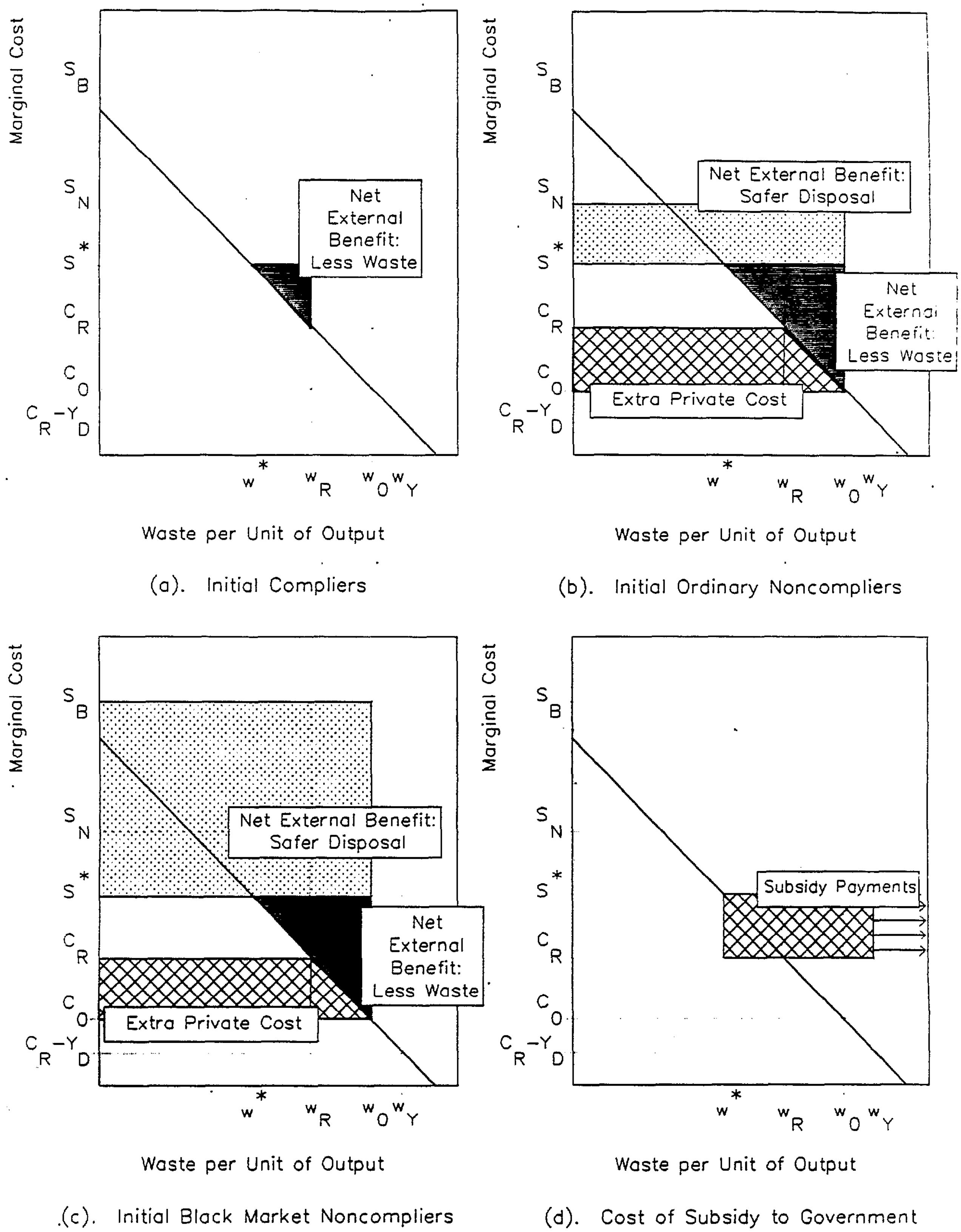


Figure 5-2. The Effects of a Waste-Minimization Subsidy

Panel (b) illustrates the situation for a firm initially in ordinary noncompliance. If it comes into compliance, the net social benefit from the reduction in the riskiness of the disposal method is shown by the lightly-shaded rectangle. Once the firm comes into compliance with the disposal regulations, its unit disposal costs rise to C_R and, like the firm initially in compliance, its marginal opportunity cost of waste generation rises to **S** Thus, it reduces waste generation to w^* per unit of output, yielding additional net social benefits shown by the darkly- shaded triangle in Panel (b). Panel (c) shows the same effects for a firm initially in the black market; the only difference is that the reduction in social costs associated with disposal is greater.

Now let us examine why noncompliers are less likely to participate. Let C_O be the initial cost of disposal. To become eligible for the waste minimization subsidy, the firm must first come into compliance, with the disposal regulations, at a net cost per unit produced equal to the sum of the areas of the cross-hatched rectangle and triangle in either Panel (b) or (c). The rectangle represents the extra disposal costs for wastes still generated, while the triangle represents the extra cost of waste minimization for reducing unit wastes from w_O to w_R. Once the firm incurs those costs, it becomes eligible for the waste-minimization subsidy. If the base is set at w_R, however, the value of that subsidy will be no greater than it was to the initial complier shown in Panel (a). This amount is unlikely to be large enough to offset the cost of coming into compliance. If it is not sufficient, the firm will not participate and no net benefits will be reaped.

Participation in waste-minimization subsidy programs also may increase the risk that previous misconduct will be discovered, exposing the firm to legal penalties and, probably more important, to a greater chance of liability for cleanup costs and other damages. Such discovery is always possible, of course, but participation may sharply increase its probability by requiring extensive governmental evaluation and oversight, both to determine eligibility and to calculate the level of subsidy payments to be awarded. This means, of course, that even a very large waste-minimization subsidy program targeted to reach noncompliers may fail unless protection against punishment can be assured. Without amnesty, such a program might have little effect on noncompliance, instead providing still larger transfers to firms that are already in compliance and thus do not need absolution for past misconduct.

In contrast, safe-disposal subsidies can be structured to pose much less threat to noncompliers. To administer such programs, the government would not have to become closely involved with generators. Indeed, if safe disposal subsidies were administered through permitted TSDFs rather than at the generator level, the regulators would not even have to know the identities of generators. As a practical matter, it is difficult to imagine any regulatory agency forswearing the collection of additional information, particularly if by doing so it could improve the efficacy of its enforcement program. Nevertheless, the more documentation that is required to receive the subsidy, the more threatening the program will be to noncompliers and the less likely it is that they will participate, so the smaller will be any potential improvements in compliance. Thus, granting amnesty does not really harm compliance, because failing to provide amnesty does not make enforcement any easier.

To cope with these additional costs to the firm, the payoff for participation in waste-minimization subsidies could be increased in either of two ways. One would be to increase the waste-minimization subsidy rate. That, however, would distort incentives at the margin, encouraging too much waste minimization. Moreover, it probably would require a great deal of extra government expenditure. Some of the higher payments would go to pay for this excessive waste minimization, leaving a smaller net return to the firm to balance against the cost of coming into compliance with disposal regulations. As a result, the waste minimization subsidy would have to be quite large to attract noncomplying firms.

The other approach is to increase the base amount of waste from which the subsidy is paid. In Figure 5-2, for example, if the base were increased to w_Y, the total subsidy payment to the firm would increase to the cross-hatched rectangle in Panel (d), and the net return to the firm (exclusive of the cost of complying with the disposal regulations) would be the portion of that rectangle located to the right of the demand curve. It is quite possible, of course, that even this amount would be insufficient for some firms, in which case the base could be increased by even more. The political difficulties of doing this might be substantial, as it would appear that firms were being paid for waste reductions that they would have made anyway.

Combined Subsidies

Both types of subsidy programs have obvious problems. The safe disposal subsidy causes more waste to be generated, though it may offer substantial net social benefits from safer disposal. Of course, waste-minimization subsidies deal with the problem of waste minimization, but they are unlikely to attract noncompliers unless payments are provided to firms well above direct waste-minimization costs, and thus are open to charges of providing windfall profits to polluters.

Combining the two types of subsidies can alleviate some of these problems. As long as the waste-minimization subsidy is at least as large as the disposal subsidy, it will counteract the latter's distorting effects on the amount of waste generated. Ideally, the waste-minimization subsidy also should reflect the external, residual damages associated with safe disposal. Thus, the subsidy should be:

$$Y_M = Y_D + (S^* - C_R). \qquad (5\text{-}2)$$

With this subsidy, the effective marginal cost to the firm of generating a unit of waste is S^*; each unit of waste costs $C_R - Y_D$ for disposal, plus the firm forgoes a waste-reduction subsidy of YM = $Y_D + (S^* - C_R)$, so the net cost is $(C_R - Y_D) + (Y_D + S^* - C_R) = S^*$. As a result, the firm engages in waste minimization efforts to the point where its marginal cost is equal to S^*; i.e., it reduces its waste generation per unit of output to w^*.

For firms that participate, the net effect of this combined subsidy is the same as shown in Panel (a) of Figure 5-2 for a pure waste-minimization subsidy. The key difference, however, is that firms are far more likely to participate because the subsidy payments can be made quite high without setting a very high base from which to measure waste reductions. Conversely, however, the cost to the government may be substantially higher, which, as discussed below, may entail sizable inefficiencies.

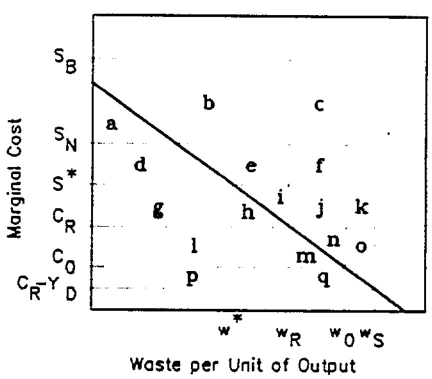
Comparison of Subsidies

Table 5-1 compares the net benefits of disposal and waste minimization subsidies in terms of the areas shown in Figures 5-1 and 5-2; a guide to those areas is provided in the diagram below the table itself. The first section shows the effects of a safe-disposal subsidy, while the second one shows the effects of a waste-minimization subsidy, offered either alone or in conjunction with a safe-disposal subsidy. In both cases, the net benefits shown are for firms that come into (or remain in) compliance with disposal regulations after receiving the subsidy; no costs or benefits apply to firms that remain out of compliance, as they do not take part in either subsidy plan.

Table 5-1 suggests that the waste-minimization subsidy is always preferable to the disposal subsidy; for each class of firms, it yields the same benefits in terms of disposal-risk reductions, and it yields higher waste-minimization benefits. This conclusion may be misleading, however, because Table 5-1 does not address the question of the relative numbers of firms that fall into various categories. Unless a waste-minimization subsidy calculates payments from a very large base level of waste, as shown earlier, it may not induce many firms to switch from noncompliance (ordinary or black market) to compliance. Thus, although a waste-minimization subsidy may yield higher net benefits for firms that switch, say, from ordinary noncompliance to compliance, it may induce fewer such changes in behavior than a disposal subsidy. Combining the two types of subsidies seems likely to work better than either instrument alone.

Table 5-1. Comparison of Net Benefits of Alternative Subsidies

Initial Disposal Method	Waste Minimization Net Benefits	Disposal Risk Net Benefits
	SAFE-DISPOSAL SUBSIDY	
Compliance	-j-k-n-o	0
Ordinary Noncompliance	-k-o	+a+b+c+d+e+f
Black Market	-k-o	+a+b+c+d+e+f
	WASTE-MINIMIZATION SUBSIDY	
Compliance	+i	0
Ordinary Noncompliance	+i+j+n	+d+e+f
Black Market	+i+j+n	+a+b+c+d+e+f



Guide to Table 5-1

Notes; Net benefits measured in terms of areas defined in Figures 5-1 and 5-2. Net benefits are for firms that come into compliance as a result of a subsidy; net benefits are unchanged for firms that remain in either type of noncompliance. In cases where the attribution of net benefits is unclear, they have been classified as resulting from reductions in disposal risk.

Problems Associated with Subsidies

Our analysis thus far generally has been favorable to the use of subsidies in connection with hazardous wastes. Our simple analytic framework, however, has ignored several important problems. In particular, we have not taken account of the broader allocative inefficiencies of subsidies, nor have we addressed the problems likely to arise in designing practicable measures on which to base subsidy payments, especially for waste-minimization subsidies.

Generic Allocative Inefficiencies of Subsidies

Many early students of the use of economic incentives for environmental protection believed that subsidies and charges would yield equivalent results; at the margin, a subsidy for emission (or waste) reduction yields the same incentive for control that a tax on emissions (or waste) would. Indeed, under some circumstances, a subsidy can be thought of as a tax coupled with a lump-sum subsidy. More recent analyses, however, have shown that subsidies for reducing externalities suffer at least two serious drawbacks. First, they are likely to lower final-product prices, thus increasing production of those goods above socially efficient levels. Second, because of the inefficiencies associated with taxation to finance subsidies, sums that may appear to be simple transfers actually entail some deadweight losses.

Impacts on product prices. All of our analysis has focussed on the social costs and benefits per unit of output; we have assumed, implicitly, that the quantity produced of the final product is independent of the method used to regulate wastes. In fact, however, different methods of regulation will have different effects on production costs, and thus on product prices and quantities. Taxes or charges generally will raise production costs and prices, while subsidies, whether directed at disposal or waste minimization, will lower production costs and prices, thus increasing the quantity consumed.

To the extent that waste-related subsidies drive the prices of waste-intensive final goods below their social costs, such increases in quantity will be undesirable (Nichols 1984). The size of the welfare loss will depend on the share of total cost attributable to hazardous waste disposal and the own-price elasticities of demand in final goods markets.

Table 5-2 shows that typical output effects of a subsidy would be quite small. In 1984, solid and hazardous waste disposal comprised only one-tenth of one percent of the total value of shipments for U.S. manufacturers. Unsurprisingly, disposal costs were the largest for the chemical industry, but even there waste disposal costs were less than 2% times the average for all manufacturing, and still a very small fraction of total costs. Since even very large subsidies for compliance will not significantly reduce total costs, final goods prices would probably decline by an imperceptible amount.

Table 5-2. Gross Annual Costs for Solid and Hazardous Waste Disposal as a Percentage of the Value of Manufacturing Shipments, 1984

SIC	Manufacturing Sector	Percent of Value of Shipments
28xx	Chemicals	0.255
33xx	Primary Metals	0.237
26xx	Paper	0.217
32xx	Stone, Clay & Glass	0.174
38xx	Instruments	0.159
37xx	Transportation Equipment	0.159
20xx	Food	0.140
34xx	Fabricated Metals	0.120
-NA-	**Average for All Manufacturing**	**0.106**
30xx	Rubber & Misc. Plastics	0.103
24xx	Lumber & Wood Products	0.086
36xx	Electrical & Electronic Equipment	0.084
29xx	Petroleum & Coal	0.072
35xx	Nonelectrical Machinery	0.071
39xx	All Other Manufacturing	0.051

Source: CRS (1987: Figure 12).

Of course, this does not mean that output effects are negligible everywhere. Certain service industries, for example, may be characterized by both highly elastic demand and large fractions of total cost attributable to hazardous waste disposal. Nevertheless, the potential for worrisome output effects seems likely to be localized.[4]

<u>**Impacts on markets for other factors of production.**</u> In the isolated situations in which hazardous waste disposal does indeed comprise a

[4]The dry cleaning industry may be a good example where output effects are worth examining carefully. Demand is generally believed to be elastic (higher prices lead to less-frequent cleaning and more reliance on washable fabrics) and the cost of properly disposing of dry cleaning solvents (most commonly perchlorethylene) may be a large fraction of total costs. Note that the dry cleaning industry was strongly opposed to the elimination of the SQG exemption in the 1984 RCRA Amendments. <u>See</u> SQG Hearings (1983).

relatively large percentage of total production cost, subsidies for safe disposal or waste minimization may have important side effects in markets for other factors of production. A subsidy will cause a significant change in demand for other factors that are either strong substitutes or complements. If safe disposal is subsidized and this "other factor" is some collection of inputs that together describes a form of waste minimization, then the substitution induced by the subsidy would be undesirable; it would result in more rather than less hazardous waste generation. If instead the "other factor" is illicit disposal, then the resulting substitution effect is highly beneficial. The subsidy achieves precisely the kind of change in the input mix that regulators desire.

These relationships can be compactly summarized in the form of an elasticity measure. Let $\epsilon_{X,w}$ represent the cross-price elasticity of demand for some other input X given a change in the cost of hazardous waste disposal, w. The magnitude of $\epsilon_{X,w}$ depends on the values of three parameters: (1) the proportion of total costs attributable to hazardous waste disposal, K_w; (2) the elasticity of substitution between hazardous waste and the alternative input, X; and (3) the own-price elasticity of demand for the good produced by the firm, $\epsilon_{Q,P}$:[5]

$$\epsilon_{X,w} = K_w \left[\sigma_{X,w} + \epsilon_{Q,P} \right] , \tag{5-3}$$

[5]For a proof of this relationship, see Allen (1938: 505-508). In the usual two-input case, $\epsilon_{X,w}$ must be positive because X and w have to be substitutes. In a multiple-input world, however, there are both substitutes and complements. A negative value for $\epsilon_{X,w}$ indicates a complement.

Unless the share of total cost attributable to hazardous waste disposal (K_w) is relatively large, the values of the other parameters typically will not matter; subsidizing hazardous waste disposal, for example, will have little effect on firms' demands for other inputs -- including, by the way, the collection of inputs that together comprises waste minimization. As we indicated previously, across broad industrial classifications K_w is in fact quite low.

In those isolated areas in which K_w is relatively large, the effect of a subsidy on the demand for other inputs will therefore depend primarily on the relevant elasticity of substitution. For firms already in compliance with disposal regulations, the substitution elasticity that matters is between safe disposal and waste minimization. A low elasticity implies that a safe-disposal subsidy would have little deleterious effect on waste minimization. However, high values would argue against subsidizing safe disposal; firms that can easily reduce the amount of waste they generate may respond to a safe-disposal subsidy perversely, by generating more waste. Waste-minimization subsidies would be preferred under these circumstances, provided, of course, that they could be appropriately defined and targeted.[6]

The implications of the analysis are reversed for firms initially out of compliance with disposal regulations. A high elasticity of substitution between safe- and unsafe-disposal suggests that a subsidy on safe disposal could elicit significant improvements in disposal behav-

[6]We discuss definitional problems later in this chapter.

ior. Conversely, a low value implies that even large subsidies would have little effect.

Efficiency effects of financing subsidies. In keeping with most of the literature on economic incentives for environmental protection, we have treated subsidy payments as simple transfers that do not have efficiency implications, except insofar as they change the behavior of the firms towards which they are directed. Thus, to the extent that subsidy payments exceed the cost of reducing disposal risks or reducing waste generation, the cost to the government will be offset by a benefit to firms, with no change in social net benefits.

The problem, of course, is that the taxes used to finance subsidy payments are likely to have distorting effects in other markets. As is well-established in the public finance literature, raising $1 in revenues generally has a social cost will in excess of $1. Terkla (1979), based on work by Browning (1976) and Feldstein (1978), has estimated that at the margin, taxes on labor impose a cost of about $1.35 for each dollar of revenue raised and that taxes on capital are even more costly, imposing a burden of about $1.60 per $1 raised. In his extensive study of income tax reform, Bradford (1986) reports estimates of other researchers ranging from $1.17 to $1.65 per $1 raised. Such excess burdens may well be sufficient to offset the net benefits of subsidies shown earlier in this chapter, particularly if many firms already are in compliance; for those firms, the subsidy payments will tend to be very large relative to any waste-related net benefits derived.

This applies with equal force to both safe-disposal and waste-minimization subsidies. It argues that subsidies are unlikely to be desirable unless the waste-related net benefits are substantial and it is very difficult to achieve them by other means. Examples of such cases are likely to involve a large number of firms out of compliance with disposal regulations, using disposal methods that have much higher social costs than those associated with RCRA-approved methods.

Tax distortions also may argue for attempting to modify subsidy schemes to minimize payments to firms already in compliance. Payments, for example, might be limited to firms that could show that they were out of compliance with disposal regulations before the subsidy or that they had reduced wastes below levels that were otherwise economical for them. Such an approach might resemble efforts now made in connection with emissions trading or averaging programs to avoid the awarding of "paper credits."[7] These modifications pose problems of their own, however.

One of the strongest objections is a practical, administrative one; it would be extraordinarily difficult to determine which actions resulted from the subsidy and which would have occurred anyway. Moreover, if the regulatory agency were successful in doing so, it would be placed in the politically uncomfortable position of paying money to "bad actors" while denying funds to firms that had come into compliance prior to the subsidy program. The long-term incentive effects of such differentiation also would be undesirable; it would be in a firm's self-interest to

[7] Emission-reduction credits are, essentially, quantity-based subsidies. In exchange for reducing emissions below some level (typically defined by standards), firms receive credits that they can sell.

delay compliance and to devote resources to efforts to show that it would not have complied in the absence of a subsidy. Moreover, differential awarding of the subsidy could create inefficiencies by artificially altering competitive positions.

Despite these problems, it may make sense to design subsidies in some cases to reduce payments to firms already in compliance. Prime candidates would include waste streams that come from several different, easily distinguishable industry groups with very different compliance rates. In such cases, subsidies could be limited relatively easily to those segments with low initial compliance rates. Differentiation also would be more desirable if the different industry groups were not competing in the same final-product markets, so that differential subsidies would not create losses there.

The Problem of Defining What Qualifies for Waste-Minimization Subsidies

In our analytic framework in the first part of this chapter, we assumed that "safe disposal" and "waste minimization" are concepts that can be defined clearly enough in operational terms to make direct subsidy programs possible. For "safe disposal," this assumption seems reasonable; presumably the subsidy would apply to disposal at facilities in conformance with RCRA rules.[8] For waste minimization, however, the problem of definition is much more severe, and is likely to make it impossible to subsidize waste-minimization directly, forcing the regulator

[8] In theory, of course, it would be more desirable to have the disposal subsidy vary with the riskiness of the disposal method, rather than being a simple on-off determination. The more sophisticated approach, however, seems impracticable both politically and administratively.

instead to rely upon proxy measures. The use of such proxy measures, however, creates other serious problems.

As we discussed in Chapter 1, definitions of waste minimization range from the very broad (e.g., EPA's) to the exceedingly narrow (e.g., OTA's), with plenty of room for disagreement and confusion in between Unfortunately, the language used by Congress in the 1984 RCRA Amendments does not help resolve these disputes, inasmuch as all parties claim that their definition most accurately reflects Congressional intent.[10]

These definitional disputes highlight just how difficult it would be to devise appropriate units for measuring waste minimization and calculating subsidy payments. The information regulators would require is far beyond what they can effectively manage or responsibly comprehend. Thus, the same factors that inhibit the use of standards to directly regulate waste minimization also frustrate the development and implementation of direct incentives for waste minimization. The larger the subsidy becomes, the more it will look like an entitlement program for which exceptions, variances, special allowances, and judicial appeals must be allowed to preserve horizontal equity and substantive due process.

Moreover, a narrowly targeted waste minimization subsidy program must be administered at the firm or plant level, which implies very high administrative costs even in the absence of complicated measurement problems. There could be literally hundreds of thousands of subsidy ap-

---

[9]The definitions provided by EPA and OTA should not be construed as polar cases; broader and narrower definitions are certainly plausible.

[10]Congress defined 39 terms in RCRA Section 1004. Waste minimization, however, was not one of them.

plicants, each requiring extensive examination and oversight. A program large enough to simply attract most complying firms would have to employ a legion of regulatory officials, each properly trained in the intricacies of a wide range of industrial technologies as well as the details of the program itself. Since noncompliers will remain outside of the program unless subsidy payments are large enough to overcome their added compliance costs, a program capable of reaching most of them will have to be much larger in scope and cost -- larger, perhaps, by orders of magnitude.

Indirect Waste-Minimization Subsidies

Most of the waste-minimization subsidies proposed or implemented thus far are targeted on either capital or information. In part these foci reflect the difficulties of direct targeting. In addition, however, they appear to reflect beliefs about specific market failures in these areas.

Capital subsidies. Subsidies provided by government often are directed at capital. Such subsidies may take the form of low-interest loans often financed through the issuance of tax-exempt bonds, direct subsidies for capital expenditures, loan guarantees, or special depreciation rules for certain types of capital. Their appeal is due in no small part to the fact that it is often relatively easy to structure capital subsidies in ways that conceal or reduce their apparent costs.[11]

[11] For a lucid discussion of government involvement in credit markets and its implications for public accountability, see Leonard (1986).

It also, however, appears to reflect a widespread view that that government intervention is intrinsically legitimate and largely effective when it is targeted on decisions involving capital (Leone 1986: 72).

Subsidies that are limited to capital or to any other class of inputs suffer from the problem that they distort production decisions. This problem is well known and has been widely discussed in connection with other types of subsidy programs. In the case of public transit, for example, subsidies for capital but not operating expenses have encouraged excessive reliance on fixed rail systems as opposed to less capital-intensive modes, such as buses. In the case of sewage-treatment plants, subsidies targeted on capital are alleged to have led to the construction of overly expensive plants, with inadequate provision for operation. As a result, very costly plants operate at relatively low levels of efficiency because of poor maintenance (Schultze 1977: 57).

For firms that heretofore have not devoted much attention to waste minimization, the most cost-effective approaches are unlikely to require large capital expenditures. Only after firms have taken a variety of "housekeeping" and other steps are capital-intensive technology changes likely to be appropriate (National Research Council 1985). If this characterization is correct, then capital subsidies either will have little impact on the firms that we most want to reach -- those outside the regulatory system that engage in unsafe disposal and that have little incentive to minimize wastes -- or they will encourage firms to choose cost-ineffective methods of waste minimization.

Such problems are likely to be exacerbated by the fact that capital-related waste minimization subsidies probably would have to be limited not just to "capital," but to specific forms of capital because of problems in determining eligibility. The narrower the subsidy, the more inefficient it will tend to be.

Suppose, for example, that a firm can reduce waste generation in one of two ways, both of which require sizable capital expenditures. Method A involves redesigning the entire production process, including the purchase of new equipment that generates less waste and lowers production costs. Unfortunately, however, it is impossible to identify what portion of the cost of the new equipment relates to waste minimization.

Method B, in contrast, involves the purchase of single-purpose equipment for in-process recycling of materials, thus reducing waste generation. Unlike method A, there is no problem of allocating joint costs. In such an instance, the subsidy probably would have to apply only to Method B, thus providing an inappropriate incentive to choose it over Method A. Such a result would seem particularly ironic in light of the emphasis that waste-minimization advocates tend to give to finding integrated, "holistic" solutions, rather than focusing on narrow "end-of-pipe" technologies.[12]

Information subsidies. Providing information on waste minimization at little or no cost to firms has wide appeal. The traditional economic

[12]The obvious solution may appear to be to offer the subsidy for A as well as B. That creates at least two additional problems, however. First, the total size of the subsidy program grows, requiring more revenues and thus creating more tax distortions. Second, it biases the choice in favor of processes in which it is impossible to disentangle waste-minimization-related costs from other types of cost, so as to increase the expenditures that get subsidized.

rationale for information provision is that it often is a public good and, as such, will not be provided at efficient levels in the market. This rationale is strongest for supporting general research on waste minimization, particularly if such research is unlikely to result in patents that will allow private firms to capture its rewards. It also provides some foundation for disseminating general information to firms or trade associations. The public goods rationale, however, provides little or no rationale for public provision of firm-specific technical assistance.

Although it is hard to argue that firm-specific information is a public good, it may be justified as a proxy for subsidizing waste minimization more directly. Relative to capital subsidies, it has the advantage of less distortion in the choice of approaches to waste minimization. It has the disadvantage, however, of failing to provide a very large subsidy, and thus may have little, impact. In addition, because firm-specific technical assistance inevitably involves the identification of individual generators and often will include on-site visits, it may be relatively unsuccessful in attracting firms that have been out of compliance with regulations, which may fear that participation will expose them to possible prosecution and liability for past behavior.

Summary and Conclusions

In a world of imperfect compliance and costly or otherwise limited enforcement, subsidies may offer useful ways of encouraging firms to handle wastes more safely. Subsidies for RCRA-approved disposal methods offer a relatively simple and direct way of promoting safer disposal of hazardous wastes. Their primary drawback is that they discourage waste minimization.

Waste-minimization subsidies can rectify the problem of inappropriate marginal incentives for waste minimization and, if structured appropriately, also can encourage firms to engage in safer disposal. Definitional and administrative problems, however, render direct subsidies for waste minimization impracticable, and the most common proxy -- capital expenditures on waste minimization -- is likely to promote inefficient forms of waste minimization and is unlikely to secure much participation from firms now out of compliance with RCRA rules. Information subsidies are likely to have fewer drawbacks, but also are unlikely to have a major impact.

Both types of subsidies also suffer from the generic problems of distorting final-product prices (and thus encouraging excess production of waste-intensive goods) and of requiring the raising of revenues to finance them. Aside from the obvious problems of new expenditures during a time of heightened concern about budget deficits, the taxes needed to finance subsidies will create deadweight losses of their own, losses that easily can outweigh the net benefits they produce from safer disposal or reduced waste generation.

Chapter 6:

COMBINING TAXES AND SUBSIDIES INTO A COORDINATED REGULATORY INSTRUMENT

In this chapter we show how taxes and subsidies can be combined into a unified system that may improve incentives at both the waste generation and disposal stages. A special form of this combination -- the deposit-refund system -- is less flexible in certain respects, but enjoys enough popularity in other contexts to make it a promising strategy worthy of additional research. In Chapter 7 we analyze how such a system might perform if applied to the problem of used lubricating oil, an issue of current regulatory interest to the EPA.

Input Taxes as Proxies for Waste Minimization Subsidies

We showed in Chapter 2 that if compliance could be ensured, then the optimal level of waste minimization and disposal risk could be obtained through a waste-end tax. Such a tax would raise the cost of RCRA disposal, however, and might well drive some firms into illegality. The problem was that some generators possess the ability to evade the tax, with perverse consequences for net risk reduction.

In many cases, regulators may find it much easier to tax certain chemical feedstocks and other materials that are inputs to hazardous waste-generating production processes. The number of sellers may be relatively small, and in addition they may be subject to regulatory oversight for other reasons. By levying taxes at the point of production, firms that generate hazardous wastes will be severely limited in their capacity to evade the incentive.

For input taxes to function as perfect proxies for taxes on waste generation, the input used as the target for the tax must be related by fixed-proportions production technology to the hazardous waste stream.[1] Strictly speaking, this condition is rarely, if ever, met in practice. Even so, it is easy to find cases that closely approximate it. For example, solvent wastes derive only from virgin or recycled materials of the same type. As a firm reduces the amount of solvent it uses, its level of waste generation declines by a proportionate amount. Similarly, used lubricating oil can only be derived from its uncontaminated precursor, whether virgin, re-refined, or synthetic. The amount of lubricating oil generated as waste is proportional to how much new oil is installed.[2]

Suppose that regulators levy a tax on input X at the rate t. If α units of waste are generated per unit of input used and that relationship is fixed, then such an input tax is equivalent to a tax on waste generation equal to:

$$T = \alpha t \qquad (6\text{-}1)$$

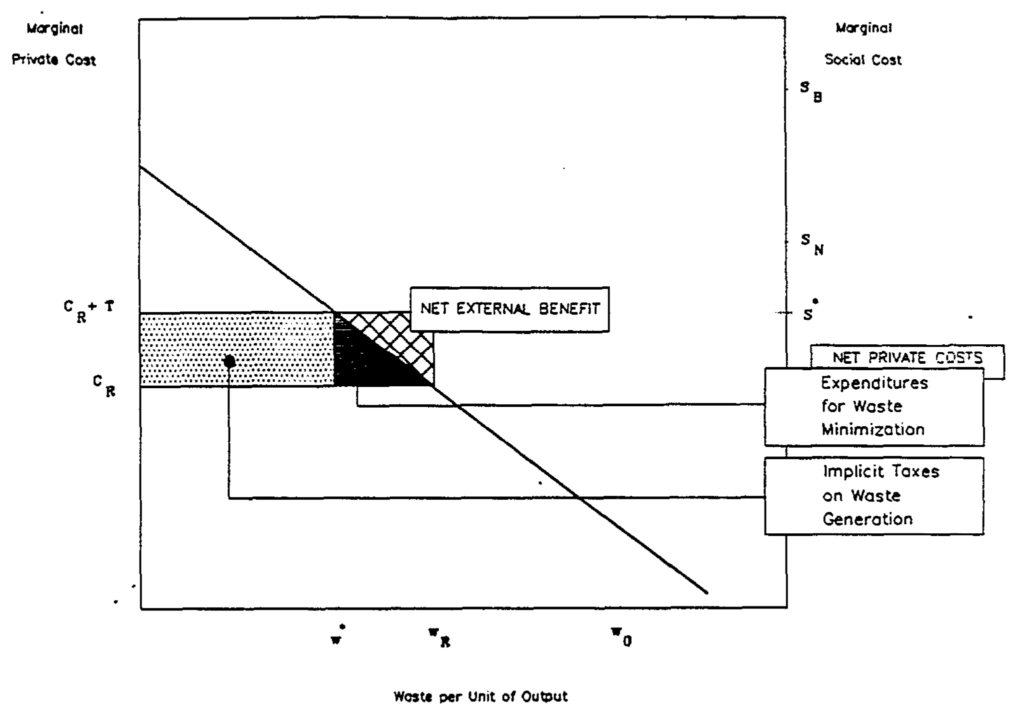
[1]The production technology need not exhibit fixed-proportions with respect to *all* inputs, nor must there be fixed-proportions between inputs and output. Fixed-proportions need only characterize the relationship between hazardous waste generation and whatever input or output measure is used as the proxy.

[2]This proportion varies greatly depending on the application. Apart from expensive engine overhauls or more fundamental changes in internal combustion technology, however, the only way to reduce the generation of used automotive motor oil without reducing the consumption of new oil is to perform oil changes less frequently -- a strategy that may impose substantial costs in the form of premature engine wear.

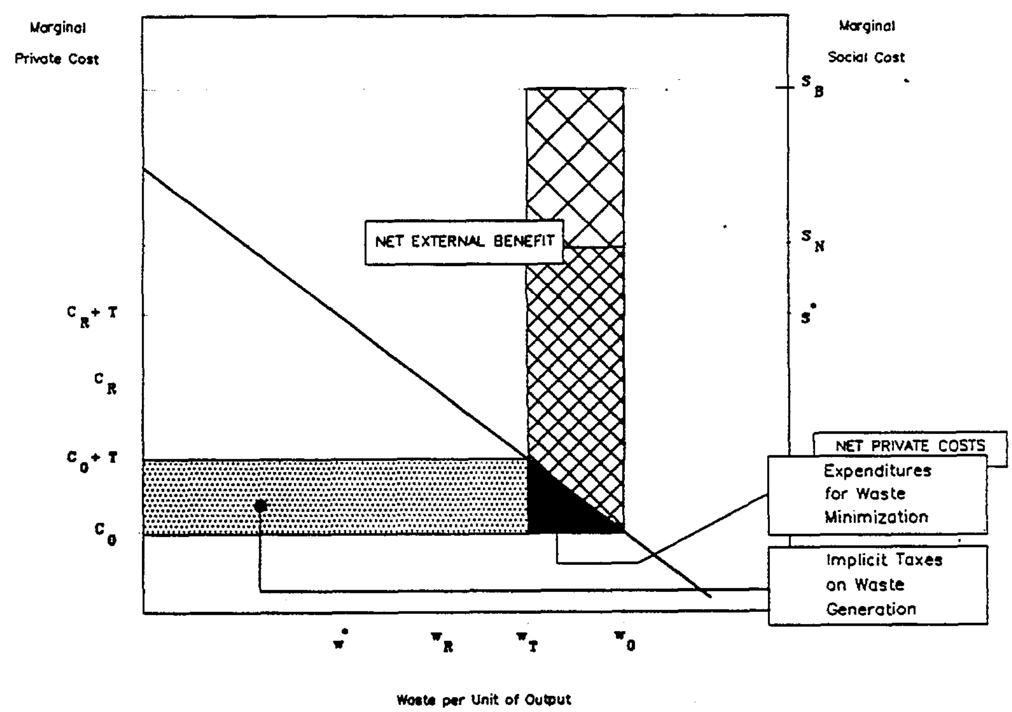
In general, for simplicity we will discuss such a tax in terms of its effective rate on waste, T.

Panel (a) of Figure 6-1 shows how the tax affects a complier having hazardous waste disposal costs equal to c_R per unit of waste (per unit of output). The tax raises the firm's opportunity cost of continued waste generation to c_R + T, an amount that precisely equals the full social costs of waste generation combined with safe disposal, s^*. This motivates the firm to reduce wastes from w_R to w^* at a net cost equal to the area of the darkly-shaded triangle. The firm pays an implicit tax on waste generation of Tw_R per unit of output -- the area of the lightly-shaded rectangle. The net external benefit consists of the value implicit in this reduction in hazardous waste generation, the area of the cross-hatched triangle located above the demand curve. As long as the fixed-proportions relationship between hazardous waste generation and the taxed input persists (and the proportion itself is constant), the input tax will create the same incentive to reduce waste generation as a waste-end tax. Unlike the waste-end tax, however, the input tax cannot be evaded by withdrawing from the regulated hazardous waste disposal system.

The effect is similar among noncompliers, as Panel (b) illustrates. A noncomplier cannot circumvent the tax, which raises the firm's opportunity cost of waste generation to c_O + T. This induces a reduction in waste generation from w_O to w_T. Net additional expenditures for waste minimization equal the darkly-shaded triangle. The noncomplier also pays implicit taxes equal to Tw_T on the remaining hazardous waste generated -- the area of the lightly-shaded rectangle. All other



(b). Compliers



(b). Noncompliers

Figure 6-1. Effects of a Targeted Input Tax

things being equal, the noncomplier has a higher tax burden per unit of output because it generates more waste. This comports with the notion that noncompliers should pay more because of the greater damages they impose on the environment. Ideally, the tax per unit of *waste* would also be higher inasmuch as the noncomplier creates higher residual risks even for the same amount of waste generated. Unfortunately, a simple input tax cannot accomplish this goal.[3]

Waste generation by the noncomplier creates more residual risk and higher social costs -- S_N if the firm engages in what we have called "ordinary noncompliance," and S_B if it participates in the black market. Thus, the input tax captures a net benefit that is larger than what was obtained from the complying firm. This gain equals the narrow cross-hatched area above the demand curve if the firm practices ordinary noncompliance, or both cross-hatched areas combined if it operates in the black market.

In addition, the input tax will increase the price of final goods, the magnitude of that effect depending on the taxed input's share of total production cost and demand elasticities in final goods markets. Unlike both safe-disposal and waste-minimization subsidies, however, the input tax *raises* rather than *lowers* final goods prices -- a result that is consistent with efficient incentives in these markets as well.

[3] In Figure 6-1, both the absolute reduction in waste generation and net expenditures on waste minimization appear to be identical for compliers and noncompliers. This is an artifact of the linear demand curve. Suppose instead that the curve were convex. Then the absolute reduction in waste generation would be greater for the noncomplier (i.e.) $w_0 - w_T > w_R - w^*$).

Combining Tax and Subsidy Instruments

Unfortunately, input (or output) taxes used as proxies for waste generation cannot alter disposal incentives. Firms that initially engage in some form of noncompliance will continue to do so after the tax is levied. The tax cannot induce a firm to switch to compliance because such a change in behavior would not reduce the firm's tax burden. Compared to a conventional waste-end tax, however, this inability to alter disposal incentives should be viewed as a benefit: the input tax cannot cause a firm to backslide into the black market, because doing so would not reduce the firm's tax burden.

As we showed in Chapter 5, safe-disposal subsidies can enhance the rate of compliance with disposal regulations. They make it relatively less expensive to use disposal methods that result in significantly reduced external residual damages. Unfortunately, they also encourage more waste generation.

One possible remedy for this conundrum is to combine the input tax with the safe-disposal subsidy. The tax levied up front would create an incentive for waste minimization, while the safe-disposal subsidy would lower the cost of complying with disposal regulations. How large must be the tax and subsidy to cause a representative noncomplier to switch to an approved waste disposal method while simultaneously achieving efficient incentives for waste minimization?

Consider first a small tax-subsidy program in which T_1 represents the input tax and Y_1 denotes the safe disposal subsidy. To achieve efficient incentives for waste minimization, the difference between the tax and subsidy should equal the level of residual external damage resulting from safe disposal. That is, when the two instruments are applied simultaneously to a firm initially in compliance with disposal regulations, the net effect should be to increase the opportunity cost of waste generation from C_R to S^*. Thus, efficiency requires that:

$$T_1 - Y_1 = S^* - C_R = E^* , \tag{6-2}$$

or

$$C_R + T_1 - Y_1 = C_R + E^* = S^*, \tag{6-3}$$

Thus, for firms that comply, any values for T_1 and Y_1 are acceptable so long as Equation 6-2 remains satisfied.

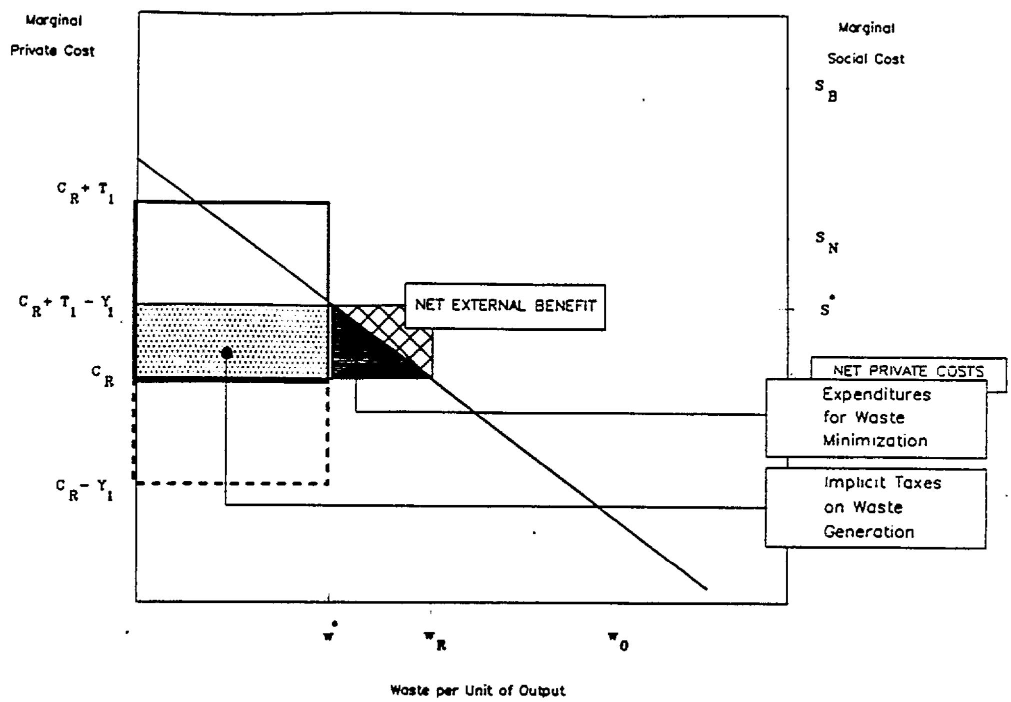
This result can be seen in Panel (a) of Figure 6-2. The input tax raises the complying firm's apparent cost of waste generation to $C_R + T_1$. However, the safe-disposal subsidy lowers the cost of approved disposal to $C_R - Y_1$. The net effect of these countervailing instruments is to raise the opportunity cost of waste generation and safe disposal to $C_R + T_1 - Y_1 = S^*$. The firm responds by reducing waste from w_R to w^*. Net private costs and external benefits are precisely the same as those illustrated in Panel (a) of Figure 6-1; to the initial complier, the tax-subsidy combination is functionally identical to a simple input tax. Input tax payments equal T_1w^*, the area of the transparent rectangle surrounded by the thick solid line. Safe disposal subsidy receipts equal Y_1w^*, the area of the transparent rectangle surrounded by the thick dashed line. The difference between tax payments and subsidy receipts is equivalent to a tax on waste generation equal to the value of residual external damage resulting from safe disposal.

How will the noncomplier respond? This depends only on whether the safe disposal subsidy reduces the cost of approved disposal below the firm's cost of continued noncompliance. Thus, the firm will switch to compliance only if:

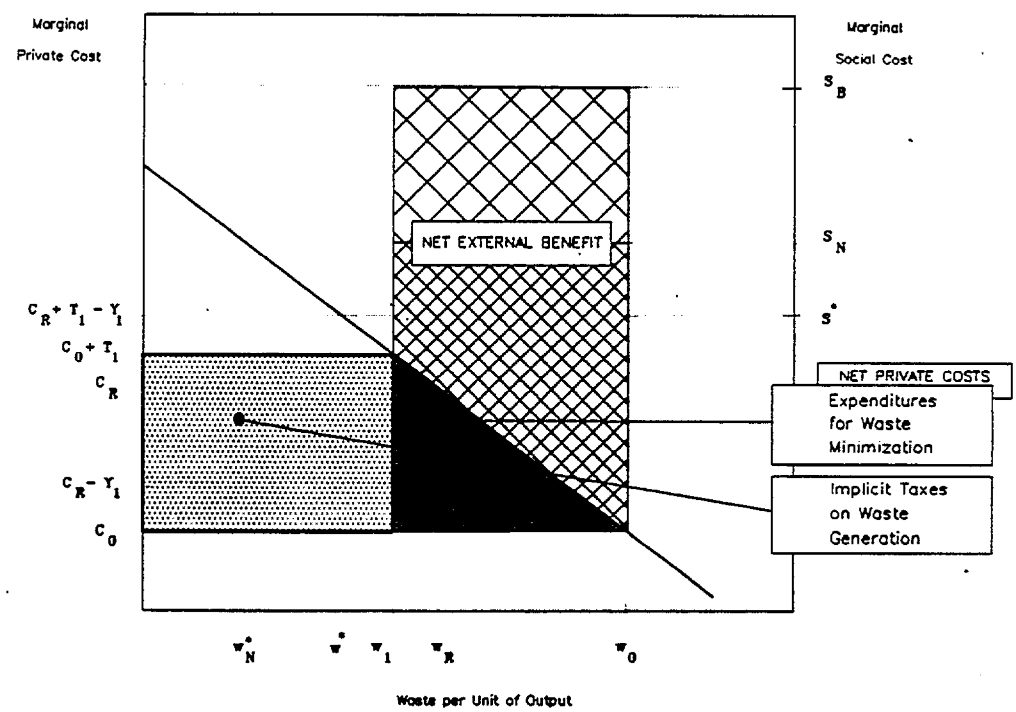
$$C_R - Y_1 < C_0 \ . \tag{6-4}$$

Panel (b) in Figure 6-2 illustrates the case in which Equation 6-4 is not satisfied. If the firm were to switch to compliance; then the net cost of safe disposal would equal $C_R - Y_1$. In the diagram, however, this amount is still greater than C_0, the firm's cost of continued noncompliance. Despite the subsidy, switching to compliance is not attractive.

However, the firm will still have to pay the input tax, which is large enough to induce a sizable reduction in hazardous waste. The tax increases the noncomplier’s opportunity cost of waste generation from C_0 to $C_0 + T_1$. This induces a reduction in waste generation from w_0 to w_1. Expenditures for additional waste minimization are substantial, equal to the area of the darkly-shaded triangle. In addition, the social gains from this reduction in waste generation may be substantial. The net external benefit equals the area of one or both of the cross-hatched



(a). Initial Complier



(b). Initial Noncomplier, No Switch to Compliance

Figure 6-2. Effects of a Small Combined Tax-Subsidy Instrument

regions above the demand curve, depending on the characteristics of the firm's initial noncompliance behavior. Thus, even if the tax-subsidy is too weak to cause a firm to switch to compliance, it may still yield a considerable net benefit because of the incentive it provides non-compliers to reduce waste.[4]

Clearly, the larger the tax and subsidy rates the greater is the likelihood that the combined instrument will induce noncompliers to switch. Consider, for example, the tax rate T_2 and the subsidy Y_2, where $T_2 = T_1 + k$ and $Y_2 = Y_1 + k$, k denoting a constant. Efficient waste minimization incentives are preserved because the difference between the tax and subsidy rates is unchanged:

$$\begin{aligned} T_2 - Y_2 &= (T_1 + k) - (Y_1 + k) , \\ &= T_1 - Y_1 , \\ &= S^* - C_R , \end{aligned}$$

$$T_2 - Y_2 = E^* . \tag{6-5}$$

The opportunity cost of waste generation net of both the tax and the subsidy remains equal to the full social cost of safe disposal:

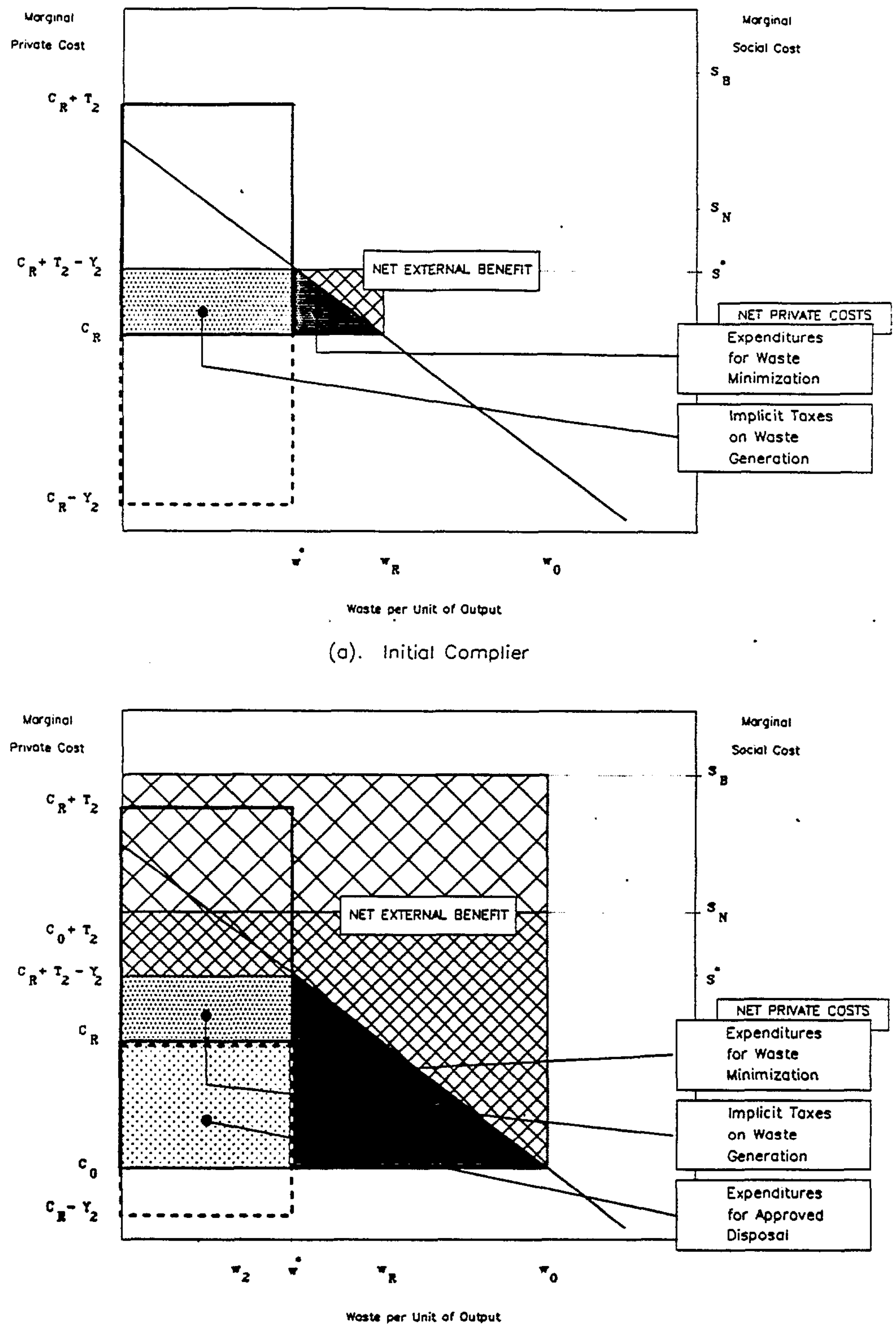
[4]The input tax depicted in Figure 6-2 is not large enough to reduce the representative noncomplier's level of waste generation to the optimal amount, which is either w_N^* or zero depending on whether the firm engages in ordinary noncompliance or black market disposal.

$$C_R + T_2 - Y_2 = C_R + (T_1 + k) - (Y_1 + k) ,$$
$$= C_R + T_1 - Y_1 ,$$
$$= C_R + E^* ,$$
$$C_R + T_2 - Y_2 = S^* . \qquad (6\text{-}6)$$

These are the same conditions that were derived in Equations 6-2 and 6-3.

Figure 6-3 illustrates the effects of this larger tax-subsidy instrument. The complier's case is presented in Panel (a). The tax increases the apparent cost of waste generation to $C_R + T_2$. However, this increase is matched by an identical rise in the amount of the safe disposal subsidy, which reduces the net cost of approved disposal to $C_R - Y_2$. The absolute amount of input taxes paid is equivalent to the area of the transparent rectangle surrounded by the thick solid line. It is larger than in the earlier case, but the full amount of this increase is also reflected in a larger safe disposal subsidy, the area of the transparent rectangle surrounded by the thick dashed line. Thus, providing that it has been designed so as to maintain efficient incentives for waste minimization, the larger tax-subsidy program has no additional effect on a compliant firm. In theory, it would be indifferent between the two incentive instruments.[5]

[5]The larger the tax and subsidy rates, the more important it will be to achieve accurate targeting and measurement. Unit transactions costs may also become larger. We discuss these issues more extensively in the context of the case study in Chapter 7.



(a). Initial Complier

(b). Initial Noncomplier, Switch to Compliance

Figure 6-3. Effects of a Large Combined Tax-Subsidy Instrument

In Panel (b) of Figure 6-3 we depict the case where the larger tax-subsidy instrument is in fact sufficient to cause the noncomplier to switch. The larger subsidy makes the net cost of safe disposal $C_R - Y_2$. Since this amount is less than the cost of continued noncompliance, C_0, the firm will shift to an approved waste disposal method so as to claim the subsidy. The opportunity cost of waste generation and safe disposal becomes $C_R + T_2 - Y_2 = S^*$, an amount that is less than $C_0 + T_2$ -- the opportunity *cost* if the firm persists in noncompliance. The firm reduces waste generation to w^*, the same amount of waste (per unit of output) that compliers generate and an amount that is less than what it would be induced to generate by the small tax-subsidy program. The firm increases its net expenditures on waste minimization by a larger amount, and spends for safe disposal a large fraction of what it otherwise would have had to pay in implicit taxes.

As expected, the large tax-subsidy program offers a greater net external benefit. If the firm initially engaged in ordinary noncompliance, the instrument captures all of the narrow cross-hatched area in Panel (b).[6] Both cross-hatched areas are captured if the firm initially participated in the black market. Once a firm has been induced to switch into the regulated waste management system, further increases in tax and subsidy rates have no further effect. Beyond the firm's switch point it becomes a complier, and as such it will be indifferent to increases in the level of tax and subsidy rates. Any such increases

[6] The rectangular portion of this area that is located to the left of w^* is the social gain from shifting w^* units of waste into safe disposal. Of the remainder, part is due to waste minimization and part can be attributed to either waste minimization or safe disposal.

do mean larger tax payments and subsidy receipts, but no net increase in costs. However, raising tax and subsidy rates intensifies the pressure on recalcitrant noncompliers to change their ways.

The Optimal Tax-Subsidy Instrument

Strictly speaking, for compliance with disposal regulations to be assured, the optimal incentive consists of a simple tax on waste disposal based on residual external damage. Because this approach is infeasible, we search for the combination of input tax and safe disposal subsidy that obtains the greatest improvement in social welfare possible under the circumstances. Thus, the tax-subsidy device is a "second-best" remedy. All the usual caveats concerning optimization in the absence of a competitive equilibrium apply. What we shall term "optimal" should be construed in this more limited context.[7]

If compliance with disposal regulations were always socially desirable, there would be no limit to the magnitude of the tax-subsidy instrument. Full compliance could require regulators to drive the cost of safe disposal to zero, or even below for wastes that can be profitably (albeit illegally) recycled. More likely, however, the marginal social benefit of eliciting a switch to compliance will, at some point, fall below the marginal social cost of making that switch. The identity of the optimal tax-subsidy combination will depend on the magnitude of the residual external damage from improper disposal.

[7] If the necessary conditions for optimality are violated anywhere, then endeavoring to satisfy them in one sector of the economy (e.g., the market for safe hazardous waste disposal) does not ensure a net increase in social welfare; broader intervention in otherwise efficient markets may be warranted. _See_ Lipsey and Lancaster (1956).

For simplicity, assume that there are but two firms, one complier and one noncomplier. Denote the external residual damage from non-compliant disposal as E_0, where E_0 equals the difference between the full social cost and the firm's private cost of noncompliant disposal.

Now, the marginal external benefit of shifting a unit of waste into compliance equals the realized reduction in residual external damages, or $E_0 - E^*$. Thus, the optimal (second-best) subsidy, Y^*, equals:

$$Y^* = E_0 - E^* \; . \tag{6-7}$$

The magnitude of the optimal subsidy is therefore equal to the sum of the absolute value differences in social and private costs:

$$\begin{aligned} Y^* &= E_0 - E^* \; , \\ &= (S_0 - C_0) - (S^* - C_R) \; , \end{aligned}$$

$$Y^* = (S_0 - S^*) + (C_R - C_0) \; . \tag{6-8}$$

The optimal tax, T^*, is derived from the condition first presented in Equation 6-3 in the context of the hypothetical small tax-subsidy program. Substituting the expression for the optimal subsidy derived in Equation 6-7 yields:

$$\begin{aligned} C_R + T^* - Y^* = C_R + E^* &= S^* \; . \\ C_R - T^* - (E_0 - E^*) &= S^* \; . \\ T^* &= S^* - C_R - E^* + E_0 \; . \end{aligned}$$

$$T^* = E_0 \; . \tag{6-9}$$

Thus, the optimal tax is simply the external residual damage caused by disposing of a unit of waste outside of the regulated waste management system, The greater the risk posed by noncompliant disposal, the larger will be the optimal tax rate.[8]

An important aspect of the tax-subsidy instrument is the limited amount of information regulators need to establish appropriate rates. Admittedly, the task of estimating residual external damages is not an easy one, what with the myriad uncertainties surrounding chemical risks and exposure levels. These difficulties plague every regulatory strategy, of course, whether it is based on economic incentives or traditional design or performance standards. Informed judgments concerning relative risks simply cannot be avoided.

To implement a tax-subsidy instrument, however, regulators need not be concerned with firm-specific details beyond estimating residual external damages. Production technologies, cost functions, final goods markets, and other similar data that are critical to the task of designing standards are irrelevant for setting tax and subsidy rates.

Other Responses of Noncompliers

Under a standards-based regime, firms can plausibly argue that they cannot afford to implement the changes mandated by regulators. The rules might well be appropriate for the average firm, they might claim, but our enterprises are different in certain critical respects that regulators have not taken into account. Regulatory design thus becomes embroiled in disputes over the nebulous concept of "affordability."

---

[8]If ordinary and black market noncompliance exist simultaneously, then the optimal subsidy rate will depend on the distribution of firms in each noncompliance category and the residual external damage reductions obtained as they are sequentially brought into compliance.

Standards that threaten to drive firms out of business often appear on their face to be excessively stringent, and intense political pressure may be brought to bear on regulators to lighten the burden so as to keep these firms alive and perhaps save politically sensitive jobs. The affordability argument can be most persuasive, even in cases where closure is both economically and environmentally desirable.

The issue of affordability is largely irrelevant when economic incentives are the chosen regulatory approach. Firms that cannot afford to pay for the external damages they cause have no economic basis for protection. This is particularly evident if a tax-subsidy instrument is applied in the case of hazardous wastes. As we showed in Chapter 2, existing incentives for waste minimization are quite powerful. Absent noncompliance there is little need for additional government intervention. Thus, the optimal tax-subsidy instrument stands to impose only trivial additional costs on firms that already comply with RCRA rules. Substantial new burdens are imposed only on noncompliers, both those that respond by switching to compliance and those that continue to resist. In effect, the tax-subsidy instrument penalizes regulatory evasion, both past and present. Past violators must pay a hefty one-time charge to gain legitimacy; current violators pay implicit taxes equivalent to the expected value of the residual external damages they create by refusing to comply. Thus, complaints concerning "affordability" seem likely to arise principally among those firms that heretofore failed to abide by the rules. These firms are faced with an unpleasant choice: the tax-subsidy system will impose large costs on them if they remain silent, and may well drive them out of business, but seeking special consideration signals a high probability of past misconduct, actions that can be punished through regulatory- and law-enforcement channels.[9]

Manifestations of Combined Tax-Subsidy Instruments: Deposit-Refund Systems

The progress of incentive-based regulatory strategies has been considerably more rapid in the environmental economics literature than it has been in practice. Given policy makers' resistance to relatively simple incentive instruments, it may seem highly improbable that they could be motivated to support a more complex strategy such as a tax-subsidy regime. Barriers to an explicit tax-subsidy may seem insuperable in light of the public's considerable opposition to taxes and the political difficulty of directly subsidizing hazardous waste disposal. The few times that coordinated tax-subsidy schemes have been proposed they have not fared well. One memorable example is the Carter Administration's National Energy Plan of 1977, the purpose of which was to raise the relative price of energy without imposing a net increase in taxes. Several features of this plan involved tax-subsidy instruments, but they were soundly defeated in the political arena.[10]

---

[9] Shut-downs caused by the tax-subsidy instrument enhance efficiency because they remove from the market firms that cannot survive once they are compelled to bear the full social cost of their production. Moreover, shut-downs are especially effective ways to reduce risk.

[10] One element of the plan would have taxed gasoline $0.50 per gallon to raise its price closer to world levels. To avoid transferring considerable wealth to domestic oil producers, the proceeds of the tax would have been rebated via reduced payroll taxes. Friedman (1984; 92-94) estimates that gasoline consumption would have declined between six and 12 percent net of the income effect due to the rebate.

Ironically, one form of tax-subsidy has become increasingly popular: the deposit-refund system. At least nine states mandate deposit-refund systems for beverage containers.[11] In Massachusetts, there is a proposal under consideration by the legislature to create a similar system for used lubricating oil. Recently, Suffolk County, New York enacted a deposit-refund system for lead-acid automotive batteries.

Analytically, deposit-refund systems can be considered constrained forms of tax-subsidy instruments. They differ in several important respects that make them less flexible (and hence, less efficient) but more politically palatable. In the remainder of this section we discuss briefly the nature of these differences and the implications they have for regulating hazardous waste generation. In Chapter 7 we develop a model of a deposit-refund system and apply it to a particular hazardous waste stream -- used lubricating oil.

Existing Deposit-Refund Systems

Deposit-refund systems appeared first in private market contexts. Market-initiated deposit-refund systems arise in situations where sellers want to expand the boundaries of the transaction to encompass secondary aspects of buyers' behavior that are hard to monitor *ex post*. In addition to the sales price, buyers make a separate payment that is held by the seller pending the fulfillment of specific post-sale contractual obligations. Once these obligations have been properly dis-

[11] These nine states are: Connecticut, Delaware, Iowa, Maine, Massachusetts, Michigan, New York, Oregon, and Vermont. California has a somewhat different system in which redemption values are allowed to fluctuate according to specified conditions.

charged the payment is returned. Deposit-refund systems are thus intended to overcome certain principal-agent problems that, occasionally arise within the context of market exchange.[12]

Examples of deposit-refund systems are easy to find. Firms that rent equipment or vehicles often charge a refundable deposit to ensure that users return property on time and in proper working order. Bottled water is sold in special refillable containers, so a deposit is typically levied to discourage consumers from using these containers for some other purpose. Landlords require renters to provide security deposits that become equivalent to liquidated damages in the event that the tenant breaches the rental contract. In each of these cases buyers have substantial or complete control over outcomes that sellers consider relevant. Traditional insurance contracts are generally infeasible in these cases because the underlying problem is asymmetrical information and incentives rather than true uncertainty. Risk can only be shifted rather than spread. Deposit-refund systems force risk-bearing upon those parties with the best information and influence over subsequent events.

Governments occasionally have a stake in these transactions. Ironically, economists had yet to develop explanations for the existence of these systems by the time policy makers intuitively understood what

[12] Similar instruments also are used as vehicles for price discrimination. For example, tire manufacturers offer trade-in allowances based on the proportion of rated miles a tire actually delivers in service. The difference between the actual mileage and the rated mileage is converted into a discount on a replacement purchase. In effect, this practice enables manufacturers to sell tires at a lower price to repeat customers and maintain market share. It need not have anything at all to do with tire performance.

could be achieved through their use.[13] Public policy objectives may argue for modifying the structure of an existing market-based deposit-refund system, reviving one rendered obsolete by changes in technology or preferences, or possibly creating a new one. Some communities, concerned that landlords might be profiting unfairly from the security deposits they retain, require that tenants receive annual interest payments on them. Governments have greatly expanded the use of performance bonds in construction projects to provide security against breach of contract. Such bonds constitute one of the few ways hazardous waste treatment, storage, and disposal facilities can satisfy RCRA financial responsibility requirements. Many state governments have enacted legislation to revive deposit-refund systems for certain classes of beverage containers, systems that had faded away due to changes in bottling technology and consumer demands for disposability.

Special Attributes of Deposit-Refund Systems

Deposit-refund systems differ from more generic tax-subsidy instruments in at least three ways. First, there is generally a transparent linkage between the deposit and the refund that may be best described as a property right. Second, deposit and refund rates typically are set equal in nominal terms, a balancing mechanism that conforms to the linkage arrangement and reinforces its economic purpose with political and psychological legitimacy. Third, tax-subsidy instruments can only

[13] The only book-length analysis of deposit-refund systems is Bohm (1981). It is to some extent an attempt to explain the economics behind deposit-refund systems that already had been applied to containerized beverages without much awareness of their economic effects.

be brokered by governments to effect public policy outcomes, whereas deposit-refund systems can be run in either the public or private sectors, and either for public or private purposes. Each of these differences deserves brief elaboration.

Deposit and refund linkage. The relationship between the deposit and the refund is typically transparent to both buyer and seller; there are rarely intermediaries involved. Paying the deposit creates an implicit property right to the refund, an asset that is liquidated only upon redemption. In contrast, for a tax-subsidy instrument to be effective market participants need not understand the connections between the individual components.

This need for linkage clearly restricts the range of applications for which a deposit-refund system may be suitable. However, it also offers certain intangible advantages. First, the pejorative connotations associated with taxes and subsidies are absent. Deposits are not perceived as taxes, and refunds are not viewed as subsidies. Instead, they are widely interpreted as extensions of the terms of market exchange based on legitimate economic or political considerations. Political legitimacy may in fact induce a degree of voluntary participation far in excess of what might be obtained from an analytically identical but less obvious tax-subsidy instrument.[14]

Nominal rate equivalence. Typically, deposit and refund rates have to be equal in dollar terms to achieve this property-rights form of

[14] For example, beverage container deposit-refund systems enjoy widespread public support and high voluntary participation. Many consumers expend more resources in the act of redeeming empties than the cash value of the refunds they receive.

linkage. Such nominally identical rates seem simple and intuitively fair, whereas deviations from this condition may be predicated upon complex calculations that escape the intuition of virtually everyone. The practice is so prevalent that exceptions to the rule should attract more attention than the practice itself.

In market-based deposit-refund systems, this phenomenon has no special economic significance. If, after deducting the expenses of operating the system, competitive sellers collect more in deposits than they return in refunds, they will lower prices accordingly and thereby make the real deposit less than the nominal amount. Similarly, if operating the system entails nontrivial administrative costs, then sellers will raise prices sufficiently to cover these costs and raise the real deposit above its nominal amount. As long as a competitive price system is available to make adjustments, nominal rate equivalence thus will be of no consequence.

A similar analysis can be made concerning government-mandated systems operated by private entities. Prices simply adjust to account for discrepancies between mandated and market-determined deposit and refund rates. The case of mandatory beverage container deposit-refund systems provides a useful example. Superficially, these systems may appear to generate windfall profits to the parties required to collect deposits and pay refunds -- typically wholesalers and bottlers. If unclaimed deposits fail to cover handling costs and administrative expenses, then these firms will raise prices and thereby force consumers to bear a portion of the deposit indirectly. But if unclaimed deposits exceed these costs, then competitive pressures will force prices downward until windfall gains are **exhausted.**[15]

Nominal rate equivalence may have significant economic consequences, however, if government attempts to operate the system. Leaving aside the issue of administrative costs, nominal rate equivalence typically will result in inefficient incentives. Unlike systems operated in the private sector, there are usually no market processes beneath the system that are capable of making these price adjustments. Under a government-run system, nominal rate equivalence is consistent with efficiency only under highly restrictive conditions. First, there can be no residual external damage resulting from whatever behavior constitutes redemption. In a hazardous waste-related application, this would mean that RCRA disposal would have to fully extinguish risks to human health and the environment. Second, there must be a one-to-one correspondence between the number of physical units subjected to the deposit and the number of physical units eligible for the refund. Even if these conditions hold, efficiency can be achieved only if the deposit and refund are set equal to the expected value of residual damages resulting from noncompliant disposal. Otherwise, the need for nominal rate equivalence means that either the deposit rate will be too low or the refund rate will be too high. In the absence of these conditions, the system will

[15]**Governments** have occasionally sought to tax away these "windfall profits." If expropriation ever proves successful, then wholesalers will behave as if these funds do not partially offset handling and administrative costs;' they will raise prices still further. The illusion of windfall profits is one of several significant side-effects of nominal rate equivalence based on the intuitive property-rights linkage. This illusion is abetted by the separate accounting and reporting procedures that are typically required, practices which ensure that total unclaimed deposits grow ever larger over time.

be required to either offer refunds that are too large or levy deposits that are too small. In either case, efficient incentives will be lost.

This is not to say that a government-run deposit-refund system suffers any financial inadequacy. As we shall indicate below, the revenue generating capacity of deposit-refund systems may make government operation a particularly attractive option.

The identity of the brokerage agent. Tax-subsidy instruments must be administered directly by government, because only government enjoys the power.of taxation. In contrast, deposit-refund systems can be implemented with or without government involvement. Of course, if such a system were intended to overcome an environmental externality, then it would not be viable without governmental initiative.

Like the tax-subsidy instrument, a deposit-refund system could generate a substantial surplus of receipts over disbursements. For this reason, policy makers may be particularly interested in establishing the government as the broker. Any surplus collected from the program can be used to offset incentive-distorting taxes or reduce the deficit. In addition, some of these funds may be needed to cover administrative costs.

An alternative approach is to mandate that certain entities in the private sector perform the brokerage function, particularly if it is politically impossible to earmark the surplus for program administration. This may not save on the total cost of administration, but it would reduce the amount that must be paid out of government agency budgets.[16]

From an efficiency perspective, the decision as to where the system should be brokered should be based on comparative advantage across a range of important criteria. Choosing government brokerage enhances public accountability and reduces the need for regulatory oversight, but it may also subject the system to delays and inefficiency. Private sector brokerage can be expected to maintain the tightest cost control, but it also raises the threat of conflicts in interest between brokers and regulators. Unfortunately, there is no simple solution to this question.

Problems with Technology and Targeting[17]

So far, we have assumed that there exists a fixed-proportions relationship between the item subjected to taxation and the generation of hazardous wastes. Also, we have assumed that there are no significant technical or administrative problems associated with targeting either the input tax or the safe-disposal subsidy. When these assumptions hold, the tax-subsidy instrument appears very attractive because it simultaneously enhances efficient waste minimization and safe disposal.

---

[16] Policy makers can have the best of both worlds by mandating private sector brokerage, then taxing away unclaimed deposits. We shall discuss the implications of this strategy in Chapter 7 in the context of a Massachusetts proposal to establish a deposit-refund system for used lubricating oil.

[17] In the following discussion, deposits and refunds can be considered synonymous with taxes and subsidies.

In this section we address the issue of how well the instrument might perform if these assumptions are violated.

Fixed-Proportions Technology

The fixed-proportions assumption enables the input to act as a perfect proxy for hazardous waste generation. This phenomenon seems to apply perfectly in the case of beverage containers. Empty containers derive only from filled ones, and there is a one-to-one correspondence between the number of containers filled and the number emptied.[18] However, the case of used motor oil is more ambiguous. The ratio of new oil installed to used oil generated is not fixed, but instead varies considerably across vehicles. Nevertheless, the ratio may be tightly distributed due to obvious similarities in technology and the relationship between oil use and engine performance.

The capacity to substitute away from the taxed input (and by implication, hazardous waste generation) is an important and desirable attribute. Waste minimization depends upon it. In both the beverage container and used oil examples, however, little substitution is technically feasible. Containerized beverages cannot exist without the container; motors will require lubrication as long as there is friction.

[18] There is, however, no such correspondence between the number of filled containers and the number of littered empties -- a more accurate description of the relevant waste stream. In practice, this creates problems because deposit-refund systems for beverage containers mandate a new and more expensive disposal technology merely for purposes of administering the system -- that is, empties must be redeemed at the store rather than at the nearest refuse barrel. Fortunately, this problem does not arise with respect to hazardous wastes. An appropriate analogy might be empty chemical drums, for which routine disposal is clearly not acceptable and the administrative needs of a deposit-refund system would not require a new and more expensive disposal technology.

Of course, substitution will be feasible in other cases. For example, there are alternatives to chemical solvents as degreasing agents, alternatives that do not result in the generation of hazardous wastes.

Even in the strict fixed-proportions case, some capacity for substitution may be present. Beverage bottlers cannot change the one-to-one correspondence between filled and empty containers, but they can change the bottling technology in certain important ways. In states where there are mandatory bottle bills, for example, plastic containers have become the dominant form of packaging for soft drinks. This change may well have been motivated by many reasons besides the advent of bottle bills. However, plastic containers are lightweight and virtually unbreakable, and therefore less expensive to collect and handle once they are empty. Even if these laws had no effect on the proportion of containers littered (i.e., the noncompliance rate), the switch from glass to plastic would probably qualify as a reduction in residual external damages.[19]

In the beverage container example, eliminating the possibility of breakage constitutes both a private and public benefit. Substitutions motivated by the incentive instrument need not be salutary, however. In

[19]Container size has also increased, with 2-liter bottles becoming almost omnipresent. One purpose served by larger containers is to reduce the magnitude of the deposit relative to the price of the product. To the extent that bottle bills distort consumption decisions by charging a deposit in excess of the expected value of residual external damages from disposal, a switch to larger containers will reduce the inefficiency caused by this distortion. If the external residual damage from littering is a function of the number of containers littered but not their size, and size does not adversely affect the likelihood of redemption, then the substitution of larger containers constitutes a net social benefit. Whether it is also a <u>net</u> social benefit depends on how much consumers are disadvantaged because larger containers are less desirable than small ones.

a different context it seems just as plausible that firms might respond in ways that reduce the burden of the input tax or deposit but result in unexpected new environmental, public health, or occupational risks.[20]

Fixed-proportions relationships that vary across firms. Even if the fixed-proportions assumption is satisfied across all firms, it may not be the case that all firms share the same relationship. In such a case, an input tax will achieve efficient waste minimization incentives only if the firm's actual ratio of taxed input to waste happens to equal the ratio implicit in the tax rate. Unless there is a technological basis for this equality to exist, its occurrence should be viewed as an accident. Firms that initially use less of the taxed input per unit of waste generated than the ratio implied in the tax rate will experience too strong an incentive to minimize waste; for them, the tax will be too high. Conversely, firms that use more will face too weak an incentive because the tax will be too low. New social costs from excess control will arise in the former case; potential gains from waste minimization will go unrealized in the latter.

Variable-proportions. More commonly, the fixed-proportions assumption simply will fail altogether. There could be many ways to reduce the amount of hazardous waste generated that are not tied to a single offending factor of production. Simply cutting back on a particular input may not be as effective as housekeeping improvements, management reforms, or perhaps a comprehensive restructuring of the production pro-

[20] If risk rises with the number of possible exposure pathways, then recycling or pre-treatment may result in more risk than simple disposal.

cess. The cost-effective strategy may even involve changing the attributes of the final good being produced.

Unfortunately, targeting the tax on a specific input fails to account for these alternative waste minimization strategies. Instead, it would create new inefficiencies in production. Firms would be stimulated to adopt waste-minimization methods that reduced reliance upon the taxed input irrespective of whether these, decisions were cost-effective. Options that failed to reduce use of the taxed input would be discouraged. Like a technology-based standard that imposes an inefficient pollution control technology, the tax would cause an increase in the social marginal cost of waste minimization. The socially optimal level of waste minimization would inevitably decline to the extent that the input tax made waste minimization more expensive relative to continued disposal.

In the variable-proportions case, the choice of the target for the tax should be based on relative targeting inefficiency.[21] In general, targeting inefficiency can be minimized by selecting as the target the input for which demand is most inelastic. This keeps behavioral distortions to a minimum. If all input demands are relatively elastic, then output may be a better target than any of the inputs.[22]

How close to fixed-proportions is close enough? There is no simple rule to determine how closely an input-waste relationship must

[21] Concerns about administrative practicality are discussed later in the context of more general issues related to incentive targeting.

[22] There may be alternative technologies that can produce the same output without generating (as much) hazardous waste. In this case, output taxes would have to be based on a schedule that takes account of differences in production technology.

resemble fixed-proportions for a tax-subsidy instrument to perform in a satisfactory manner. The answer depends on at least three considerations. First, the underlying risks associated with the behaviors policy makers seek to change determine how important it is to intervene. The more risk implied by a firm's failure to comply with relevant disposal regulations, the larger will be the tolerable departure from strictly fixed-proportions. If the risks are small, however, then deviations from fixed-proportions may imply relatively large social *costs* from imperfect targeting.

The minimum resemblance to fixed-proportions also depends on the next-best regulatory strategy available. If it has serious incentive problems or suffers from difficult administrative hurdles, then a tax-subsidy instrument may look more attractive despite serious targeting inefficiency. It is worth remembering that under a tax-subsidy approach, the largest burdens are reserved for those firms that heretofore have failed to comply with other rules and regulations that are presumably in the public interest. Inefficiency associated with imperfect targeting of the input tax may be a justifiable sacrifice if it eliminates the competitive advantages associated with regulatory evasion and thereby removes a substantial number of bad actors from the market.

Finally, deviations from fixed-proportions will be more tolerable to the extent that better targets are not available. What we mean by "better targets" in this context is other inputs (or perhaps outputs) that are not closely related to hazardous waste generation, but for which demands are relatively inelastic. Taxes levied on such unrelated factors may cause fewer distortions in production and consumption decisions.

It is worth remembering that in the absence of concerns about non-compliance, a simple tax on hazardous waste generation is the preferred regulatory instrument. The objective of any second-best incentive strategy should be to approximate such a tax as closely as possible.

Tax Targeting

A range of problems might arise with respect to the input tax used to fund the safe-disposal subsidies. When any of these problems occurs, the tax will create inefficiencies in production.

One possibility is that the fixed-proportions relationship described earlier does indeed exist, but it is undesirable to tax the correct input. This situation might occur, for example, if the tax would create serious new economic distortions. Taxes on petrochemical feedstocks in addition to what is already mandated under CERCLA/SARA might fit this description. Neither the magnitude nor the distribution of Superfund tax burdens has any relationship to past hazardous waste disposal problems; after all, no input tax levied today can discriminate across firms based on yesterday's disposal practices. Neither can an input tax create incentives for proper disposal of tomorrow's wastes. Thus, these taxes cannot internalize the residual external damage from disposal, whether or not it occurs in accordance with RCRA rules. Instead, they distort production decisions by failing to discriminate across disposal methods. Having already established incentive-distorting taxes to finance Superfund, any additional levies would result in still greater distortions -- even if in the absence of Superfund feedstock taxes they could be justified on efficiency grounds.

Another possibility is that the fixed-proportions relationship exists, but not across all users of the taxed input. For example, methane is a common input in the production of certain plastics, and is therefore related (although not necessarily by fixed-proportions) to the generation of hazardous waste. However, methane is also used as a fuel. If it is determined that methane used to produce plastics should be subjected to an input tax, then methane used for fuel should be exempt. Distinguishing exempt applications may be relatively straightforward in this case, but for other materials it may be much more difficult. As the number of possible exceptions rises, it may become increasingly difficult to deny variances in marginal cases, and firms would clearly have incentives to seek exemptions whenever possible. Taxing all uses of an input offers the advantage of administrative simplicity while sacrificing efficient incentives in production. But making the effort to avoid these inefficiencies can quickly create an administrative nightmare.

If the fixed-proportions relationship fails, then efficient targeting of the tax simply will not be possible. The theoretically optimum tax rate would have to vary dynamically across inputs, production volumes, firms and industries -- an impossible administrative task. Taxing any particular input would cause firms to substitute away from it without necessarily reducing hazardous waste generation.[23] In general, the less elastic the demand for the input, the smaller will be the distortion caused by a tax. Thus, the amount of inefficiency created can be controlled by careful selection of the target.

[23] In perverse cases, firms may be able to actually increase the amount of waste they generate, a strategy that becomes increasingly attractive with the size of the subsidy.

Output taxes may be a worthwhile option to consider if suitable candidates cannot be found among the inputs. Prices in factor markets would be left alone, so no distortions in production decisions would be created. Of course, output taxes could cause inefficiencies in consumption. In general, however, the presence of a negative environmental externality implies that final goods prices are below the full social cost of production. Thus, an output tax may restore more appropriate relative prices in final goods markets, a result that enhances efficiency. However, if these taxes over-correct for the residual external damage from hazardous wastes, then they will drive final goods prices too high and thereby create inefficiencies in consumption.[24]

Subsidy Targeting

In general, targeting a subsidy on safe disposal should be a relatively straightforward task. Since the EPA already has regulations in place that prescribe appropriate disposal alternatives, eligibility for the subsidy would presumably arise for disposal at any approved destination.[25]

---

[24]Output taxes would offer no incentive for waste minimization beyond what is contained in the output effect. As we indicated in Chapter 5, this is likely to be quite small except in unusual circumstances.

[25]Recycling poses a potential difficulty. Whether the subsidy should be offered for wastes destined for recycling depends on, among other things, whether the recycled material is (or can be) subjected to the input tax along with its virgin equivalent. If the recycled material is taxed, then subsidies must be offered for waste destined for recycling to maintain constant relative prices between primary and secondary markets. Subsidizing recycling without taxing recyclers' output creates a relative price advantage for recycled materials that is unlikely to be justified based on relative risks, and may result in more waste generation. Taxing recyclers' output without subsidizing the waste they purchase creates the reverse asymmetry, in which too little waste is recycled.

Failing to treat functionally identical disposal alternatives in a similar fashion creates the potential for serious targeting inefficiencies. An excellent example of this problem is the case of deposit-refund systems for beverage containers, which are intended to reduce littering.[26] To receive refunds, consumers must return their empty containers to a retailer or reclamation center. Because of its obvious impracticality, refunds cannot be provided for containers disposed with domestic refuse. This means that arguably equivalent disposal alternatives are not treated equivalently. Thus, what we might call "compliance disposal" consists of a new disposal technology that is unambiguously more costly than its predecessor. "Ordinary noncompliance" (i.e., disposal with domestic refuse) entails virtually identical social costs, but it is treated no differently than "black market disposal" (i.e., littering).

This asymmetry in refund targeting is inefficient. Many (perhaps most) containers used to be disposed of properly, but the availability of the refund causes them to be diverted to a new disposal path. The

[26] An often-claimed secondary purpose is to reduce solid waste. At least with respect to aluminum cans, deposit-refund systems do indeed reduce the amount of solid waste because the demand for empty cans apparently is perfectly elastic. However, markets for glass cullet and plastic scrap are limited. As more states have established similar systems, the supply of these materials has grown enormously, thereby depressing prices and making disposal the cost-effective destination. Thus, in our discussion concerning beverage containers and targeting, we assume that litter reduction is the dominant benefit to be achieved through a deposit-refund system.

social benefit obtained by this diversion is trivial at best, but substantial social costs are borne in the process.[27]

A similar phenomenon may arise if hazardous waste regulators establish overly restrictive eligibility standards for a safe-disposal subsidy. For example, regulators might become particularly enamored with innovative disposal methods that employ only the most advanced technologies. Should regulators make only these innovative alternatives eligible, however, costly inefficiencies in waste disposal decisions will result. Firms would send wastes to expensive "high-tech" disposal facilities even if other (unsubsidized) alternatives were cost-effective. The direction and pace of technological change also would be altered, as firms redirect their research and development investments towards similarly exotic ventures.

In theory, every disposal alternative that is less risky than illegal dumping should be eligible for the subsidy, because "safe disposal" is a relative rather than absolute concept. Subsidy rates would be scaled inversely with residual risk. In practice, of course, this is both administratively infeasible and politically untenable. It would require, for example, providing subsidies for ordinary noncompliance as long as it entailed less risk than illegal dumping -- an unimaginable policy. Considering only legal alternatives, it may be administratively impossible just to design and implement a system that offers different subsidies for each of them in accordance with relative risks.

[27]Presumably, there is a benefit in the form of reduced solid waste disposal costs. However, this benefit exists only to the extent that containers reclaimed through the system are cost-effectively recycled rather than disposed.

Any deviation from comprehensive targeting creates inefficiency. In practice, therefore, some targeting inefficiency is inevitable. Never-the less overly restrictive eligibility rules that grant preferential status to emerging technologies over traditional disposal methods could make this inefficiency far greater than it has to be.

Qualitative Conditions for Efficiency and Effectiveness of a Tax-Subsidy Instrument

The efficiency of a tax-subsidy instrument will be largely determined by the degree of complementarity between the taxed input and waste generation. The more complementary they are, the better the input will perform as a proxy for waste minimization. Reductions in the use of the taxed input will then translate into lower levels of waste generation.

The efficacy of the instrument depends on whether waste minimization or safe disposal is our primary objective. If waste minimization dominates, then all that is required is a simple input tax. Firms will respond to the tax depending on the share of total costs attributable to hazardous waste disposal, the own-price elasticities of demand for their output, and the degree to which there are opportunities for substituting away from the taxed input in ways that do not result in more waste.[28]

The tax-subsidy approach makes sense only if safe disposal is relatively more important than waste minimization. In this regard, the efficacy of the instrument will depend on how large the subsidy is relative to the difference between compliance and noncompliance disposal costs. No subsidy will be large enough to capture all noncompliers, of

[28] See Chapter 5 for more discussion of the relative important of these three factors.

course, but a very large subsidy may not be necessary if a substantial number of firms are currently close to their switch points. Deriving the socially optimal subsidy requires taking account of the distribution of firms and the reductions in compliance disposal cost that are necessary to induce them to switch. This task can be very difficult in practice; nevertheless, the optimal direction for change is unambiguous because any safe-disposal subsidy improves upon no subsidy at all.

Chapter 7:

AN APPLICATION OF COMBINED TAX-SUBSIDY INSTRUMENTS: A DEPOSIT-REFUND SYSTEM APPLIED TO USED OIL

"Used oil" is a high-volume waste, with over one billion gallons generated each year by millions of diverse entities, ranging from individuals who change the motor oil in their own cars and trucks to large industrial firms. The EPA defines used oil to include oils derived from petroleum- and synthetic-base fluids that are used as lubricants, hydraulic fluids, metal-working coolants and insulating fluids, which become contaminated through use or subsequent mismanagement.[1] Congress has instructed EPA through the Resource Conservation and Recovery Act of 1976 (RCRA), the Used Oil Recycling Act of 1980 (UORA), and the Hazardous and Solid Waste Amendments of 1984 (HSWA), to encourage the recycling of these wastes and to ensure that they are disposed of in such a manner that adequately protects human health and the environment.

Concerns about the possible health risks from used oil have been around for many years. These concerns arise for two reasons. First, lubricating oil may become contaminated in use by potentially toxic or carcinogenic heavy metals such as lead, cadmium, and arsenic. Second, and more importantly, used oil is known to have been used as a vehicle for the illicit disposal of chlorinated solvents and other hazardous

[1] ***See*** 50 FR 49261 (Nov. 29, 1985). Oil that is spilled or leaked prior to use is considered "waste oil."

wastes. By better controlling the disposition of used oil, these hazards presumably can be **reduced.**[2]

In this chapter we analyze whether a deposit-refund system applied to lubricating oil might be an attractive regulatory instrument for efficiently reducing risks to human health and the environment. The analysis is preceded by descriptions of existing markets for used oil and the methodology to be employed.

An Overview of Used Oil Markets

The most recent estimate available suggests that about 1.3 billion gallons of used oil were generated in 1985. This is believed to represent about half of all new oil purchased, the remainder having leaked or carbonized in use. As Figure 7-1 illustrates, three-fifths of this waste stream came from automotive sources, and do-it-yourself oil changers (DIYers) account for about one-fourth of the automotive share. About 60 percent of the total amount of automotive oils purchased are believed to be generated as used oil. However, generation rates vary considerably among automotive sources; for example, 73 percent of motor oils become waste but only 10 percent of hydraulic and transmission **fluids.**[3]

---

[2]**For** a classic treatment in the expose genre, see, e.g., Epstein, Brown and Pope (1982: ch. 6). Representative policy documents that discuss the potential risks of used oil include New York State Legislative Commission (1986); Carnegie Mellon (1988); and New England Waste Management Officials' Association (1988). The EPA's "burning and blending" regulations are substantially based on concerns about adulteration. See 50 FR 49164 (Nov. 29, 1985).

[3]**Volume** data are from Temple, Barker and Sloane (1987b). Generation rates come from Franklin (1983: Table 1).

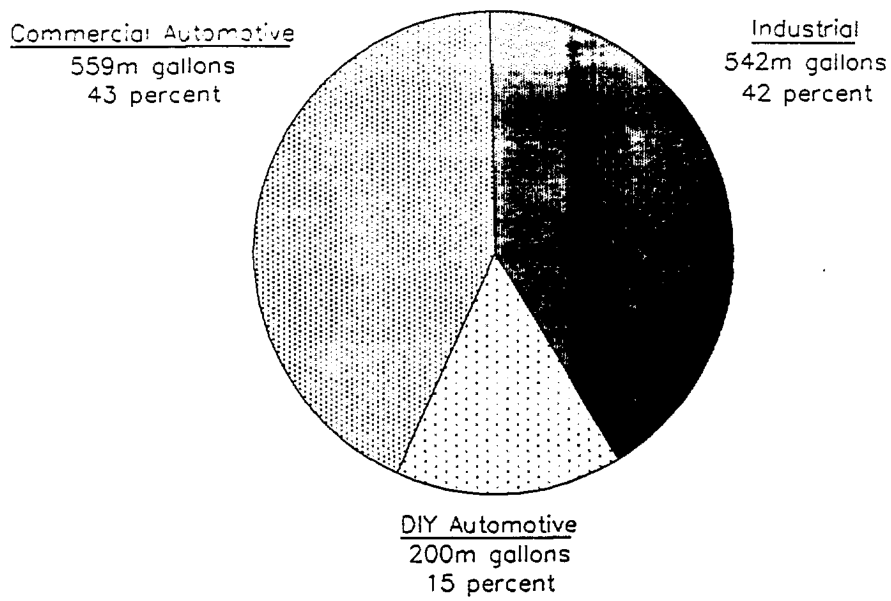


Figure 7-1. Used Oil Generation by Sector, 1985

The remaining two-fifths of all used oil generated in 1985 came from a variety of industrial settings. Industrial generation rates average 34 percent, but vary considerably across applications. For example, only 10 percent of process oils become waste, but 90 percent of electrical transformer insulation oils become waste when equipment is disposed -- oils that often contain high concentrations of polychlorinated biphenyls (PCBs).

Slightly more than half of all used oil generated was estimated to have been reclaimed for delivery to the secondary **market.**[4] The remainder was recycled in-house, used to oil roads as a means of dust suppression, burned, or disposed in various ways. About one-third of all used oil generated was disposed or dumped.[5]

Figure 7-2 shows that the ultimate disposition af used oil depends primarily upon the sector from which it came. Most oil generated in either the industrial or commercial automotive sectors enters the secondary market, and a substantial fraction of what remains is used by generators for functionally similar purposes. For example, roughly 60 percent of used oil from commercial automotive generators and 70 percent from industrial sources enters the secondary market. Most of this oil is burned as a fuel supplement, with small percentages going to re-refining, road oiling, disposal, and non-fuel industrial uses. Roughly half of the remainder is used for functionally similar purposes outside of the secondary market; hence, it is included within the "secondary market" category in Figure 7.-2. In contrast, very little used oil from

---

[4]**The** EPA and many others regularly refer to the secondary market as the "used oil management system," or UOMS. We prefer to call it a secondary market because it is aptly characterized by a systemic lack of management.

[5]**The** aggregate estimate given is 1,248 million gallons, but this excludes 46 million gallons generated by industrial firms and reused in-house; when added, this gives a total of 1,294 million. (Temple, Barker and Sloane 1987b: Table I). Estimates broken down by sector, however, sum to 1,311 million (Temple, Barker and Sloane 1987b: Table III). The 17 million gallon discrepancy appears to coincide with a reduction in the estimated amount of DIY oil returned to service stations (Temple, Barker and Sloane 1987b: 4). This amount either vanished from the aggregate estimate or was double-counted in the sectoral breakdown. We have used the sectoral breakdown.

DIYers is recovered for secondary market use, and what remains is mostly disposed or dumped.

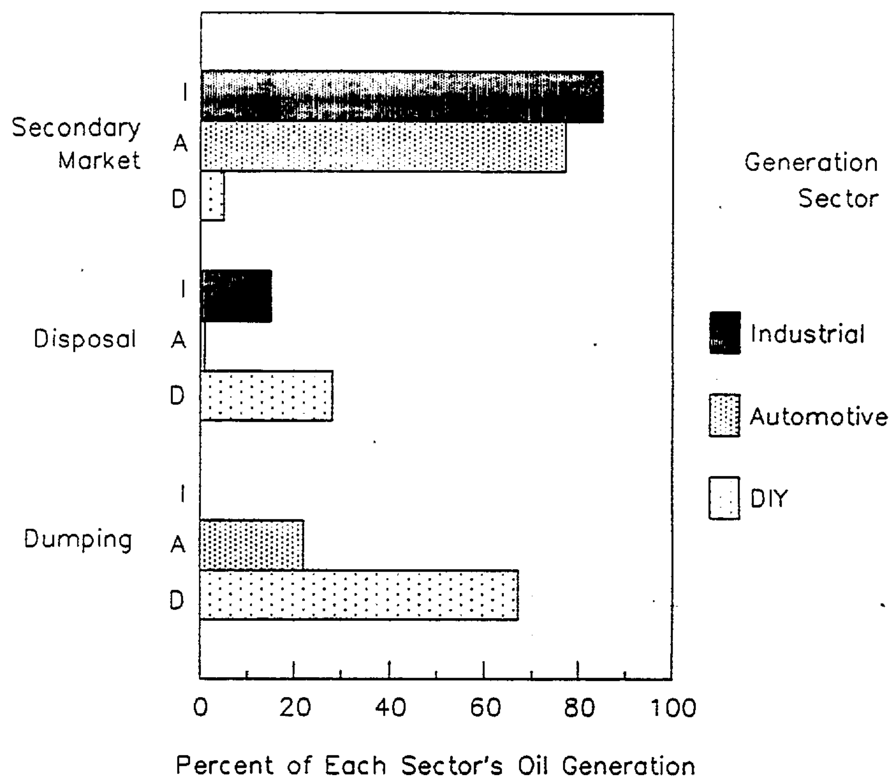


Figure 7-2. Estimated Used Oil Disposition by Sector, 1985

Estimated secondary market reclamation rates for the industrial and automotive sectors have been stable through repeated estimation efforts. In contrast, estimates for DIYers are all over the map. A study performed in 1981 by a consultant for the EPA relied upon an independent market research firm's estimate that DIYers generated 350 million gallons per year (Sobotka 1981: I-2). In 1983, a different EPA consultant arrived at a virtually identical figure (Franklin 1983). However, the following year this firm revised its estimate downward to 239 million gallons, a 32 percent decline (Franklin 1984). The contractor that produced the EPA's 1985 RIA relied extensively upon Franklin's estimates. DIY generation in 1983, however, was estimated at 194 million gallons, an additional 19 percent decline from the earlier estimate (Temple, Barker and Sloane 1987b). In a recent revision, this firm offered an estimate of 200 million gallons for 1985 (Temple, Barker and Sloane 1987a: 1987b). But another market research study aimed at analyzing the potential for "quick-lube" establishments estimates the volume of lube oil sold to DIYers in 1984 at roughly 467 million gallons (Kline 1985). If 73 percent of this oil was generated as waste (the generation rate derived by Franklin and used by Temple, Barker and Sloane), then DIY generation would have amounted to 341 million gallons -- a figure 70 percent larger than Temple, Barker and Sloane's most recent estimate.

Estimates for the proportion of DIY oil recycled also vary. In the 1983 Franklin study, the assumed recycling rate was 8 percent, but an 11 percent figure was used in Franklin's 1984 follow-up report (Franklin 1983: 1984). The 1981 market research study, which was specifically intended to estimate the amount of used oil that could be recovered from DIYers, concluded that about 14 percent was then being recycled. This figure was used in the RIA and in at least one subsequent revision by the contractor that produced it (Temple, Barker and Sloane 1985a; Temple, Barker and Sloane 1987a). Inexplicably, just four days later the same contractor published another memorandum in which the DIY recycling rate for 1985 was cut to just 5 percent (Temple, Barker and Sloane 1987b). According to the consultant, "widespread anecdotal evidence suggests that a large fraction of service stations no longer accept DIY oil," a phenomenon attributed to the exogenous fall in virgin oil prices that occurred late in the year and continued in 1986.

Thus, it is difficult to make appropriate point estimates for either the amount of used oil generated by DIYers or the amount that they recycle. The estimates differ by nearly a factor of two because of differences in data sources and estimation methods, not year-to-year fluctuations in the amount of lubricating oil sold or generated as used oil. Variations matter because whatever DIYers *do not* return to service stations and recycling centers is disposed, dumped, or otherwise used in environmentally suspect ways. Furthermore, one of the EPA's continuing objectives is to seek ways of getting more DIY oil into the secondary market (Versar 1986: Temple, Barker and Sloane 1987a). Therefore, to maintain consistency with the data used for the other generation sectors, we have chosen the 200 million gallon estimate and the 5 percent reclamation rate.[6]

Used oil gets from generator to end-user in a variety of ways. It is collected from generators by independent collectors, vertically integrated used oil processors, and "gypsies" -- independents that sell directly to virgin fuel oil dealers (VF-ODs) without intermediate processing. There is some evidence that the number of "gypsies" has

[6] If the estimates made by Kline (1985) are correct, then our estimated net benefits from the deposit-refund system will be too low.

mushroomed since the promulgation of the EPA's burning and blending regulations, and the pejorative characterization of these outfits reflects the fact that the EPA and others consider them illegitimate actors in the used oil business (Temple, Barker and Sloane 1987b: 5). In 1983, approximately 700 collectors were believed to exist (Temple, Barker and Sloane 1985a: Table IV-1 since then, continued horizontal and vertical integration has apparently reduced their number. No reliable estimate of the number of "gypsy" collectors is available, but self-described "legitimate" collectors consider them to be a widespread phenomenon (Temple, Barker and Sloane 1987b).

Once collected, used oil generally is sold to an intermediate processor or directly to a lube oil re-refiner. "Processing" usually involves minimal intervention; water is evaporated and solids are allowed to settle. Processors sell this "processed" used oil for use as boiler feedstock, certain non-fuel applications, road oiling, and blending with virgin fuel oil. Tank bottoms are sold to asphalt plants. In 1983, there were an estimated 240 intermediate processors (Temple, Barker and Sloane 1985a: Table III-4). As of 1985, increasing concentration appears to have reduced the number of processor firms but not the amount of oil they handle (Temple, Barker and Sloane 1987b: 6). More recent data concerning industry structure apparently are unavailable.

Historical Attempts to Regulate Used Oil

Based on its authority under RCRA, the EPA first proposed to list certain waste oils as hazardous in 1979 (43 FR 58946, Dec. 18, 1979), but apparently this proposal was quietly abandoned. The EPA was first specifically empowered to regulate used oil in 1980 through the UORA; the HSWA broadened the Agency's responsibilities four years later.[7] The legislative history of these initiatives displays rising Congressional concerns about the risks posed by used oil, and increasing frustration with the EPA's lack of action (Harris, Want and Ward 1987: 171-177).

In the findings which form the foundation for the Used Oil Recycling Act of 1980 (UORA), Congress stated that:

> 1) used oil is a valuable source of increasingly scarce energy and materials;
>
> 2) technology exists to re-refine, reprocess, reclaim, and otherwise recycle used oil;
>
> 3) used oil constitutes a threat to public health and the environment when reused or disposed of improperly; and that, therefore, it is in the national interest to recycle used oil in a manner which does not constitute a threat to public health and the environment and which conserves energy and materials.[8]

Unlike Congressional directives with respect to other putative hazardous wastes in which generation was supposed to be curtailed or eliminated if possible, the Congress has consistently emphasized the need for recycling used oil. The HSWA directed the EPA to commence regulating used oil within a year after the statute was enacted, but also instructed EPA to ensure that "regulations do not discourage [its] recovery or recy-

[7] RCRA Sec. 3012, Public Law No. 96-463, 94 Stat. 2055 (October 15, 1980), redesignated as RCRA Sec. 3014 as amended by Sec. 502 of HSWA, Public Law No. 98-616, 98 Stat. 3277 (November 8, 1984).

[8] Used Oil Recycling Act, Sec. 1; amended as RCRA Sec. 3012 (Pub. L. No. 96-463, 94 Stat. 2055), redesignated as RCRA Sec. 3014 by HSWA Sec. 502.

cling..."[9] Thus, used oil may be considered a hazardous waste, but it enjoys a special position in the regulatory system: the creation and maintenance of viable recycling markets is not merely sanctioned by Congress, it is required.

Proposed Management Standards

In 1985, the EPA proposed an extensive set of management standards governing the collection, storage, processing, and disposition of used oil that entered the secondary market. These standards were intended to alter the pattern of end-uses. Certain disposal practices would have been prohibited, burning sharply restricted, and the amount of used oil destined for re-refining dramatically increased. Indeed, an emphasis on the perceived environmental and conservation benefits from re-refining dominates the EPA's proposal. According to the RIA, the regulations would cost about $1.3 million for every case of cancer avoided (Temple, Barker and Sloane 1985a: Table I-6).

Several aspects of the RIA are highly suspect, however. Among other things, it assumes full compliance with the proposed standards and end-use restrictions without any need for regulatory enforcement.[10] Thus, the estimated reduction in cancer incidence due to the regulations

[9] **RCRA** Sec. 3014(a). Any regulations proposed must also be accompanied by an analysis of the effects of such regulations upon the used oil recycling industry. In effect, the RIA thus has a statutory basis as well as an administrative one by virtue of Executive Order 12291.

[10] **See** Temple, Barker and Sloane (1985a: Table V-18), which purports to measure social costs but does not take account of any expenditures for regulatory enforcement or its evasion.

represents at best the upper-bound of what the rules could possibly achieve in practice.

Another dubious assumption is the virtual absence of a supply response. The RIA estimates that the regulations would cause a decline of $.02 per gallon in the value of used oil at the generator level on a base of $.21 per gallon -- a 9.5 percent decrease.[11] But the quantity of used oil delivered to the secondary market was projected to decline by only 0.15 percent (Temple, Barker and Sloane 1985a: Table V-19). Thus, the analysis implicitly assumes that the supply elasticity for used oil generators is less than 0.02 (0.15/9.5 = 0.016). By assuming that the amount of used oil reclaimed is virtually immune to changes in its price, the analysis effectively avoids any need to deal with the unpleasant possibility that the standards might deter recycling and thus violate RCRA Sec. 3014(a).

This scenario seems contrived. For authorized end-users, the proposed management standards would have increased the cost of used oil as a feedstock, quite possibly by much larger amounts than those forecast in the RIA. Because used oil would be less attractive as a substitute for virgin petroleum products, these firms would have bid prices downward accordingly. Intermediate processors would have had to cut the prices they paid collectors, who in turn would have reduced how much they offered to pay to generators. Lower prices would have resulted in less used oil reclaimed and a corresponding increase in the amount dis-

---

[11] Temple, Barker and Sloane (1985: Table V-21). All prices were estimated prior to the drop in virgin oil prices that occurred in 1986. More recent anecdotal evidence suggests that prices have fallen into the negative range; i.e., generators now must pay to have used oil collected rather than *be paid* by the collectors.

posed, dumped, or used in some environmentally inappropriate manner. Thus, any realized reduction in end-use risk would have been achieved at the cost of diverting more used oil away from the secondary market and possibly to environmentally more damaging places. A less extreme assumption for the supply elasticity -- say, 0.5 -- would imply a reduction in the quantity of used oil reclaimed by more than 30 million gallons instead of the mere 1 million gallon decline predicted.

Presumably, this is 30 million more gallons that would be disposed, dumped, or used in an environmentally suspect manner. It is conceivable that the gains from improved management of the oil still reclaimed would outweigh the losses resulting from more oil being directed elsewhere. But it seems equally (if not more) plausible that the new social costs from increased dumping could exceed these benefits. Apart from questions concerning the *economic* cost of the proposed standards, it is not obvious that their promulgation would have offered net *environmental* benefits.

This closely follows the pattern described in Chapter 4: Tightening regulatory standards on the disposal of hazardous waste (cf. the end-uses of used oil) leads some generators to abandon the regulated waste management system (cf. the secondary used oil market). Those generators that were not in the system before (cf., especially, DIYers) now have even less of an incentive to dispose properly. Whether the gains from tighter controls exceed the losses from increased noncompliance is a crucial question that, unfortunately, the existing literature has failed to examine.

Evidence of the EPA's Low Regard for Economic Incentive Instruments

When it first proposed the burning and blending regulations that were ultimately promulgated in November 1985, the EPA considered a tax-rebate system as an alternative to a traditional standards-based regime.[12] As sketched by the Agency's consultant, this system would have levied a tax on the manufacture of new lube oil and offered an income tax rebate to preferred end-users for every gallon of used oil they purchased. Revenues from the tax were intended to match foregone income taxes. The EPA rejected this approach, labeling it ineffective at protecting human health and the environment and impractical to administer. Unfortunately, the basis for this rejection is hard to fathom. While the specific proposal and the analysis upon which it was based both have their flaws, the factors that were cited to discredit the proposal were largely irrelevant, presented in a misleading manner, or simply the product of faulty analysis.

The EPA concluded that the tax-rebate would be ineffective because, among other things, it failed to "ensure that no used oil [went] to unacceptable users." Thus, the EPA held the tax-rebate instrument to a standard of efficacy that no regulatory instrument could possibly achieve. This strongly suggests that the tax-rebate was rejected for unstated political or organizational reasons, and that a dispassionate analysis was never conducted.

In addition, the EPA suggested that used oil marketers might subvert the tax-rebate system by discounting used oil fuel by an amount

[12] __See__ 50 __FR__ 1684 (Jan. 11, 1985), and Sobotka (1981).

equal to the rebate and continue to sell to unacceptable users. This claim is either the product of faulty economic analysis or based on a transparently erroneous assumption concerning how the rebate would be targeted. As long as the rebate is paid only to approved end-users, used oil marketers are passive participants. If they sell to the highest bidders, acceptable end-users will buy more, thereby crowding out at least some unacceptable end-users. The only way fuel marketers could possibly subvert the system is if *they* rather than approved end-users received the rebate.

Furthermore, the incentive approach was criticized for its inability to prevent the adulteration of used oil with chlorinated solvents and other hazardous wastes. The EPA apparently assumed that its proposed standards would prevent this practice, an assumption clearly retained in the preamble defending the final rule. However, the incentive to adulterate used oil with hazardous wastes exists independent of the regulatory strategy employed. It results from the high cost of RCRA disposal, not whether environmentally safe used oil reclamation is achieved through command-and-control methods or economic incentives. An incentive system probably has an advantage inasmuch as the EPA could, if it chose, restrict eligibility for the rebate to those end-users that are either equipped to safely handle adulterated used oil or at least detect its presence and refuse it.

The answer to the adulteration problem lies in which regulatory strategy offers the more beneficial effects at the margin. A standards-based approach with voluntary compliance and minimal enforcement (such as the approach preferred and ultimately adopted by the EPA) can be expected to have little or no effect on adulteration. Extensive enforcement might discourage it, but it is unclear where the adulterants would go if not into used oil. An economic incentive system, however, enjoys at least the capacity to reduce the risks posed by adulteration by diverting whatever used oil gets adulterated to environmentally more appropriate end-uses.

Administrative problems cited by the EPA appear more suggestive of a desire to protect the agency's turf rather than the environment. A tax-rebate system was determined to be "impractical" because some acceptable end-users do not pay taxes; the Congress has to set and periodically adjust tax and rebate rates; and the Internal Revenue Service (IRS) must be relied upon for administration and enforcement. By designing the system to pay cash, however, taxpayer status would become irrelevant. Not-for-profit entities such as hospitals and schools would probably respond even if payments were made only on an annual basis. Regular Congressional involvement may be perceived as a nuisance, but it does enhance public accountability; moreover, the Congress might delegate the rate-setting responsibility to the EPA so long as the agency kept within specified ranges. Finally, the involvement of the IRS probably constitutes an asset; its reputation for tough enforcement would accentuate rather than detract from the instrument's effectiveness. In short, the EPA's administrative problem with a tax-rebate system is that it might lose authority and control, not that end-users would fail to respond to financial inducements or that the IRS would do a bad job.

The EPA also claimed that a tax-rebate system could cost more to administer than a traditional standards-based regulatory program. This seems peculiar given the Agency's historically limited interest in cost control and the likelihood that it is the IRS that would bear much of the burden. It is worth noting that in the preamble defending its promulgation of these rules, concerns about administrative cost appear to have vanished.

The EPA's burning and blending regulations were promulgated in November 1985. However, the management standards and end-use controls that were proposed at the same time were indefinitely postponed. Currently, the agency is debating between a standards-based approach that would feature a mandatory retailer take-back provision and a marketable permit system. Within the EPA, each alternative has its advocates and opponents; however, we are not persuaded that either approach is fundamentally sound. If effective, the mandatory take-back scheme promises huge distortions in retail markets for lubricating oil and enormous administrative costs. To implement the marketable permit system under existing statutory authority, its advocates anticipate having to rely upon Section 8 of the Toxic Substances Control Act (TSCA) -- a strategy that invites administrative difficulties and internal turf battles.

In this case study we focus on a variant of the rejected tax-rebate instrument -- a deposit-refund system. We have chosen not to analyze a permit-style equivalent system because the risk characteristics of used oil do not suggest any basis for preferring it to a price-based regime.[13] Moreover, deposit-refund systems have been used in other con-

[13] The presence of nonlinearities in marginal damage (e.g., threshold effects) is probably the most important basis for preferring a quantity-based approach. The EPA has not alleged the existence of such nonlinearities.

texts, and where they have been used they appear to enjoy considerable public support.

Regulators inevitably have to design policies based on data that are limited both in scope and quality. Used oil clearly fits into this category. In this case study we offer a preliminary assessment of the merits of a deposit-refund system. First we develop a conceptual model that requires the specification of a small number of parameters that capture the essence of the problem. The model is then applied to the case of used oil, relying upon both the available data and informed judgment. Doubts concerning the validity of the underlying data can be resolved by substituting different values for the parameters and analyzing the results obtained.

A Conceptual Model

So far we have limited our discussion and analysis to representative firms that may or may not comply with disposal regulations. To properly evaluate a potential deposit-refund system that might be devised for a specific hazardous waste stream, we must extend the analysis to the market level.

In addition, the issues we have dealt with in earlier chapters involved another tradeoff -- that between generating and preventing the generation of hazardous wastes. The waste-minimization issue is a minor one with respect to used oil; barring revolutionary changes in engine technology that dramatically reduce friction, used oil is here to stay. The only relevant issues are (1) how much of the oil purchased for use can (and should) be recovered as waste, and (2) what is to be done with it.

Generators of hazardous waste (or used oil) differ across untold dimensions. The one that matters most for our purposes is the gap between compliance and noncompliance disposal costs. In terms of the model presented in Chapters 4 through 6, this gap equals the unit cost of satisfying regulatory requirements (C_R) less the cost of noncompliance (C_O). The gap will vary across firms as the result of variations in both C_R and C_O. For those firms that initially comply with disposal regulations, $C_R - C_O < 0$; the full cost of noncompliance is greater than the burden of satisfying regulatory requirements. For a small number of firms C_O may be very high, perhaps because of extraordinarily large intangible costs associated with being caught out of compliance. These are the firms that will comply with disposal regulations at virtually any cost. A larger number of firms will have noncompliance costs that are not as severe, but still high enough to dominate. These firms will comply unless the cost of compliance is driven way up.

For noncompliers, however, the cost of compliance dominates and $C_R - C_O > 0$. Some of these firms will have noncompliance costs that approach zero; the likelihood of detection and punishment may be trivial, and they may not own intangible assets (such as community goodwill) that would be lost if they were caught. Presumably, the majority of noncompliers will not be extreme cases, but instead firms with noncompliance costs that, although not trivial, still dominate.

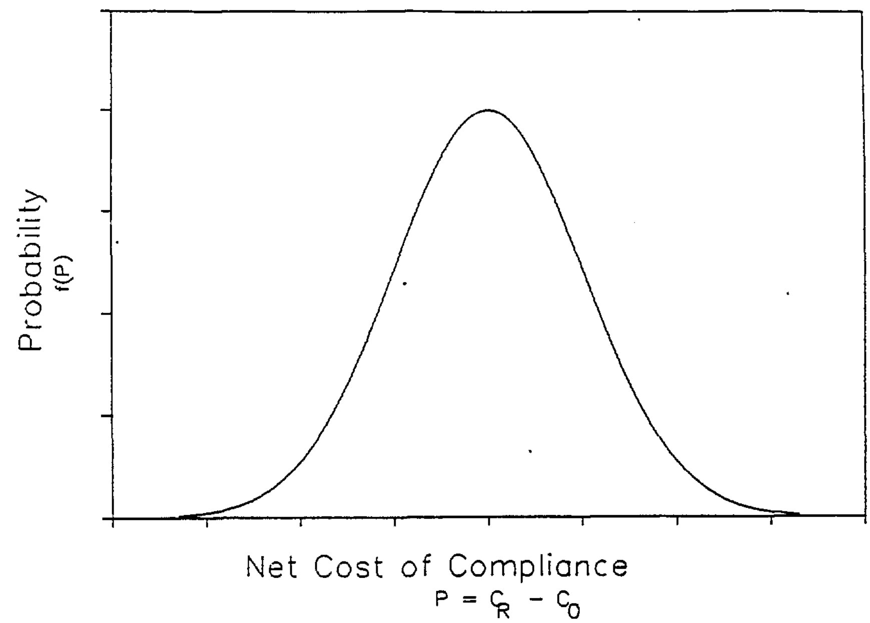


Figure 7-3. Distribution of the Net Cost of Compliance Across Firms

Figure 7-3 illustrates this relationship as a normal probability distribution, with the horizontal dimension defined as the net cost of compliance, $P = C_R - C_O$. In general, compliers are found on the left side and noncompliers reside on the right. The dividing line between them depends on the location of the zero point; placing zero far to the right, for example, implies a low noncompliance rate because compliance is cheaper for most firms.

We can use this relationship to construct a formal probability model. If both the number of firms and the amount of waste are large, then a continuous distribution provides a sufficient approximation for the discrete case. Let the random variable p vary continuously within the interval [a, b]. Then the cumulative distribution of p up to a given realization P can be expressed as:

$$F(p) = pr\{a \le p \le P\} = \int_a^P f(p)\,dp, \tag{7-1}$$

where f(p) is the probability density function of the continuously distributed random variable.

Conceptually, the independent variable in Equation 7-1 is a price; the higher the price the larger the value of the function F(p). In the same vein, the dependent variable F(p) is an implicit quantity -- the proportion of wastes disposed of in the prescribed manner. By inverting the cumulative distribution F(p) we obtain this price-quantity relationship in the "backwards" fashion that is the convention in economics. Thus, the inverse function $F^{-1}(g)$ indicates the price that corresponds to any given quantity *generated, g.* It is equivalent to the market supply curve for appropriate waste disposal.

Figure 7-4 illustrates the effects of the refund component of a deposit-refund system, where the market supply curve is the inverse of the cumulative probability distribution, $F^{-1}(g)$. The vertical axis measures the "price" received by the generator for waste properly disposed; high prices received correspond to high net compliance costs in the probability model. In dollar terms, this "price" will be positive if the waste stream in question can be profitably recycled. For most hazardous wastes, however, the market "price" will be negative; the firm

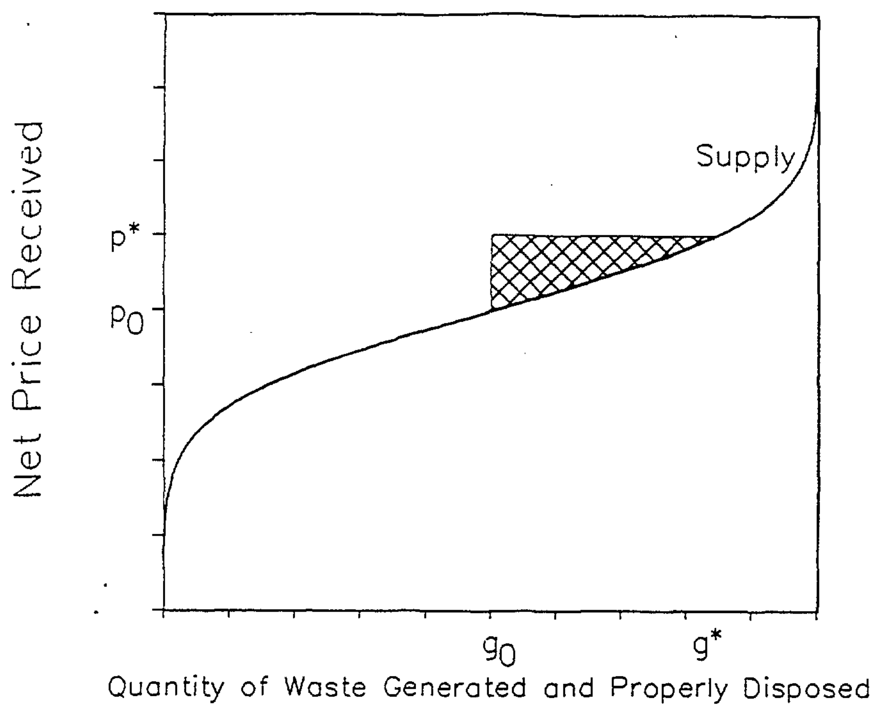


Figure 7-4. Effects of an Efficiency-Based Deposit-Refund System on Safe Waste Disposal

must pay (handsomely) for safe disposal. In the supply and demand framework illustrated in Figure 7-4, higher "prices" thus imply lower cash disposal costs.

The total amount of waste generated is G, g_0 of which is disposed of properly at the market price, p_0. This price fails to reflect the external residual damages from proper disposal, or perhaps more importantly, the damages that arise when a firm fails to engage in proper disposal. Let p* represent the optimal "second-best" price as described in Chapter 6. If waste generators received p*, they would "supply" to the the safe-disposal market the optimal "second-best" amount of the total waste stream, g*.

A refund equal to $(p^* - p_0)$ per unit of waste fully internalizes social cost under the circumstances given. The net benefit of the incentive consists of the reductions in residual damage obtained, less the additional costs borne to achieve them. This equals the cross-hatched area above the supply curve, the magnitude of which depends on the initial compliance rate and the level of residual external damage resulting from unsafe disposal. In Figure 7-4, the slope of the supply curve is almost flat in the relevant range; firms are relatively responsive to price changes, so the social benefit from additional compliance is relatively large. But if the initial compliance rate is either very high or very low (where the supply curve is steeply-sloped), then the optimal refund will have less effect on firms' choice of disposal method and thus achieve a smaller net benefit.[14]

The probability approach enables us to overcome two difficult problems, one conceptual and one empirical. The conceptual problem is that negative prices are a legitimate phenomenon arising any time a firm voluntarily *pays* for disposal. For some waste streams (e.g., used lubricating oil), positive and negative market prices may coexist simultaneously; a price of zero is relevant only insofar as it identifies the direction in which cash payments are made.

The empirical problem is that the available data are out-of-date and of uneven quality. In great measure due to changes in virgin oil prices and government policies, prices for used oil have fallen consid-

[14]As an aside (but a very important one), note that under an efficiency-based deposit-refund system waste generators pay the full social cost of hazardous waste *disposal irrespective of their choice of disposal method.* Therefore, firms will also engage in the socially optimal level of waste minimization without any additional intervention.

erably over the last several years. Quantity data have suffered even greater uncertainty. As difficult as it is to collect accurate data concerning legitimate market transactions, similar empirical work is highly impractical with respect to illicit activities.

By using a probability model we can estimate the effects of alternative policy interventions based on how they would alter behavior from some known (or at least widely accepted) baseline. This enables us to avoid relying upon price and quantity data that are outdated at best and possibly seriously flawed.

Applying the Model to Used Oil

The probability model allows us to derive market supply curves for each of the three used oil generation sectors. Data from the 1985 RIA are combined with informed intuition to produce baseline cases for analysis. The published risk assessment conducted pursuant to this RIA is used to develop estimates of the optimal deposit and refund rates. Combining these two elements enables us to estimate the net benefit from reclaiming the socially optimal quantity of used oil.

Several parameters must be specified to derive these estimates. First, we assume that the relative cost of compliance is distributed normally across units of used oil generated.[15] Because of the need to allow for negative prices and the absence of any theoretical basis for a

[15] If firms were identical, then this assumption implies that the relative cost of compliance is distributed normally across firms as well. For our purposes, only the amount of used oil reclaimed must be distributed normally. Given an estimated 650,000 used oil generators, we expect that this assumption is appropriate even if many generators are very large entities.

more exotic probability distribution, we do not test the effects of non-normal specifications.

Second, we hypothesize a specific standard deviation (σ) for each used oil generation sector to indicate relative degrees of dispersion; several alternative values for σ are tested for each sector.

Third, in our baseline scenario we assume that society places a value upon cancer prevention of $1 million per case. We test the sensitivity of the analysis with respect to values ranging up to $10 million per case.

Fourth, we assume in our baseline scenario that the deposit-refund system entails zero transactions costs. Because of their importance, we subsequently relax this assumption and consider a range of transactions costs ranging from two cents to two dollars per gallon reclaimed.

Beyond the relatively straightforward quantitative issues lies an additional area in which important questions must be addressed, questions that require considerable thought and careful planning in the design and implementation of a deposit-refund system. We conclude the chapter with a discussion of these issues.

Deriving Supply Functions

We assume that the amount of used oil reclaimed in each sector is normally distributed, the independent variable being the net price received by the generator. We take account of the estimated proportion of used oil that was reclaimed from each sector at the prices prevailing in 1983. Thus, the probability distribution selected for the industrial sector is constrained such that about 70 percent of the used oil generated in 1985 actually enters the secondary market. Similarly, 60 percent of used oil from the commercial automotive sector and 5 percent of DIY oil is reclaimed at the price prevailing within each sector. Our estimates depend only on relative price and quantity changes resulting from the refund and not on the actual levels of prices and quantities that prevailed at the outset. Thus, we need to select standard deviations for the underlying normal distributions, but do not need to constrain them with specific means.

As a first cut we assumed that the standard deviations for the three generation sectors would be the same. Differences in reclamation behavior would thus result only because of systematic variations in the average net prices received across sectors. This implicitly assumed that industrial and automotive generators would act just like DIYers if they faced the same net prices; conversely, if DIYers received net prices equivalent to what other generators earn, then their reclamation rate would be just as high.

What factors might cause net prices to differ systematically across sectors? Reclamation is probably relatively easy for industrial and automotive generators. Collectors know that these generators exist and will compete to service them. Moreover, they generate enough used oil such that economies of scale in collection and transportation can be exploited.[16] However, collectors are unlikely to serve DIYers except insofar as they deliver their waste to a larger volume collection point,

[16] Larger quantities also imply economies of scale in testing, which already may be necessary to limit potential CERCLA liability. If a deposit-refund system requires testing as a vehicle for eligibility certification, then large quantity generators will have lower unit transactions costs associated with the regulatory instrument.

such as a service station.[17] This may involve relatively large costs, especially if DIYers have to make a special trip.

In contrast, disposal and dumping seem likely to involve trivial costs for DIYers and many automotive service establishments. Apparently, the preferred practice among DIYers is to dispose of used oil with their domestic refuse. During slow periods, service station managers can send employees out to dump used motor oil without incurring additional labor costs. Dumping by neither the commercial- nor DIY-automotive generator seems likely to be detected or subjected to punishment. At least in relative terms, therefore, industrial generators probably face the greatest liability risks.

Given a benchmark for the cash price received by used oil generators and specified standard deviations, the price necessary to induce any given reclamation rate can be calculated using the areas under the unit normal. According to the RIA, prices received by generators in 1983 ranged from $0.00 to $0.40 per gallon; $0.21 was used as the benchmark for the analysis (Temple, Barker and Sloane 1985b: II-23). Prices corresponding to several reclamation rates are presented in Table 7-1 for a range of standard deviations. The benchmark price of $0.21 per gallon is assigned arbitrarily to a reclamation rate of 60 percent, which corresponds to the estimate for the automotive sector. Small standard deviations imply that generators respond within a narrow range

[17] Some states have mandated curbside collection of used motor oil (New England Waste Management Officials' Association 1988; Carnegie Mellon 1988). These programs are claimed to involve trivial costs, but it seems highly unlikely that such claims would withstand careful analysis. Curbside collection involves high marginal costs -- special containers, collection equipment different from the standard load packer, and low levels of public participation.

of prices; conversely, large standard deviations mean that reclamation behavior is relatively insensitive to price.

Table 7-1. Implicit Prices Required for Given Reclamation Rates, 1985

Implicit Prices, 1985 ($/gal)

Percent Reclaimed	Standard Deviation (σ) $0.25	$0.50	$1.00	$1.50	$2.00	$3.00	Ex Ante Rate for Sector
99%	$0.73	$1.25	$2.28	$3.32	$4.36	$6.43	
95	0.56	0.91	1.60	2.30	2.99	4.38	
90	0.47	0.72	1.24	1.75	2.27	3.29	
70	0.28	0.35	0.48	0.62	0.75	1.02	Ind
60	0.21	0.21	0.21	0.21	0.21	0.21	Auto
50	0.15	0.08	-0.04	-0.17	-0.30	-0.55	
25	-0.02	-0.25	-0.72	-1.18	-1.65	-2.57	
10	-0.17	-0.56	-1.32	-2.09	-2.86	-4.39	
5	-0.26	-0.74	-1.69	-2.64	-3.59	-5.48	DIY
1	-0.43	-1.08	-2.37	-3.66	-4.95	-7.53	

Notes:

1. Means for σ=1 derived from Lindley and Scott (1984: Tables 4 and 5).
2. Ex ante reclamation rates: 70% (industrial); 60% (commercial automotive); and 5% (DIY automotive).
3. Base price of $0.21/gal obtained from Temple, Barker and Sloane (1985b: 11-23) and assigned arbitrarily to the automotive sector.
4. Other prices calculated from areas under unit normal. Prices are calculated by subtracting the area corresponding to the baseline price (60%) from the area corresponding to the selected alternative reclamation rate (e.g., 5%, 70% 95%), multiplying by the chosen standard deviation, then adding the result to the baseline price ($0.21).

Consider the first column, in which the standard deviation, σ, equals $0.25. An increase in price of $0.07 per gallon (to $0.28) implies a 70 percent reclamation rate -- the same reclamation rate estimated for the industrial sector. A $0.47 per gallon decrease in price (to $-0.26) implies only a 5 percent reclamation rate -- the rate estimated for the DIY sector. Thus, if the benchmark price is reasonable and a constant standard deviation of $0.25 is applied across the three sectors, then these relative prices would have to be approximately correct. Compared with automotive generators, industrial generators would have received $0.07 per gallon more and DIY generators $0.47 per gallon less in 1983.

These price differences do not seem plausible because they imply a degree of price-responsiveness that is apparently counterfactual. Since the 1985 RIA was published, virgin oil prices have declined considerably. As expected, this has reduced the net price generators receive for used oil. Recent anecdotal evidence puts 1987 prices in the range of $-0.45 to $0.00 per gallon -- a decline of about $0.40 per gallon (Carnegie Mellon 1988: 83). But a decline of this magnitude equals 1.6 standard deviations if σ - $0.25. Reclamation rates for the industrial and automotive sectors would have to have declined to about 14 and 9 percent, respectively. The rate for DIYers would have plummeted to less than one-tenth of one percent. Total used oil reclamation would have fallen from 55 percent to just 10 percent.[18]

Such projections do not comport with the anecdotal evidence. Used oil reclamation has probably fallen off because of the decline in virgin oil prices, but industry insiders have not suggested such a large-scale

[18] These forecasts are calculated as follows. The standard normal deviates corresponding to 70, 60, and 5 percent are 0.5244, 0.2533, and -1.6449. From these amounts subtract 1.6 standard deviations ($0.40/$0.25). This results in standard normal deviates of -1.0756, -1.3467, and -3.2449, which imply the reclamation rates reported in the text.

collapse (Temple, Barker and Sloane 1987b). There has been widespread concern that this price decline has hampered DIYer reclamation; however, the perception is not that the few DIYers who used to return oil are now less willing to do so, but rather that fewer service stations are willing to accommodate them. Thus, it is the "demand" for DIYer oil that has fallen rather than the supply.

Table 7-2 indicates the reclamation rates for each sector implied by a $0.40 per gallon exogenous price decline. The smaller the standard deviation, the larger the decrease in reclamation is presumed to have occurred. Standard deviations as low as $0.50 seem implausible; even $1.00 may be too small.

Table 7-2. Implicit Reclamation Rates Subsequent to Exogenous Decline in Used Oil Prices, 1987

Implicit Reclamation Rates, 1987 (%)

Sector	Base Reclamation Rates, 1985	Standard Deviation (σ)					
		$0.25	$0.50	$1.00	$1.50	$2.00	$3.00
Ind.	70%	14%	39%	45%	60%	62%	65%
Auto.	60	9	29	44	49	52	55
DIY	5	0.1	0.6	2.3	2.8	3.2	3.8
Total	56	10	29	38	46	49	51

Notes:

1. Exogenous price decline assumed to be $0.40 across all sectors.
2. Aggregate reclamation rates calculated using relative proportions of used oil generated by each sector as weights: industrial (.4189); automotive (.4266); DIY (.1546).

For the constant-variance hypothesis to be plausible, at least one pair of corresponding columns from Tables 7-1 and 7-2 must make sense. Small values for σ are needed to keep prices received by industrial and automotive generators reasonably close, since no systematic price variation between these two sectors has been documented. But large values are necessary to achieve the relatively moderate responses to falling prices that apparently have occurred.

Moreover, high standard deviations seem especially likely for the DIY sector. Those who bother to take their used oil to a receptive service station or collection center are probably highly motivated by a desire to dispose of it properly. Given the value of time spent and the difficulty of locating a receptive service station, reclamation may indeed cost them \$2.00 to \$5.00 per gallon. For most DIYers, however, paying such an amount would contradict the cost-saving intent of changing their own motor oil. Under these assumptions, 95 percent of DIY sector oil would be reclaimed only if DIYers received a payment of about \$3.00 per gallon, assuming a standard deviation of \$2.00. This comports with our intuition, as well as a number of studies that have noted the difficulty of motivating DIYers to change their behavior.[19]

Looking ahead toward the efficacy of a refund, note that small values of σ imply relatively large effects on reclamation behavior -- a desirable result. Simply assuming that σ is small will ensure that the estimated social benefit from regulatory intervention will be relatively large. We find it difficult to imagine, however, that generators would

[19] See, e.g., New England Waste Management Officials' Association (1988); Carnegie Mellon (1988); Versar (1986); and New York State Legislative Commission (1986).

be highly responsive to price *increases* while the available evidence suggests that they have been markedly unresponsive to price *decreases.*

In sum, the constant variance hypothesis seems highly implausible. Low variances enable prices in the industrial and automotive sectors to be reasonably close, but they imply a degree of price-responsiveness that is not supported by any evidence. Higher standard deviations are needed to achieve less price-responsiveness, but they create differences in prices between the industrial and automotive sectors that are apparently counterfactual. Meanwhile, systematic differences in reclamation costs, potential liabilities, and scale economies suggest that the three sectors are in fact characterized by variations in price-responsiveness.

These contradictions can be resolved by allowing the standard deviations to differ across sectors. We assume that the industrial sector has a standard deviation of $0.50; commercial automotive generators have a standard deviation of $1.00; and the standard deviation for DIYers equals $2.00.[20]

Optimal Deposit and Refund Rates

The optimal second-best deposit rate derived in Equation 6-9 was shown to be equivalent to the residual external damage resulting from improper disposal, an amount that we denoted there as E_0. If there is a range of improper disposal alternatives, then the optimal deposit equals the value of risk reduction achieved weighted by the probabilities that the marginal gallon is removed from each disposal alternative.

[20]These assumptions are subjected to a sensitivity analysis later in the chapter.

As part of the 1985 RIA conducted in support of EPA's proposed used oil management standards, a risk assessment was performed to evaluate the relative risks posed by used oil across a variety of end-uses and dispositions. At several stages in this risk assessment, highly conservative assumptions were used that make the results, at best, suggestive of worst-case rather than typical risks. Since a critique of the risk assessment is beyond the scope of this report, we use the values derived for the RIA as a very conservative (and perhaps highly implausible) upper-bound.[21] Thus, our estimates of the optimal deposit and refund rates will likely be biased on the high side. Regulators would be well-advised to conduct a more appropriate risk assessment based on expected value methods before embarking on this or any other regulatory initiative with respect to used oil.

Table 7-3 summarizes the values obtained from this published risk assessment. Only cancer risks were included in the quantification of health effects, presumably because other types of outcomes were considered even more problematic. As is typical of EPA risk assessments, the number of cancers estimated to result from exposure to used oil was calculated in terms of 70-year lifetimes; these values are reported in column [1] for several alternative dispositions. However, the amount of oil related to each disposition was estimated in annual equivalents. To calculate the expected number of cancers per gallon of used oil requires that the total number of cancers be divided by 70 to yield the expected annual incidence. Thus, dividing column [1] by column [2], then divid-

[21] For a discussion of the EPA's risk assessment practices, see Nichols and Zeckhauser (1986).

ing the resulting quotient by 70 gives the estimated number of cancers per gallon. Multiplying by an appropriate value for cancer avoidance gives an estimate of the value of external residual damages posed by a gallon of used oil disposed or used in the identified manner. In Table 7-3 we use values of $1 million and $10 million per case.

Table 7-3. Quantification of Risk per Gallon Based on Published Used Oil Risk Assessment

End-Use or Disposition	Cancers/ Lifetime	Gallons per ear (10^6)	Cancers/ Gallon	Value of Residual Damages/Gallon	
				$10^6/ case	**$10^7/ case**
	[1]	[2]	[3]	[4]	[5]
Secondary Market					
Road Oiling	88	69	1.8×10^{-8}	$0.02	$0.18
Urban Burning	6,660	442	2.2×10^{-7}	0.22	2.20
Asphalt Plants	112	90	1.8×10^{-8}	0.02	0.18
Non-Secondary Market					
Space Heaters	192	34	8.1×10^{-8}	0.08	0.81
Incineration	18	15	1.7×10^{-8}	0.02	0.17
Landfill, Lined	889	25	5.1×10^{-7}	0.51	5.10
Landfill, Unlined	6,813	120	8.1×10^{-7}	0.81	8.10
Dumping	3,940	241	2.3×10^{-7}	0.23	0.23

Notes:

1. Source for column [1]: Temple, Barker and Sloane (1985a: Table V-42).
2. Source for column [2]: Temple, Barker and Sloane (1985a: Table V-36).
3. Column [3] = column [1] + column [2] + 70.
4. Column [4] = column [3] x **$10^6/cancer.**
5. Co1unm [5] = column [3] x **$10^7/cancer.**

Even if we hold fixed the value of preventing a case of cancer, these residual damage estimates span more than an order of magnitude. The highest estimate belongs to disposal in unlined landfills. Ironically, certain of the end-uses that the EPA had sought to restrict or prohibit (e.g., road oiling and incineration) were estimated to entail relatively little risk even under the extremely unfavorable assumptions.[22] For example, uncontrolled dumping was estimated to pose less than half the risk of disposal in lined landfills.[23]

To estimate the optimal deposit we must weight these risk values by the amount of unreclaimed oil that is directed to each end-use or disposition. Table 7-4 provides these weights. The per-gallon risk estimates reported in column [3] of Table 7-3 are multiplied by the quantities for each sector/end-use combination given in Table 7-4. These values are summed for each sector and normalized again as per-gallon risks. We report them in column [1] of Table 7-5. For example, the estimated risk from industrial oils that wind up burned in urban settings equals (39 x 10^6 gallons) x (2.2 x 10^{-7} cancers/gallon) - 8.58 cancers. After performing similar calculations for each of the remaining end-uses, the total number of cancers is summed and then divided by

[22]For example, the estimated risk posed by road oiling is tied for second-lowest of all end-use risks, yet the EPA proposed to ban it. One-tenth of the oil used for road oiling was assumed to be highly contaminated with chlorinated solvents (Temple, Barker and Sloane 1985a: IV-55). Of the 88 lifetime cancer cases estimated attributed to road oiling, 49 (55%) were due to the highly-contaminated fraction (Temple, Barker and Sloane 1985a: Exhibit IV-6). No justification was offered for this assumption. Reducing the highly-contaminated fraction to five percent reduces the expected number of lifetime cancers to 66. This would have resulted in an estimated annual cancer incidence of 1.4 x 10^{-8} cases per gallon -- a lower risk than all of the other end-uses.

[23]This may be attributable to major differences in estimation procedures between the two. For landfills, the EPA's "RCRA Risk-Cost Analysis Model" was used, a complex method requiring the specification of dozens of parameters from which a single point estimate is derived. For dumping, a considerably less formal procedure was followed. <u>See</u> Temple, Barker and Sloane (1985b: IV-11 to IV-19, IV-23 to IV-24).

the total volume of industrial sector oil that is not reclaimed in the secondary market. The resulting weighted average risk is 3.8 x 10^{-7} cancers/gallon, and is provided in column [1] of Table 7-5.

Table 7-4. Quantities by End-Use and Generation Sector, 1985

	Quantities (10^6 Gallons)				
		Non-Secondary Market			
End-Use Disposition	Secondary Market	Indus-trial	Auto-motive	DIY	Total
Re-Refining	64	-	-	-	-
Asphalt Plants	96	-	-	-	-
Disposal, lined LF	36	-	-	-	-
Urban Burning	439	39	59	9	107
Road Oiling	43	4	27	0	31
Space Heaters	-	0	34	0	34
Incineration	-	13	0	0	13
Disposal, unlined LF	-	68	5	46	119
Non-Fuel Industrial	-	46	0	0	46
Dumping	-	0	132	135	267

Notes:
1. Overall totals adapted from Temple, Barker and Sloane (1987a: Table III).
2. Non-Secondary Market sectoral breakdowns from Temple, Barker and Sloane (1987a: Table III).
3. Secondary Market Sectoral Breakdowns adapted from Temple, Barker and Sloane (1987a: Table I).
4. Non-fuel industrial uses are reclassified as belonging to the non-secondary market group because they apparently do not enter the secondary market.

The potential gain from increased reclamation equals the difference between secondary market and non-secondary market risks, and is given in column [2] of Table 7-5. Since secondary market risk is constant across sectors, the figure 2.0 x 10^{-7} cancers/gallon is simply subtracted from each of the risk estimates in column [1].[24]

Table 7-5. Risks, Potential Gains, and Optimal Deposit and Refund Rates, by Sector

	Estimated Risk (cancers/gal) [1]	Potential Gain (cancers/gal) [2]	Optimal Deposit ($/gal) [3]	Optimal Refund ($/gal) [4]
Secondary Market	2.0 x 10^{-7}			
Non-Secondary Market				
Industrial	3.8 x 10^{-7}	1.8 x 10^{-7}	$0.38	$0.18
Automotive	3.6 x 10^{-7}	1.6 x 10^{-7}	0.36	0.16
DIY	5.5 x 10^{-7}	3.5 x 10^{-7}	0.55	0.35
Wtd. Avg.	4.3 x 10^{-7}	2.3 x 10^{-7}	0.43	0.23

Optimal deposit rates are calculated by multiplying the appropriate risk estimates in column [1] by the implicit dollar value placed upon cancer prevention. For illustrative purposes we have used a figure of $1 million per case; these rates would be an order of magnitude larger if cancer prevention were valued at $10 million per case instead.

The optimal refund rate equals the dollar value of reductions in risk obtained by shifting a gallon of used oil into the secondary market. Thus, the potential risk gain reported in column [2] is multiplied by the social value of cancer prevention -- in Table 7-5, $1 million per case of cancer prevented.

---

[24] Once used oil enters the secondary market it is virtually impossible to track. Thus, we treat oil that enters the secondary market as a homogeneous commodity.

From this table it is apparent that unreclaimed DIY sector oil poses the greatest risk. Diverting this oil into the secondary market reduces risk by nearly a factor of three. Increased reclamation in the industrial and automotive sectors reduces risk by about half.[25]

Results with Zero Transactions Costs

Baseline scenario. The net benefit of a used oil deposit-refund system consists of the reduction in environmental risks obtained, less the social cost of achieving it. As a first approximation we calculate the net benefit under assumptions that give reasonable upper-bound estimates. We assume that the program perfectly discriminates across generation sectors (i.e., each sector receives its individually optimal

[25] As we indicated in Chapter 6, the deposit rate should be normalized by the input-waste ratio. In the case of used oil, about 40 percent of all oil generated either leaks or carbonizes in use. Thus, one might argue that the deposit rates shown in Table 7-5 should be adjusted downward by 40 percent. This yields optimal deposits ranging from $0.22 - $0.33 per gallon. However, oil that leaks or burns in use still enters the environment, causing residual external damages to air and water. Leaving the deposit rates unchanged implies that these damages range from $0.14 - $0.22 per gallon. The risk from crankcase leaks seems likely to be similar to the risk from road oiling absent any contamination from chlorinated solvents. An upper-bound estimate of this risk based on the RIA is 5.7×10^{-9} cancers/gallon. The risk from carbonization is approximately equal to that posed by urban burning, which was estimated in the RIA at 2.2×10^{-7} cancers/gallon. At $1 million per cancer, this implies a residual external cost ranging from about one-half cent to 22 cents per gallon. The appropriate normalization factor is the weighted average. According to Carnegie Mellon (1988: 15), leaks comprise between 30 and 45 percent of the total. Using 40 percent gives an expected value of 1.3×10^{-7} cancers/gallon, or a residual damage of 13 cents per gallon. Thus, normalizing reduces the optimal deposits for the automotive and DIY sectors to $0.29 ($0.16 + $0.13) and $0.48 ($0.35 + $0.13) per gallon, respectively. Generation rates for industrial oils are highly variable, and we do not have data concerning the fate of industrial oils not generated as "used". Thus, the proper normalization factor for the industrial sector is unknown.

refund); that there are no transactions costs (i.e., the deposit-refund system is costless beyond what is already captured by the supply curves); and that the standard deviations applicable to each sector are as specified earlier. We calculate net benefits using four different implicit values for cancer prevention ranging from $1 million to $10 million per case. The optimal refunds under these assumptions range from $0.16 to $3.50 per gallon, as shown in Table 7-6.

Table 7-6. Optimal Refund Rates for Baseline Scenario, by Sector

Value of Cancer Avoidance, ($/Case)	Optimal Refund Rates for Baseline Scenario, ($/gal)		
	Industrial Sector	Automotive Sector	DIY Sector
$ 1 million	$0.18	$0.16	$0.35
2 million	0.36	0.32	0.70
5 million	0.90	0.80	1.75
10 million	1.80	1.60	3.50

Absolute increases in secondary market prices are highest for the DIY sector, reflecting the greater risks posed by non-secondary market disposal. However, the relative price increase is greatest for the industrial sector because it is assumed to be much more price-responsive; the optimal refund raises the net price generators receive by at least 0.36 standard deviations ($0.18/$0.50).

These refunds achieve significant increases in the amount of used oil reclaimed. Predicted reclamation rates and percentage changes from the baseline are presented in Tables 7-7 and 7-8. Very high valuations on cancer avoidance imply large refunds, which elicit virtually complete reclamation in the industrial and automotive sectors. More than half of all DIY sector oil might be reclaimed. A refund of $0.90 per gallon is sufficient to attract all but one percent of the used oil generated in the industrial sector. A refund of $0.70 per gallon doubles the proportion of DIY sector oil reclaimed.

For each sector, the net benefit of the optimal refund equals the value of risk reduction obtained, less the added expenditures for reclamation. Estimated net benefits are reported in Table 7-9.[26] As the value of cancer prevention rises, the estimated net benefit increases almost fourfold.[27] Nevertheless, only if very large values are assigned to cancer avoidance will the aggregate net benefit ever exceed one-half billion dollars. If instead cancer avoidance is worth just $1 million per case, then the aggregate net benefit of the refund is less than $10 million.

---

[26]Mathematically, the net benefit for each sector (with perfect discrimination and zero transactions costs) equals the area underneath the cumulative unit normal distribution, less the amount already captured by the initial reclamation rate, multiplied by the total amount of oil generated, or:

$$NB = V\sigma \left[\int_{p_0}^{p^*} \left[\Phi(p)\ dp \right] - [\phi(p_0)](p^* - p_0) \right],$$

where: p_0 the initial price, in standard deviations; p^* = the optimal price including the refund, in standard deviations; $\Phi(p)$ = the cumulative normal probability distribution; $\phi(p_0)$ = the density of the cumulative normal at the initial price; σ = the value of the standard deviation, in dollars; and V = the total volume of oil generated, in gallons. Alternative valuations placed on cancer prevention are incorporated in p^*.

[27]If the supply curve were linear, then the net benefit would rise with the square of the value placed on cancer prevention. Our estimates are smaller because of the convexity of the supply curve, which becomes more pronounced at very high and very low reclamation rates.

Table 7-7. Reclamation Rates for Baseline Scenario, by Sector

Value of Cancer Avoidance, ($/Case)	Percent			
	Industrial Sector	Automotive Sector	DIY Sector	Weighted Average
(Before Refund)	68.6%	61.4%	5.0%	55.7%
$ 1 million	80.1%	67.4%	7.1%	63.4%
2 million	88.6	72.9	9.8	69.7
5 million	98.9	86.2	22.1	81.6
10 million	100.0	97.1	54.2	91.7

Table 7-8. Percentage Change in Quantity Reclaimed for Baseline Scenario, by Sector

Value of Cancer Avoidance, ($/Case)	Percent			
	Industrial Sector	Automotive Sector	DIY Sector	Weighted Average
$ 1 million	16.7%	9.7%	41.6%	17.6%
2 million	29.1	18.7	95.3	34.9
5 million	44.1	40.4	341.2	88.4
10 million	45.7	58.0	983.5	195.9

Whatever the implicit benefit valuation used, net benefits primarily arise from increased reclamation in the industrial and automotive sectors. Gains from increased DIY reclamation are relatively low because the DIY sector comprises a small fraction of the total and the relative risks from non-secondary market disposal are not extraordinarily high. At $1 million per cancer prevented, benefits from the DIY sector amount to just one-tenth of the total; even if cancer prevention is valued at $10 million per case, the DIY sector contributes less than one-fourth of the aggregate net benefit.

Table 7-9. Net Benefit for Baseline Scenario, by Sector

Value of Cancer Avoidance, ($/Case)	$ Millions			
	Industrial Sector	Automotive Sector	DIY Sector	Total
$ 1 million	$ 6	$ 3	$ 1	$ 9
2 million	22	11	3	35
5 million	100	60	24	184
10 million	252	199	137	588

<u>**Alternative standard deviations.**</u> The magnitude of the net benefit depends on the responsiveness of used oil generators with respect to the price of used oil in the secondary market; the higher the price, the more oil will be reclaimed. Generators' price-responsiveness is manifest in the implicit elasticity of the supply curve used to represent them. The more elastic the supply is assumed to be, the more responsive they are with respect to price changes and the greater will be the effect of a refund.[28]

[28]Because of the negative-price phenomenon, the usual notion of supply elasticity is inappropriate here. In this chapter we use the term "price-responsiveness" when we want to convey the conceptual notion behind elasticity without implying the technical term.

Table 7-10. Percentage Change in Estimated Net Benefit from Baseline Scenario with Alternative Standard Deviations

Value of Cancer Avoidance, ($/Case)	Percent Change			
	Industrial Sector	Automotive Sector	DIY Sector	Weighted Average
Low Standard Deviations				
$ 1 million	80	93	114	88
2 million	63	87	140	77
5 million	26	65	180	59
10 million	11	34	143	50
High Standard Deviations				
$ 1 million	-47	-43	-57	-46
2 million	-45	-48	-57	-46
5 million	-35	-45	-60	-42
10 million	-21	-39	-64	-37

Notes:
1. Low Standard Deviations: σ_I = $0.10; σ_A = $0.20; σ_D = $0.40.
2. High Standard Deviations: σ_I = $2.50; σ_A = $5.00; σ_D = $10.00.

We estimated net benefit based on both larger and smaller standard deviations. These results are summarized in Table 7-10. In the Low Standard Deviation case, values for σ are one-fifth as large as in the baseline scenario. As expected, greater price-responsiveness results in higher net benefit estimates. However, aggregate increases range only from 50 to 88 percent; only for the DIY sector are estimated net benefits more than double their baseline levels. Moreover, the percentage change declines as we move toward higher social valuations for cancer prevention. Thus, the greater the implicit value of life-saving, the less important are the distributional parameters that determine the supply curve.

In the same vein, systematically lower net benefit estimates arise for standard deviations higher than our baseline assumptions. The High Standard deviation case uses values for σ that are five times larger than the baseline. Estimated aggregate net benefits decline from 37 to 47 percent. As before, the percentage differences in the estimates typically decline as the implicit social value of cancer prevention increases. These changes are much less pronounced, however, and do not occur in each case.

The choice of distributional parameters clearly matters, but it matters less than the implicit social value placed upon cancer prevention. To get aggregate net benefits to approach $1 billion, implausibly high rates of price-responsiveness (i.e., low σ's) must be combined with relatively high values for cancer prevention. Under the less extreme assumptions suggested by our baseline case, the social gains from additional used oil reclamation seem relatively modest. Since the underlying risk assessment was intentionally biased to protect against worst-case events, even these gains may be illusory.

Sectoral discrimination infeasible. So far we have assumed that regulators can perfectly discriminate across sectors; that is, each can be assigned its sector-specific optimal refund rate. In practice this is unlikely to be administratively feasible. Industrial oils may be sufficiently different from automotive lubricants that little discrimination may be possible. However, variations in refunds predicated upon fine distinctions could lead to extensive cheating.[29]

Discriminating between commercial- and DIY-automotive oils seems especially impractical; the only difference between them is the place of generation. The relatively large spread in optimal refund rates between these two sectors would probably foster cheating, as commercial automotive generators attempted to obtain the larger DIY-sector refunds. Note that the larger the amount used as the value of cancer avoidance, the larger will be the optimal deposit and refund rates, and the greater will be the gap between the sectors. As the gap increases, so does the temptation to cheat.[30]

If perfect discrimination is the ideal, then a complete inability to discriminate is the polar opposite. Instead of sector-specific refund rates, regulators would select a single rate to apply to all sectors. Any intermediate refund rate will exceed the optimum in at least one sector and be too small in the other(s). Where the refund rate is too low, the system will fail to capture some of the potential social benefit. Where it is too high, too much used oil will be reclaimed and therefore create new social costs. The optimal refund rate is obtained at the point where the marginal social loss from excess control in one

[29]As we noted at the beginning of this chapter, waste generation rates differ by application as well as sector. For simplicity we have not considered this aspect of the problem; we expect that the administrative costs of designing and implementing application-specific deposits and refunds would greatly exceed the benefits even in those cases where it is feasible to do so. A carefully targeted system for PCB-laden transformer. oil might be an attractive exception.

[30]We discuss cheating in greater detail later in the context of a deposit-refund system proposed for Massachusetts.

(or more) sectors precisely equals the marginal social benefit foregone in the other(s).[31]

Figure 7-5 illustrates this problem. Let p_I^*, p_A^*, and p_D^* represent the optimal sector-specific refund rates and let $\hat{p}$ be some intermediate value. Industrial and automotive generators will reclaim oil at the levels $\hat{g}_I$ and $\hat{g}_A$, which exceed the sector-specific optima g_I^* and g_A^*. This generates new social costs from excess control equal to the darkly-shaded areas below each sector's supply curve. DIYers respond to the refund by reclaiming $\hat{g}_D$, an amount that is less than the optimum for the sector, g_D^*. Thus, the sub-optimal refund results in foregone benefits equal the lightly-shaded area above the DIY sector supply curve.

Whether foregone benefits exceed excess control costs is indeterminate without further specification of certain parameters. In general, the net benefit will be less than if perfect discrimination were feasible. A numerical example shows that the amount of potential net benefit lost through the use of a single refund rate need not be overwhelming, and that the "trial and error" process necessary to close in on an acceptable approximation of the optimum single rate is tedious but not as arduous as it may seem.

---

[31] The optimal single refund rate cannot be determined analytically. It can, however, be approximated through trial and error. Our discussion here is primarily concerned with how much potential net benefit is lost when a plausible heuristic rule is used to approximate the optimum.

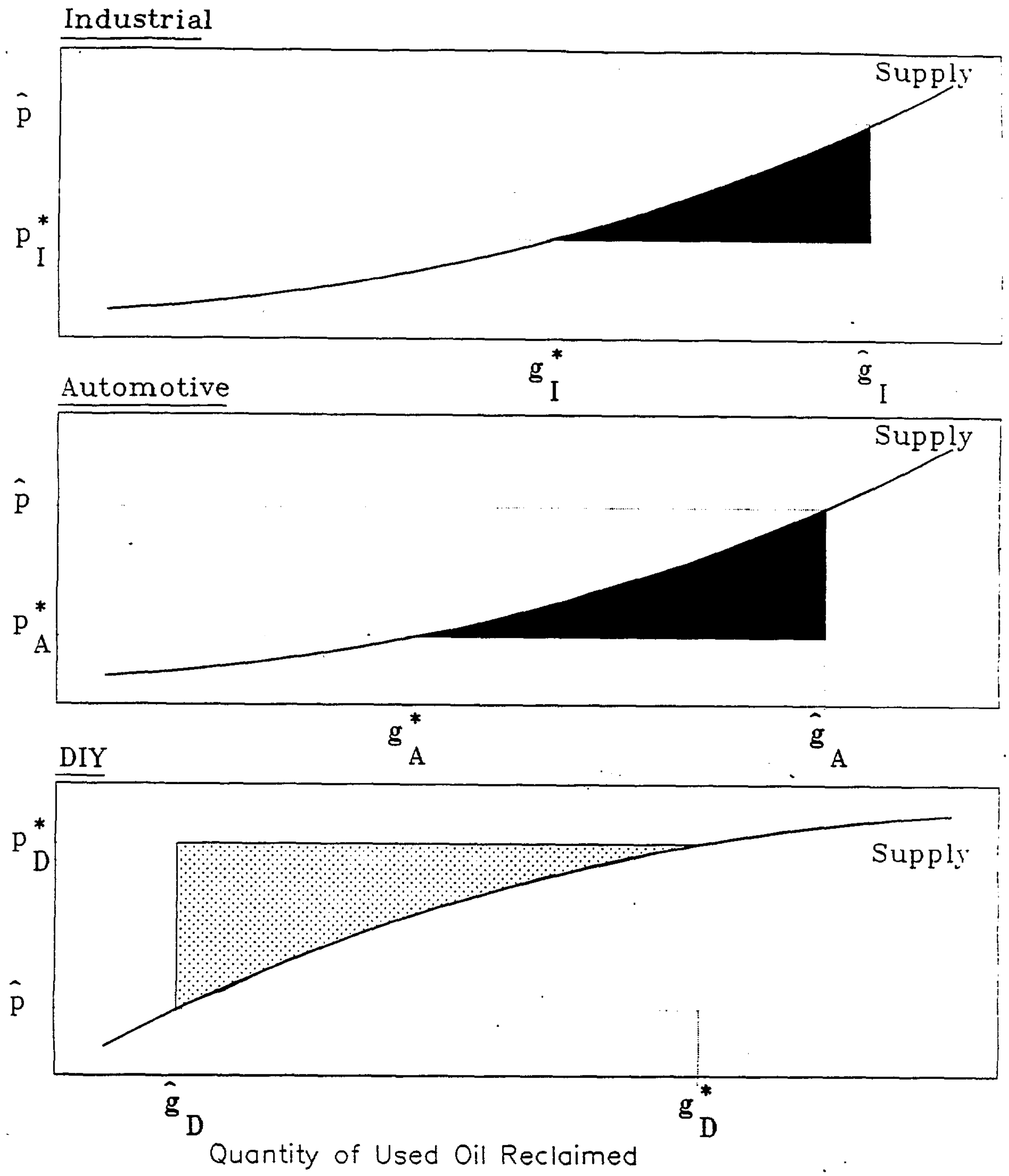


Figure 7-5. Excess Control Costs and Foregone Benefits, Nondiscrimination Scenario

Consider, for example, one plausible uniform refund rate -- the optimal rate applicable to the riskiest generation sector, DIYers. Since this rate exceeds the optima for both the industrial and automotive sectors, we can expect excess control costs to arise here.[32] The magnitude of these costs, of course, depends primarily on the difference in reclamation rates between the sector-specific optimum and the actual rate obtained from using a refund level that is too high.

Reclamation rates under this alternative scenario are presented in Table 7-11, along with the baseline rates under the perfect discrimination case. Using the DIY sector refund level increases the automotive reclamation rate by as much as 11 percentage points, up to eight points in the industrial sector. For very high valuations the sector-specific optimum rate approaches unity, and using the DIY sector refund rate causes only a small increase in the proportion of oil reclaimed above the sector-specific optimum. Excess control costs can be expected to be relatively small in these cases.

Table 7-12 confirms these expectations. In absolute dollars, excess control costs are their greatest when cancer-prevention is valued in the middle of the range examined, for that is where the gap between the actual and optimal reclamation rates is the greatest. These costs are smaller in the $1 million case because the difference between ex post prices in absolute dollars is small -- no more than $0.19 per gallon. Even a large degree of excess reclamation translates into a relatively small social loss under these circumstances. On the high side, excess control costs are low because the difference between quantities reclaimed is small. Large social losses per unit ($1.70 to $1.90 per

[32] Foregone benefits are zero because the chosen rate equals the optimum for the high-risk sector.

Table 7-11. Reclamation Rates, Baseline Scenario v. High Uniform Refund Rate

Value of Cancer Avoidance, ($/Case)	Percent			
	Industrial Sector	Automotive Sector	DIY Sector	Weighted Average
(Before Refund)	68.6%	61.4%	5.0%	55.7%
BASELINE SCENARIO: PERFECT DISCRIMINATION BY SECTOR				
$ 1 million	80.1%	67.4%	7.1%	63.4%
2 million	88.6	72.9	9.8	69.7
5 million	98.9	86.2	22.1	81.6
10 million	100.0	97.1	54.2	91.8
ALTERNATIVE SCENARIO: NO DISCRIMINATION, HIGHEST REFUND RATE APPLIED TO ALL SECTORS				
$ 1 million	88.2%	73.9%	7.1%	69.6%
2 million	97.0	83.9	9.8	77.9
5 million	100.0	97.9	22.1	87.1
10 million	100.0	100.0	54.2	92.9

Note: Uniform rate equals optimal rate for DIY sector.

gallon) translate into relatively small total losses when the number of excess units reclaimed is inconsequential.

While absolute losses from excess control are concentrated in the middle of the benefit valuation range, in percentage terms they decline as the valuation level rises. Excess control costs amount to as much as half of all benefits obtained from the industrial sector in the $1-2 million range, but drop to one percent or less for valuations of $5 million or more. Social losses in the automotive sector are considerably greater; they equal or exceed all benefits in $1-2 million range,

Table 7-12. Net Benefits, Baseline Scenario v. High Uniform Refund Rate

	$ Millions			
Value of Cancer Net Benefit, Avoidance, ($/case)	Net Benefit: Perfect Discrimin-ation [1]	Excess Control Costs [2]	Net Benefit: Uniform Refund at DIY Sector Optimum Rate [3]	Per-cent Loss [4]
$ 1 million				
Industrial	$ 6.0	$ 3.4	$ 2.6	57%
Automotive	2.8	3.1	-0.3	111
DIY	0.7	0.0	0.7	0
Aggregate	9.4	6.5	2.9	31
$ 2 million				
Industrial	$ 21.7	$ 6.3	$ 15.4	29%
Automotive	10.7	10.5	0.2	98
DIY	3.0	0.0	3.0	0
Aggregate	35.3	16.9	18.5	48
$ 5 million				
Industrial	$ 99.6	$ 1.0	$ 98.6	1%
Automotive	60.4	23.3	37.1	39
DIY	24.5	0.0	24.5	0
Aggregate	184.5	24.2	160.2	13
$10 million				
Industrial	$ 251.7	$ 0.0	$ 251.7	0%
Automotive	198.7	6.1	192.6	3
DIY	137.3	0.0	137.3	0
Aggregate	587.5	6.1	581.5	1

Note: Uniform rate equals optimal rate for DIY sector.

decline to 40 percent at the $5 million level, and finally decline to insignificant percentages when benefits are valued at $10 million or more.

In general, the optimal uniform refund rate cannot be determined analytically because too many parameters change simultaneously. However, it can be approximated through trial and error by weighing excess control costs suffered in the industrial and automotive sectors against benefits foregone in the DIY sector. From Table 7-9 it is apparent that social benefits come initially from increased reclamation of industrial and automotive oils. Only as the reclamation rates in these sectors approach unity does the DIY sector contribute an increasing share of the total.[33] Thus, if a relatively low value placed on cancer prevention seems more plausible, then excess control costs from a super-optimal refund rate will likely swamp the additional social benefits captured from DIYers. But if instead a high value for cancer prevention appears reasonable, then the optimal reclamation rates for the industrial and automotive sectors will already approach 100 percent. Raising the refund rate above the sector-specific optima will not have much of an effect on behavior, while at the same time enabling the system to achieve much larger social benefits from increased DIY reclamation.

An important lesson can be gleaned from this analysis: If for whatever reason it is infeasible to design the deposit-refund system so as to exploit differences across generation sectors, then the substitution of uniform rates as a second-best strategy may come with severe costs. To the extent that sector-specific optimal refund rates must be sacrificed, the system inevitably will have to trade off excess control costs against foregone benefits. The results are similar in many respects to the efficiency losses that accompany uniform standards, in

[33]Less than ten percent of total net benefits come from the DIY sector in the $1 million and $2 million cases. However, benefits from the DIY sector rise to 13 percent of the total in the $5 million case, and 23 percent in the $10 million case.

which the inability to discriminate across plants according to relative control costs sacrifices potential net benefits (Nichols 1984: 19). When balanced against the risk-reduction benefits achieved, these new costs imply ambiguous and potentially negative conclusions concerning the ultimate desirability of the deposit-refund system.

Results with Nonzero Transactions Costs

The effect of transactions costs is characteristically ignored in many policy analyses. It is especially important with respect to a deposit-refund system because transactions costs are likely to be borne over all units reclaimed, including the millions of gallons already delivered to the secondary market. For benefits to exceed costs, the gains from increased reclamation must be great enough to overwhelm these additional costs as well as the excess control costs and foregone benefits (if any) arising due to the inability to discriminate by sector.

Transactions costs can take several forms. First, the government inevitably will expend resources to administer the program. The funds to pay these costs may come out of general revenues, and thus displace other programs. More likely, funds will come from new taxes or additional public borrowing, either of which imposes additional social costs on the economy. Given current political constraints on increased general taxation, policy makers may find it expedient to enact special purpose levies upon the participants in the deposit-refund system. Any such taxes, even if defended as being equivalent to "user fees," will reduce the net value of refunds received, and by extension, the effectiveness of the program.

Second, any deposit-refund system will probably entail additional costs for participants to document and certify eligibility. These costs may seem trivial at first glance, but even small unit costs can multiply into enormous sums. Benefits of the system accrue only at the margin, but the costs of complying with the requirements established by the program must be paid over all units reclaimed.

As a first cut we can use the results reported in Figure 7-1 and Tables 7-7 and 7-9 to derive rough upper-bound estimates of the maximum level of transactions costs that can be incurred by the government without entirely eliminating the net benefit. These estimates are provided in Table 7-13; they are calculated by dividing the net benefit by the volume of oil reclaimed in each sector. Note that while these maxima increase with the implicit value of cancer prevention, they should in no case be considered "large" values. Indeed, they offer precious little room for error if positive net benefits are to be preserved.[34]

Whether these maxima pose any threat to the viability of the deposit-refund system depends on the nature of the eligibility requirements established and the extent of documentation required. The tighter the rules and the more extensive the reporting requirements imposed, the higher will be generators' cost of complying with the system. Relatively high transactions costs are not implausible. Laboratory chemical analyses can run into the thousands of dollars per batch and swamp all but the most generous of refunds. Field tests for detecting chlorinated

[34] The accuracy of this approximation declines precipitously with higher values for cancer prevention due to the convexity of the supply curves.

Table 7-13. Approximate Upper Limit on Transactions Costs for Viable Program under Baseline Scenario, $/Gallon

Value of Cancer Avoidance, ($/case)	(Approximate) Maximum Transactions Costs, $/gal		
	Industrial Sector	Automotive Sector	DIY Sector
$ 1 million	$ 0.014	$ 0.008	$ 0.046
2 million	0.045	0.027	0.151
5 million	0.186	0.127	0.555
10 million	0.464	0.371	1.267

solvents have become available recently (Carnegie Mellon 1988: 70); at a cost of $5 per kit, however, even these tests become prohibitively expensive for small amounts of oil.[35] Moreover, there is no economical test available to detect the presence of heavy metals.

Typically, the government simply mandates that transactions costs be borne by regulatees without much concern for their magnitude. The EPA's burning and blending regulations are illustrative. These rules distinguish between "specification" and "off-specification" used oil. Waste that exceeds certain specified thresholds for toxic metals, flash point, or total halogens is defined as "off-spec."[36] Because of the high cost of testing (estimated by one source at $200-250 per sample and confirmed by a price list published by NUS Corporation, a major analyti-

[35] The cost of testing a full 55-gallon drum of used oil is therefore about $0.10 per gallon. Average costs decline dramatically for large tanks, of course.

[36] These thresholds are: 5 ppm (arsenic); 2 ppm (cadmium); 10 ppm (chromium); 100 ppm (lead); and 4,000 ppm (total halogens) See 40 CFR 266.40, 50 FR 49205 *et seq.,* November 25, 1985.

cal testing firm), the EPA allows generators to "certify in lieu of testing" that their used oil is indeed "spec." However, this certification does not negate the need for testing, nor does it extinguish potential liabilities should the oil subsequently be tested and revealed to be "off-spec." Moreover, the rules establish a rebuttable presumption that used oil containing 1,000 ppm or more of total halogens has indeed been adulterated with hazardous wastes. In practice, this presumption is impossible to rebut without testing every shipment. If a representative industrial shipment is 1,000 gallons and testing costs $250 per batch, the average cost is $0.25 per gallon -- an amount that exceeds the maximum transactions costs for industrial generators in all but the $10 million valuation case. Representative automotive generators with 500 gallon tanks would incur testing costs of $0.50 per gallon -- an amount that exceeds the maximum transactions costs for any benefit valuation considered. Testing is prohibitively expensive for DIYers irrespective of any plausible value assigned to cancer avoidance.[37]

Of course, the burning and blending rules are not accompanied by a refund that could at least lessen this burden. Testing requirements (whether explicit or implicit) reduce the secondary market value of used oil (whether spec or off-spec), discourage the recycling of used oil,

[37]The EPA estimated that its burning and blending regulations would cost no more than $21 million per year, or about three cents per gallon of used oil then reclaimed (50 FR 49201). Thus, the costs of testing were either not included at all, or it was presumed that because generators and marketers could "certify in lieu of testing" that only spot tests would be necessary. Choosing to certify is not costless -- costs are simply manifest in the form of price discounts applied to "certified" but untested oil. These discounts are transmitted to the generator level, where they result in a reduction in the amount of used oil reclaimed.

and lead to increased disposal and dumping. If testing is prohibitively expensive, then noncompliance becomes virtually the only viable alternative. Anecdotal evidence gathered by the EPA confirms this: indiscriminate "certification in lieu of testing" has become the rule rather than the exception.

If transactions costs are indeed borne by used oil generators, then they should be reflected as reductions in the net value of the refund. The effect of transactions costs is illustrated in Figure 7-6. As before, the initial price and quantity are denoted p_0 and g_0. The optimal refund drives the price up to p*, increasing the amount of used oil delivered to the secondary market to g*. The net benefit is equal to the area above the supply curve surrounded by the thick edge.

But transactions costs reduce the net value of the refund and diminish its capacity to stimulate additional compliance. Suppose that transactions costs reduce the net refund to p'. Generators will increase their level of reclamation only to g' rather than g*. The net benefit from increased reclamation is now equal to the cross-hatched area plus the lightly-shaded rectangle. In addition, transactions costs must be paid on all units that entered the system prior to the establishment of the refund; these costs equal the sum of the two shaded rectangles. Therefore, the net benefit after deducting transactions costs equals the cross-hatched area less the darkly-shaded rectangle.[38]

[38] The lightly-shaded rectangle cancels out.

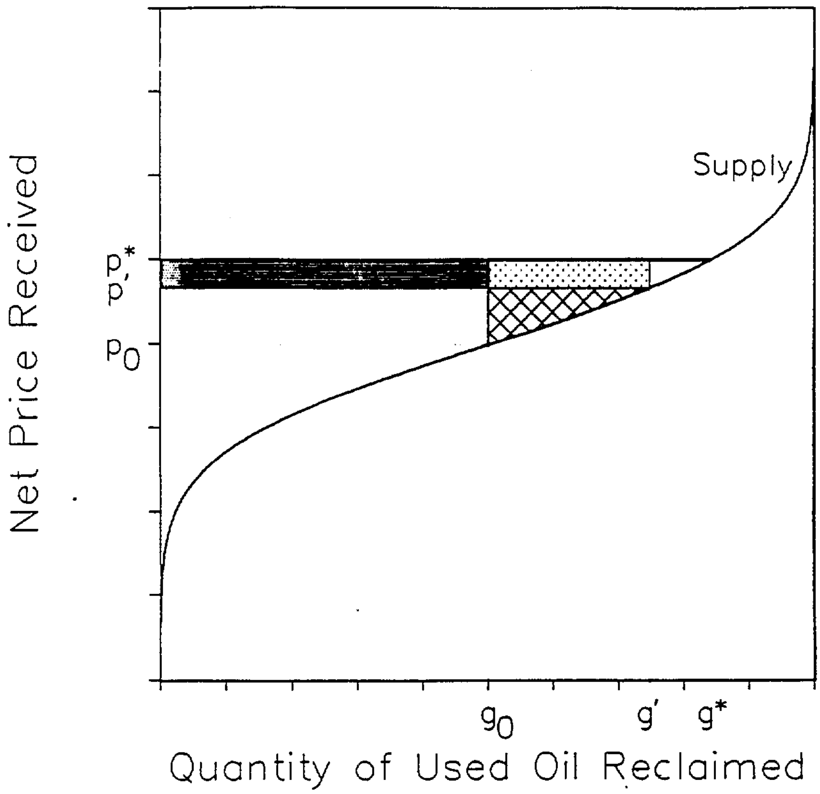


Figure 7-6. Net Benefit of Refund with Transactions Costs

Whether the social gains from increased reclamation outweigh the transactions costs borne to achieve them is not clear. The result depends on the magnitude of transactions costs per unit and the number of units on which they must be paid. The higher the initial reclamation rate, the greater will be the burden of transactions costs and the less likely it is that the benefits of the refund will exceed these costs.

A much more serious problem arises if the costs of satisfying the government's eligibility requirements exceeds the value of the refund. This may seem bizarre, but testing requirements such those described earlier could be enough to make it happen. Figure 7-7 illustrates the effects of such a program, assuming that generators wishing to comply with regulations are not able to refuse to participate in the refund system.[39]

As before, p_0 and g_0 indicate the initial price and quantity, and the optimum is denoted by p* and g*. If transactions costs exceed the value of the refund (p* - p_0), the net effect is to drive the price down below p_0, say to p". Generators respond to this lower price by reducing to g" the amount of oil they reclaim. If social benefit is created only when additional oil flows into the secondary market, then there can be no gain if transactions costs exceed the value of the refund. Such a regime only imposes new social costs, in this case an amount equivalent to the shaded area above the supply curve.[40]

This result is analytically identical to what can be expected to occur if regulators promulgate standards that increase the cost of reclamation (or proper hazardous waste disposal) and cannot ensure compliance. The expected cost per unit of complying with these standards can be interpreted as p_0 - p". Generators reduce the amount of oil reclaimed (or wastes properly disposed) according to their degree of price-responsiveness as indicated by the slope of the supply curve. If

[39] If participation is truly voluntary and transactions costs exceeds the value of the subsidy, firms simply will not participate and the program will have no effect.

[40] The optimal refund, p* - p_0, equals the expected value of residual external damages prevented by shifting a unit of waste *into* compliance. Thus, it is also equal to the expected value of residual external damages that result when a unit is shifted out. Thus, observing the shaded area in Figure 7-7, the increased residual external damages from reduced reclamation can be viewed as equivalent to an imaginary rectangle whose southeast corner is anchored on the supply curve. Reduced reclamation also results in a loss of producer's surplus equal to the area below this rectangle and bounded by the supply curve.

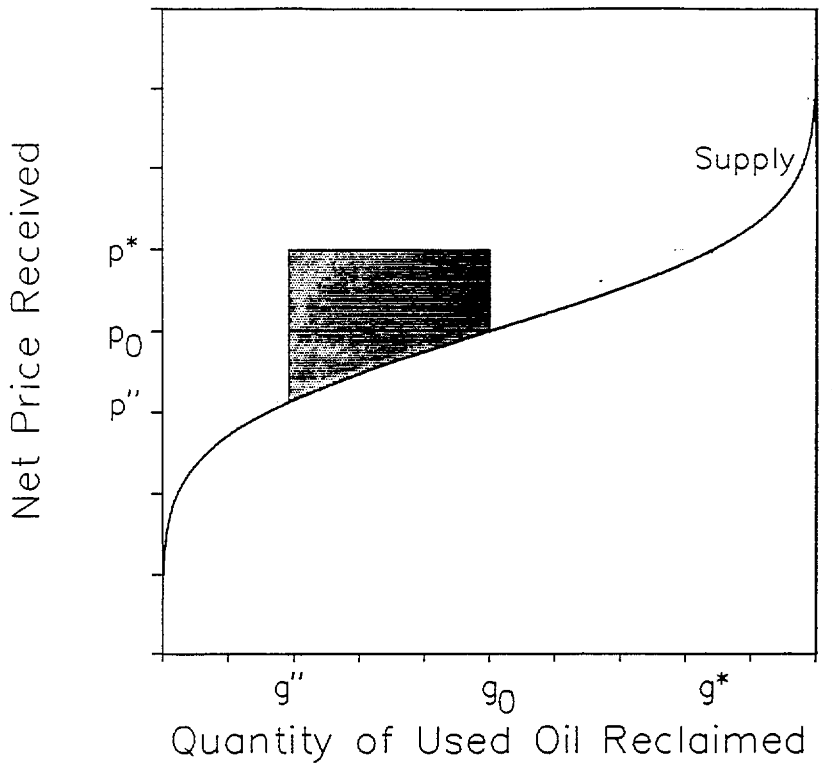


Figure 7-7. Social Loss When Refund Is Swamped by High Transactions Costs

the reclamation (or compliance) rate is initially very high or very low, then the perverse effect of the regulations will be relatively small because generators are not very responsive to price changes. Rates in between, however, will be subject to relatively large changes in behavior, and consequently large social losses.

Baseline scenario with transactions costs. We have estimated the effects of a wide range of transactions costs for both the perfect discrimination (baseline) and nondiscrimination scenarios. Because the inability to discriminate by sector unambiguously reduces net benefits, only results for the perfect discrimination (baseline) case are reported here.

Absent any theoretical basis for specifying particular amounts, we evaluated transactions costs ranging from $0.02 to $2.00 per gallon. Low levels represent what might be expected if regulators impose minimal certification requirements; high cost levels are indicative of significant testing and documentation expenses.

As indicated earlier, transactions costs reduce net benefits two ways. First, they are an absolute drain on social welfare. Eligibility requirements and documentation serve to increase the marginal social cost of whatever behavior is involved -- in this case, reclamation for use by the secondary market. Second, they diminish the value of the refund, thereby inducing a smaller increase in reclamation. When unit transactions costs exceed the refund, then then reclamation rate actually declines from its initial level.

Reclamation rates for the baseline scenario after deducting for the effects of transactions costs are reported in Table 7-14; values with asterisks represent cases in which unit transactions costs exceed the value of the refund, thereby making the net effect equivalent to a tax on reclamation. Note that for the lower cancer-prevention valuations, even "modest" levels of transactions costs below the maxima indicated in Table 7-13 cause significant declines in effective reclamation rates. Transactions costs of $0.25 per gallon reduce the effective reclamation rate by as much as 14 percent in the automotive sector, 21 percent among industrial generators. In the high valuation cases reductions of this magnitude do not materialize until unit transactions costs reach one

Table 7-14. Reclamation Rates for Baseline Scenario, Including Transactions Costs

Value of Cancer Avoidance, ($/case)	Trans-actions costs ($/gal)	Percent			
		Industrial Sector	Automotive Sector	DIY Sector	Weighted Average
$ 1 million					
	$ 0.00	80.1%	67.4%	7.1%	63.4%
	0.02	79.0	66.6	6.9	62.6
	0.10	74.1	63.7	6.4	59.2
	0.25	63.5*	57.9*	5.5	52.2
	0.50	43.9*	48.0*	4.3*	39.5
	1.00	12.4*	29.1*	2.4*	18.0
	2.00	0.1*	6.1*	0.7*	2.7
$ 2 million					
	$ 0.00	88.6%	72.9%	9.8%	69.7%
	0.02	87.8	72.2	9.6	69.1
	0.10	84.3	69.5	8.9	66.3
	0.25	76.0	64.1	7.8	60.4
	0.50	58.1*	54.4*	6.1	48.5
	1.00	21.3*	34.8*	3.6*	24.4
	2.00	0.3*	8.2*	1.1*	3.4
$ 5 million					
	$ 0.00	98.9%	86.2%	22.1%	82.6%
	0.02	98.8	85.8	21.8	81.3
	0.10	98.1	83.9	20.6	80.1
	0.25	96.3	79.9	18.5	77.3
	0.50	90.1	72.2	15.4	70.9
	1.00	61.2*	53.6*	10.2	50.1
	2.00	4.3*	18.1*	3.8*	10.1
$10 million					
	$ 0.00	100.0%	97.1%	54.2%	91.7%
	0.02	100.0	96.9	53.8	91.6
	0.10	100.0	96.3	52.2	91.1
	0.25	100.0	94.9	49.2	90.0
	0.50	99.9	91.8	44.2	87.8
	1.00	98.1	81.3	34.6	81.2
	2.00	53.4*	45.65*	18.5	44.7

Note: Asterisk following entry signifies negative net refund (i.e., unit transactions costs exceed unit refund).

dollar or more. This should offer little comfort, however, because the larger optimal refunds implied by these higher valuations make it likely that unit transactions costs will be larger than these "modest" amounts. Total transactions costs are presented in Table 7-15 for each of the four cancer-prevention valuations considered. Depending on the value placed on cancer prevention, total transactions costs range from $16 million to $24 million when unit transactions costs amount to just two cents per gallon. The increase in total costs is less than proportional to any increase in unit costs, because generators respond to transactions costs by reducing the amount of oil they deliver to the system. Total transactions costs are smaller for very high unit costs only because reduced reclamation activity drives out so many generators.

High absolute refunds seem likely to induce intense demands for public accountability. Thus, if a high value is placed on preventing cancer so as to justify a large refund, then high transactions cost scenarios become more plausible. For example, the system can technically sustain transactions costs equal to $0.50 per gallon in the $10 million case because net refunds remain positive. However, total expenditures on transactions costs alone would exceed $500 million. Unlike expenditures on improved waste management, transactions costs do not result in any life-saving benefits. If these funds could be devoted to substantive purposes an additional 57 cancers per year could be prevented.

Net benefits less transactions costs are reported in Table 7-16. If cancer prevention is valued at $1 million per case, then virtually any plausible level of unit transactions cost swamps the potential social gain. The estimated net benefit *absent* transactions costs is less

Table 7-15. Transactions Costs for Baseline Scenario

Value of Cancer Avoidance, ($/case)	Trans-actions costs ($/gal)	Transactions Costs in $ Millions			
		Industrial Sector	Automotive Sector	DIY Sector	Total
$ 1 million					
	$ 0.02	$ 9	$ 7	$ 0	$ 16
	0.10	40	35	1	77
	0.25	86	80	3	169
	0.50	119	133	4	256
	1.00	67	161	5	233
	2.00	1	67	3	70
$2 million					
	$ 0.02	$. 10	$ 8	$ 0	$ 18
	0.10	46	38	2	86
	0.25	103	88	4	195
	0.50	158	150	6	314
	1.00	116	192	7	315
	2.00	3	91	4	98
$5 million					
	$ 0.02	$ 11	$ 9	$ 1	$ 21
	0.10	53	46	4	104
	0.25	130	110	9	250
	0.50	244	199	15	459
	1.00	332	296	20	648
	2.00	47	200	15	262
$10 million					
	$ 0.02	$ 11	$ 11	$ 2	$ 24
	0.10	54	53	10	118
	0.25	136	131	25	291
	0.50	271	253	44	568
	1.00	532	449	69	1,050
	2.00	579	504	74	1,157

Note: Asterisk following entry signifies negative net refund (i.e., unit transactions costs exceed unit refund).

than $10 million; unit costs of just two cents per gallon converts this gain unto a $7 million loss. Valuing cancer prevention at $2 million per case preserves just $17 million in benefits, a 50 percent reduction from the zero transactions cost case. A small increase in unit transactions costs above two cents is sufficient to swamp these gains as well. High cancer-prevention valuations are necessary to enable the system to withstand the social losses imposed by even moderate transactions costs. Nevertheless, no plausible value can overcome a regime in which transactions costs are high.[41]

At least three important lessons can be inferred from this analysis. First, simply relying upon a high value for cancer prevention will not make regulatory intervention cost-effective. If transactions costs are high, then intervention may still fail to generate net social benefits.

Second, the level of transactions costs may be more important than the implicit value of life-saving in determining whether or not a deposit-refund system provides any social benefit. If initial compliance rates are high and nontrivial documentation cannot be avoided, then a deposit-refund system may be ill-advised. The social costs imposed upon the vast majority of participants may greatly exceed any plausible social benefit that might be gleaned from those few motivated to act responsibly only by virtue of the refund.

---

[41]Absolute refunds depend on the amount used to value cancer prevention benefits, so some levels of unit transactions costs imply negative net refunds. Presumably, policy makers would not seriously consider any deposit-refund system having negative net refunds. Thus, we include these figures only for purposes of comparison, and have identified them with asterisks. In these cases, social losses due to reduced reclamation have also been deducted.

Table 7-16. Net Benefits for Baseline Scenario with Transactions Costs

Value of Cancer Avoidance, ($/case)	Trans-actions costs ($/gal)	Net Benefits in $ Millions			
		Industrial Sector	Automotive Sector	DIY Sector	Total
$ 1 million					
	$ 0.00	$ 6.	$ 3.	$ 1.	$ 9
	0.02	-3.	-5.	0.	-7
	0.10	-36.	-33.	-1.	-70
	0.25	-90.	-82.*	-2.	-175
	0.50	-153.*	-151.*	-5.*	-309
	1.00	-209.*	-248.*	-7.*	-464
	2.00	-219.*	-324.*	-9.*	-553
$ 2 million					
	$ 0.00	$ 22.	$ 11.	$ 3.	$ 35
	0.02	12.	3.	3.	17
	0.10	-25.	-29.	1.	-53
	0.25	-91.	-84.	-1.	-176
	0.50	-172.*	-160.*	-5.	-337
	1.00	-241.*	-265.*	-9.*	-515
	2.00	-253.*	-349.*	-11.*	-613
$ 5 million					
	$ 0.00	$ 100.	$ 60.	$ 24.	$ 184
	0.02	89.	51.	24.	163
	0.10	-46.	13.	20.	80
	0.25	-33.	-55.	14.	-73
	0.50	-160.	-160.	6.	-314
	1.00	-352.*	-318.*	-7.	-676
	2.00	-359.*	-430.*	-18.*	-807
$ 10 million					
	$ 0.00	$ 252.	$ 199.	$ 137.	$ 588
	0.02	241.	188.	135.	564
	0.10	197.	146.	127.	470
	0.25	116.	66.	111.	294
	0.50	-19.	-63.	88.	6
	1.00	-289.*	-303.*	49.	-543
	2.00	-662.*	-591.*	-3.	-1,256

Note: Asterisk following entry signifies negative net refund (i.e., unit transactions costs exceed unit refund).

Third, even if net benefits can be obtained from increased reclamation in one sector, they may be negative elsewhere. For example, if cancer prevention is valued at $10 million per case and transactions costs amount to $1.00 per gallon, social benefits of about $50 million can be obtained from the DIY sector. However, the same program applied to industrial and automotive results in about $600 million in social losses. Thus, it may be desirable to implement a deposit-refund system selectively -- in the DIY sector, perhaps, but not elsewhere. Such a strategy presumes, of course, that discrimination by sector is technically, administratively, and politically feasible. If it is not, then even these gains may be illusory.

Targeting, Administration and Cheating

In the final section of this chapter we examine several qualitative issues that arise in the event that a plausible quantitative case can be made for a deposit-refund system. How should the deposit and refund be targeted? Should government or some element in the private sector act as the broker for the system, collecting deposits and disbursing refunds? Can the system be abused, either by generators through fraudulent means or by brokers through the coercive powers inherent in this adjudicatory role? If government brokers the system, what is the risk that the program will become more valuable as an indirect means of generating revenue than as a strategy for internalizing the residual external damages from hazardous waste generation and mismanagement?

Targeting and Cheating

One of the reasons used oil is an attractive candidate for a deposit-refund system is that it is relatively easy to target the incentive components. Deposits would be levied on the manufacture or sale of new lubricating oil (both virgin and re-refined); for reasons we discuss below, refunds are probably best offered at the point of end-use.

On the deposit side, it is important to ensure that virtually all sources of new lubricating oil are covered. Programs that would exempt or otherwise subsidize re-refined oil, for example, are misguided.[42] Given the relatively small number of lube oil manufacturers and the extent to which they are already identified and regulated for other reasons, levying the deposit at the point of manufacture would probably be the administratively efficient strategy. The simplicity of this approach may be lost, of course, if regulators attempt to establish sector-specific deposits. Industrial oils and automotive lubricants may be easy to distinguish but the difference in risks posed by them is apparently insignificant in most cases. DIY-automotive risks are roughly twice as large as those posed by commercial-automotive generation, but they arise from identical new oils and create identical waste streams. Regulators probably would have to mandate rules governing packaging and distribution so as to create a means for distinguishing DIY- from commercially-installed motor oil and thereby discourage cheating. These rules, as well as the costs of enforcing them, would add to the social

[42]For proposals in this vein, see, e.g., Carnegie Mellon (1988); New England Waste Management Officials' Association (1988); and New York State Legislative Commission (1986). Used oil that originates from re-refined lube oil entails the same potential environmental risks as used oil that comes from virgin lube. Thus, exempting re-refined lube oil creates a subsidy that is unwarranted based on risk considerations. Energy savings or other putative social benefits should be accounted for elsewhere rather than grafted onto a risk-based economic incentive system.

costs of the system and diminish any apparent advantage from discrimination.

The point at which refunds should be offered is not clear cut. If regulators want to discriminate across generation sectors, then multiple refund targets will be necessary; otherwise there is no way to transmit a larger refund to DIYers. Regulators might have to establish special collection facilities for DIY oil. Once again, this creates additional social costs that weaken any risk-based argument for differential refund rates.

A single refund rate saves this administrative nightmare while it sacrifices some of the potential social benefit of control. Such a refund is best offered at the point of end-use; intermediate refund points do not ensure that the oil is actually delivered to socially desirable end-uses By offering the refund only to designated end-users, regulators can also limit the number of entities they must monitor.[43]

A recent proposal for a used oil deposit-refund system in Massachusetts illustrates clearly the difficulty of selective targeting (Massachusetts Legislature 1988). This bill would establish a $0.50 per quart deposit on all automotive and marine lubricating oil *not installed on the premises;* i.e., on DIY-installed motor oil. The deposit would be collected by the retailer and held for no more than six months, after

[43]Targeting approved secondary market end-uses also enables regulators to use the price system to redirect used oil flows within the secondary market. We have not analyzed an end-use specific refund system for this report. It is worth remembering that significant diversions within the existing market may exert downward pressure on prices within the approved end-uses, thereby weakening the effectiveness of any given refund level.

which unclaimed deposits would be forwarded to the state where they would be earmarked for enforcement. Upon delivery of an equal quantity of used oil and proof of purchase, the retailer would refund the deposit to the consumer. All retailers of new oil would also be required to install used oil collection tanks to handle these returns, and they would be considered full-fledged hazardous waste generators.

Opportunities for cheating in this system are widespread. Because the deposit applies only to DIY-installed motor oil, retailers that sell to DIYers and also perform oil changes on the premises have an incentive to underreport their retail sales to DIYers and keep the deposits. With the wholesale price of new lubricating oil below $1 per quart, the deposit represents a significant source of potential profit. Moreover, the retailer need not collude with his supplier, his customers, or the hauler who takes his used oil away. Given that the state currently requires extensive reporting by used oil collectors but does not do anything with the information it receives, the chance that discrepancies will be detected and punished seems quite remote.[44] Retailers that cheat will enjoy a competitive advantage over their law-abiding counterparts because they will be able to sell at a discount.

Ironically, the proof of purchase requirement also creates additional barriers for DIYers. Few consumers will bother keeping receipts; many would find it inconvenient to return oil to the place at which they

[44]Even if discrepancies were found, it would be difficult for regulators to infer that the retailer was actually underreporting. The bill makes no provision for partial quarts; thus it is unclear what base retailers would be expected to use to collect deposits and pay refunds. Differences in interpretation would be enough to easily explain a 25 percent shortfall because of the small size of individual batches.

originally purchased it. At the margin, this system seems more likely to deter the reclamation of DIY oil rather than encourage it.

Administration and Cheating

An important question is whether the government or some private entity should hold deposits and disburse refunds. The answer turns, at least in part, on which party has a comparative advantage in providing these brokerage services. If government acts as the broker then the system will gain from greater visibility and political accountability. Conflicts of interest may arise between program objectives and revenue generation, but official brokerage at least increases the chance that these conflicts can be resolved with the imprimatur of responsible policy makers.

Private sector brokerage makes sense if the government is ill-equipped to efficiently administer the system. But regulators will have to promulgate additional rules governing how private sector brokers operate because the brokers will have a financial incentive to discourage redemption.

Under the Massachusetts proposal, the identity of the broker is ambiguous. If retailers report accurately, the state is the principal beneficiary and retailers become uncompensated conduits for revenue generation. But if retailers cheat, then they become the brokers and are rewarded handsomely for providing this service. As brokers, retailers have a financial incentive to discourage redemption: unclaimed deposits become increased profits. To control the brokers, the state will have to regulate terms of trade, such as the amount of oil that can be redeemed in any given period, the hours of operation, the form in which refunds are paid, and a host of other factors. Moreover, these regulations must be enforced -- a difficult proposition given the number of retailers and the potential number of transactions.

In general, it is desirable to keep the number of brokers to a minimum so as to maintain some capacity for administrative oversight. The Massachusetts proposal does not do this. Instead, every lube oil retailer becomes a broker, a potential source of cheating, and a threat to the program's efficacy.[45]

Private sector brokerage is most attractive when for political reasons policy makers want to obscure the full costs of the program. The system will appear to break even or perhaps make money, but the actual costs of running the program are borne elsewhere, costs that will be substantial if the volume of waste is large.

Subterfuge

In the Massachusetts proposal discussed above, the refund would be set at $0.50 per quart. Based on our models, this implies a valuation of more than $5 million per cancer avoided, the level for which the op-

[45] Like several other states, Massachusetts already has a deposit-refund system for beverage containers. The administrative problems associated with private sector brokerage are controlled by partially compensating retailers for redemption services and assigning the brokerage function to wholesalers -- a much smaller group of firms, most of which are already subject to regulation by the state Alcoholic Beverages Control Commission. Nonetheless, there have been complaints that beverage wholesalers have engaged in certain practices that, by intent or accident, effectively discourage redemption. See Marantz (1986) and Commonwealth of Mass. v. Mass. Crinc (1984), 466 N.E. 2d 792, 392 Mass. 79 (state sought and obtained an injunction prohibiting beer distributors and corporation formed by them from requiring retailers to comply with the corporation's pickup schedule or pay a penalty).

timal refund was $1.75 per gallon. Thus, the Massachusetts proposal falls in the middle of the range of benefit valuations analyzed earlier.

In that analysis we assumed implicitly that there was no practical limit to how large a refund could be, so long as it was economically justified. This may be far from the truth. With a refund at $2.00 per gallon, used oil becomes considerably more valuable than the petroleum products for which it is a partial substitute. Thus, such a large refund creates an incentive for virgin fuel to be blended with (or substituted for) used oil. No. 6 residual fuel oil can be purchased for about one-fourth this amount; "reclaiming" it through the secondary market for used oil thus offers a substantial return on investment.

In the Massachusetts proposal, retailers can readily engage in this kind of subterfuge; they will do so, of course, only to the extent that less-expensive forms of cheating are not available. If state regulators became unusually vigilant in monitoring retailers' deposit-refund system reports in an effort to detect underreporting, residual fuel oil could be covertly purchased to make up any discrepancies. A universal deposit-refund system established at the national level with refunds of this magnitude would have all sorts of problems with this kind of substitution.

At the front-end, a deposit as large as $0.50 per quart could seriously distort consumer purchase decisions and thus create new social costs. In the Massachusetts case, the deposit would probably cause a significant reduction in do-it-yourself oil changes. Retailers such as discount stores and supermarkets would stop selling motor oil rather than install used oil collection tanks. This implicit restriction on entry would drive up the price of motor oil and encourage many DIYers to have their oil changed at neighborhood service stations and "quick lube" facilities instead. These changes may be welcomed by service station owners and the quick-lube business, but they still represent inefficiencies in the production and consumption of oil change services. Inevitably, some consumers will respond by unwisely delaying an overdue oil change, thereby risking serious engine damage. Our quantitative estimates, which showed substantial negative net benefits once even minor transactions costs were included, did not include any of these additional costs.

The Massachusetts proposal also suffers jurisdictional problems if consumers choose to purchase motor oil out of state. Many DIYers can be expected to do just that. The expenditures made by consumers to evade the deposit as well as efforts to profit on interstate price differences attributable to the refund constitute additional social costs.

In sum, even if the potential environmental risks warrant relatively large deposits and refunds, there will be limits as to how large they can go. Large deposits will significantly distort purchase decisions, resulting in inefficient production and consumption decisions that create new social costs. Large refunds create opportunities for virgin materials to be substituted in place of the target waste stream, a waste of valuable resources. Controlling these these side-effects requires a costly investment in regulatory enforcement.

Concluding Comments

The assumptions that must be combined to "demonstrate" the cost-effectiveness of a deposit-refund system for used oil seem highly implausible. Because such a system would achieve increasing rates of reclamation in an efficient manner, any other regulatory strategy -- particularly a traditional standards-based approach -- would perform even more poorly. The failure lies not with the deposit-refund instrument but in the small risks posed by used oil. If the risks were significant, then the burdensome transactions costs implied by high initial reclamation rates would not necessarily dominate.

Taking into account possible administrative problems and opportunities for cheating, it becomes unclear just what an appropriate system might look like, never mind what is "optimal." A well-designed deposit-refund instrument still offers an effective and economically attractive weapon for the regulatory arsenal. In the case of used oil, however, no deposit-refund system (or any other regulatory instrument) seems likely to withstand careful analysis because the risks posed by used oil are simply too low to justify the expense.

Chapter 8

SUGGESTIONS FOR FUTURE RESEARCH

Economists frequently have argued that incentive-based instruments offer better means of reducing environmental pollution than traditional standard-setting exercises. However, incentive instruments have made far more inroads in the economics and policy-analysis literature than they have in practice.

For several reasons, tax-subsidy instruments may be less threatening to the established order than other incentive schemes. They need not be viewed as replacements for standards, but rather as supplements intended to aid regulatory enforcement. To the extent that standards raise the cost of doing right and thereby risk driving at least some law-abiding firms out of the regulated system, tax-subsidy instruments may lessen or eliminate these perverse incentives. The subsidy component creates incentives for self-identification, thereby overcoming an important and persistent barrier to effective regulation. The tax component, if applied to an input that is appropriately related to hazardous waste generation, sends valuable signals to other firms not covered by the regulatory program. Higher final goods prices send similar messages to consumers. One particularly attractive feature is the way that tax-subsidy instruments use the price system to penalize noncompliers. The market is likely to be a far more effective means of administering sanctions; it need not be concerned about the due process requirements that continually plague administrative and judicial penalty systems.

It is worth remembering that from the outset of our analysis we presumed the existence of regulated waste management system based on standards. The safe-disposal subsidy was predicated upon the prior identification of what constitutes safe disposal -- that is, a set of standards that define which forms of waste disposal are acceptable and which ones are not. Thus, a tax-subsidy instrument can be viewed as a means of achieving pre-defined standards -- an alternative, perhaps, to civil and criminal enforcement directed against all but the most egregious illegal dumpers. By substituting incentives for enforcement in this manner, enforcement resources could be directed instead toward better policing approved treatment, storage, and disposal facilities.

Our case study of used lubricating oil raises many of the issues involved in evaluating the potential merits of using tax-subsidy schemes to help regulate hazardous wastes. Used oil does not appear to be a particularly promising candidate for a tax-subsidy instrument (and probably less so for any other regulatory device), but the basic approach has sufficient promise that additional investigation is worthwhile. Future research should aim at enriching our understanding of the idea while at the same time discovering new (and perhaps superior) applications for it.

Within this research effort, certain issues deserve particular scrutiny. The issue of regulatory noncompliance has received scant attention. Analyses of regulatory proposals typically assume full compliance with the standards proposed without any consideration of the difficulties, efficacy, and potential side effects of enforcement. Transactions costs are routinely ignored; when they are included, they often grossly underestimate reality. Tax-subsidy instruments may prove to be most useful as complements to existing or proposed regulatory programs, in part because they need not threaten the established regulatory pattern, but also because they may enhance its effectiveness. Further research is also needed to better understand the kinds of conditions under which tax-subsidy instruments would perform best. To this end, additional case studies should be undertaken to provide the data needed to evaluate the more general applicability of such instruments.

Understanding Noncompliance

Little is known about noncompliance. The data are primarily anecdotal, which makes it difficult to estimate the magnitude of the problem. Further research should be undertaken to learn more about noncompliant behavior, to gain some understanding of its magnitude and the incentives for its occurrence. We should have realistic expectations about such efforts; the study of noncompliance, particularly the more sinister "black-market" varieties, may involve risks to researchers that rival the health hazards imposed on the public at large. Nonetheless, so little is known that even a little information would be a major step forward.

In the hazardous waste area, the black market poses especially difficult problems for regulatory design. Since the value of deterring black market transactions depends to a great extent on the relative risks posed by illegal waste disposal, research concerning these risks may be valuable in assessing the potential benefits of incentive programs intended to diminish their attractiveness.

This raises important issues concerning the practice of risk assessment, issues that we have alluded to on several occasions in this report but not addressed in any detail. We believe it is unwise to continue to rely on risk assessments that are imbued with excess conservatism; "worst case" analysis is a poor substitute for credible efforts to estimate the *likely* risks posed by hazardous wastes, even illegal dumping. Unfortunately, because the risks from illegal dumping stand to be large relative to those created by other methods of disposal, the temptation to use "worst case" analysis may be intensified. This temptation should be resisted, for the cause of better environmental risk management is poorly served by alarmist rhetoric masquerading as analysis.

Moreover, as our analysis of used oil clearly indicates, there is a price to be paid for "aiming too high." Overly conservative risk assessments lead to exaggeration of optimal tax and subsidy rates. Setting these rates at too high a level results in new behavioral distortions and concomitant social costs. One advantage of the tax-subsidy instrument is that it may put pressure on regulators to adopt less conservative risk assessment methods; the magnitude of safe-disposal subsidies they have to justify will reflect whatever risks they estimate.

The Importance of Transactions Costs

As our case study revealed, a tax-subsidy instrument might make sense for used lubricating oil were it not for the burden of transactions costs. This burden was relatively large primarily because of two interacting factors. First, the risks posed by used lubricating oil appear to be low, particularly in relation to the volume of waste. This means that any regulatory effort, regardless of its design, will encounter a severe handicap insofar as the costs of achieving any desired improvement may well exceed the net external benefits policy makers hope to obtain.

Second, these transactions costs apply to *all* units of used oil reclaimed for use by the secondary market, not just the new units brought into the system by the incentive. Because so much oil is already reclaimed as the result of unfettered market conditions, the marginal cost of implementing a tax-subsidy system includes increasing private disposal costs over an enormous *inframarginal* base. This problem is hardly unique to tax-subsidy instruments, of course; any regulatory method aimed at the margin will impose new costs on the units that precede it unless the margin itself can be accurately located. But if this were truly feasible, then the particular identification problems that motivated the tax-subsidy instrument would not exist and simpler regulatory approaches would suffice.

The importance of transactions costs in general have received short shrift in policy analysis and (especially) policy development. It has been far too easy to simply ignore them or systematically understate their significance. In our used oil case study, transactions costs were treated as an exogenous parameter because we were unable to credibly specify the kinds of additional requirements that might be impose on subsidy recipients. However, the results we obtained suggest the folly of maintaining the fiction that because transactions costs are hard to estimate, they can be summarily ignored.

Conditions for Efficiency and Effectiveness of Tax-Subsidy Instruments

In our used oil case study, four parameters stood out as the most important determinants of the net benefit of a tax-subsidy instrument.

First, the noncompliance problem must either be significant ex ante, or likely to become significant upon the adoption of a particular set of regulations. If noncompliance is not of much concern, the need for a tax-subsidy instrument is far from evident. The safe-disposal subsidy would have no effect on disposal choice, and the input tax would induce no additional waste minimization that could not be more readily (and accurately) achieved through a waste-end tax.

Second, the degree of price-responsiveness matters a lot. Even if significant noncompliance exists, a simple waste-end tax will be sufficient to motivate any additional waste minimization policy makers desire, *provided that* firms already in the system are unresponsive to price changes. Conversely, a safe-disposal subsidy offers little prospect of inducing desirable changes in disposal practices if noncompliers are unresponsive to the cash cost of safe disposal. Price-responsiveness is essential to obtain the efficiency-enhancing benefits of any incentive instrument; otherwise, these instruments simply provide a relatively efficient means of raising government revenue and transferring wealth -- objectives that may have their merits but are unrelated to the task of environmental risk management and can be achieved more efficiently other ways.

Third, the relative risks posed by alternative waste disposal methods determine how important it is to deter noncompliance. The greater the risks posed by unsafe disposal, the greater will be the net benefit of inducing noncompliers to change their ways. In the used oil case, these risks were relatively low, which made optimal subsidy rates correspondingly small, especially compared to transactions costs. For other waste streams, risks can be expected to be much greater.

Finally, the level of additional transactions costs needed to effectuate the tax-subsidy program greatly affects its potential attractiveness. Of course, this conclusion applies as well to other regulatory approaches, whether based on economic incentives or standards. It will be less important for a tax-subsidy instrument to the extent that regulators allow market forces to work without undue hindrance in the form of eligibility requirements, documentation, and identification.

The relationship among these parameters is unclear after just a single case study. Furthermore, the ways in which they interact will vary from case to case. Thus, we cannot provide a general relationship beyond the intuitively reasonable but vague conclusions already indicated. Future research should seek to clarify this relationship in ways that enable policy makers and regulators to identify suitable applications from among the vast field of available candidates.

Additional Case Studies

Tax-subsidy instruments are not a panacea for hazardous waste regulation. Indeed, our case study of used lubricating oil indicates that unless transactions costs can be kept very low, such a regime may not appreciably improve upon the status quo, at least in terms of human health risks. More case studies need to be undertaken to see how well the instrument might perform, particularly in cases where disposal risks are relatively high. Our methodology is by no means the last word on the subject, either, although it does respect the uncertainties in the data and offer plausible projections of costs and benefits. Should a tax-subsidy instrument be devised for a specific hazardous waste stream, effort should be devoted to the development of an empirical methodology suitable for evaluating the program after its inception. Besides the generic value of program evaluation, an incentive instrument such as this warrants special attention because policy makers may want to adjust its incentive features periodically so as to ensure that policy objectives are being appropriately addressed. Indeed, the data collected in the evaluation effort may be superior in quality to what had to be used to design the program. Evaluation results also would help test the validity of the model used to project expected benefits and costs, thereby completing the loop between policy design and implementation.

REFERENCES

Abt Associates 1985. Ruder, Eric, Richard Wells, Michael Battaglia, and Richard Anderson. *National Small Quantity Hazardous Waste Generator Survey,* (Springfield, Va.: NTIS PB85-180438).

Allen, R. D. G. 1938. *Mathematical Analysis for Economists,* (New York: St. Martin's Press)

Bailey, Martin J. 1980. *Reducing Risks to Life,* (Washington, D.C.: American Enterprise Institute).

Baumol, William J. and Wallace E. Oates. 1979. *Economics, Environmental Policy, and the Quality of Life,* (Englewood Cliffs, N.J.: Prentice-Hall).

Becker, Gary S. 1968. "Crime and Punishment: An Economic Approach," *Journal of Political Economy,* vol. 76, no. 2 (March/April), pp. 169-217.

Block, Alan A., and Frank R. Scarpitti. 1985. *Poisoning for Profit,* (New York: Morrow).

Bohm, Peter. 1981. *Deposit-Refund Systems,* (Baltimore, Md.: Johns Hopkins University Press for Resources for the Future).

Bohm, Peter, and Clifford S. Russell. 1985. "Comparative Policy Analysis of Alternative Policy Instruments," Chapter 10 in *Handbook of Natural Resource and Energy Economics, Volume I,* (Kneese and Sweeney, eds.), (Amsterdam: Elsevier Science Publishers B.V.).

Boston Globe 1987. "Firm Agrees to $400m PCB Cleanup," Boston *Globe,* November 10, 1987, p. 3.

Bradford, David F. 1986. *Untangling the Income Tax,* (Cambridge, Mass.: Harvard University Press for the Committee for Economic Development).

Brock, William A., and David S. Evans. 1985. "The Economics of Regulatory Tiering," *Rand Journal of Economics,*, vol. 16, no. 3 (Autumn), pp. 398-409.

Browning, E. 1976. "The Marginal Cost of Public Funds," *Journal of Political Economy,* vol. 84, pp. 283-298.

California Task Force 1986a. Governor's Task Force on Toxics, Waste and Technology. *Final Report: Volume 1,* (Sacramento, Calif.: Department of Health Services, State of California).

_____.1986b. Governor's Task Force on Toxics, Waste and Technology. *Final Report: Volume 2, Technical Advisory Committee Reports,* (Sacramento, Calif.: Department of Health Services, State of California).

Carnegie Mellon. 1988. *Regulation and Disposal of Used Oil,* (Pittsburgh: Department of Engineering and Public Policy, Carnegie-Mellon University).

CBO 1985. United States Congress, Congressional Budget Office. *Hazardous Waste Management: Recent Changes and Policy Alternatives,* (Washington, D.C.: Congressional Budget Office).

CEM 1985. Center for Environmental Management. *Waste Reduction: The Untold Story,* (Medford, Mass.: Environmental Management Center, Tufts University).

______. 1986. Center for Environmental Management. *Waste Reduction: The Ongoing Saga,* (Medford, Mass.: Environmental Management Center, Tufts University).

______. 1987. Center for Environmental Management. *Waste Reduction: The Hurdles Ahead,* (Medford, Mass.: Environmental Management Center, Tufts University).

CMA 1987. Chemical Manufacturer's Association. *1985 CMA Hazardous Waste Survey,* (Washington, D.C.: Chemical Manufacturer's Association).

Connolly, Dennis R. 1987. "Insurance: The Case of the Slain Messenger," *Toxics Law Reporter,* vol. 1 (April 15), pp. 1280-1287.

CRS 1987. United States Congress, Library of Congress, Congressional Research Service. *Hazardous Waste Fact Book,* (Washington, D.C.: Congressional Research Service).

Downing, Paul B., and William D. Watson, Jr. 1973. *Enforcement Economics in Air Pollution Control,* (Washington, D.C.: USEPA, 600/5-73-014).

______. 1974. "The Economics of Enforcing Air Pollution Controls," *Journal of Environmental Economics and Management,* vol. 1, pp. 219-236.

______. 1975. "Cost-Effective Enforcement of Environmental Standards," *Journal of the Air Pollution Control Association,* vol. 25, pp. 705-710.

Drayton, William. 1984. *America's Toxic Protection Gap: The Collapse of Compliance with the Nation's Toxics* Laws, (Washington, D.C.: Environmental Safety).

Environmental Forum 1984. "A Criminal Sentencing: Private Lives Which 'Merit Emulation,'. . .But 'Extremely Serious Offenses' Against Environment," *Environmental Forum,* (October), p. 8.

EPA 1986a. United States Environmental Protection Agency, Office of Solid Waste. *Report to Congress: Minimization of Hazardous Waste,* (Washington, D.C.: USEPA, EPA/530-SW-86-033).

______. 1986b. United States Environmental Protection Agency, Office of Solid Waste. *Report to Congress: Minimization of Hazardous Waste (Appendices),* (Washington, D.C.: USEPA, EPA/530-SW-86-033).

______. 1986c. United States Environmental Protection Agency, Office of Water Regulations and Standards. *Report to Congress on the Discharge of Hazardous Wastes to Publicly Owned Treatment Works,* (Washington, D.C.: USEPA, EPA/530-SW-86-004; NTIS, PB86-184017).

______. 1987. United States Environmental Protection Agency, Office of Policy, Planning and Evaluation, Office of Policy Analysis; *Unfinished Business: A Comparative Assessment of Environmental Problems (Overview Report),* (Washington D.C.: Office of Policy Analysis).

Epstein, Samuel S., Lester O. Brown, and Carl Pope. 1982. *Hazardous Waste in America,* (San Francisco: Sierra Club Books).

Evans, David S. 1986. "The Differential Effect of Regulation Across Plant Size: Comment on Pashigian," *Journal of Law and Economics,* vol. 29, no. 1, pp. 187-200.

Fadil, Frank A.. 1985. *How Fast Is Quick? The Quick Oil Change and Lubrication Industry,* (Fairfield, N.J.: Charles H. Kline and Company, Inc.)

Feldstein, Martin. 1978. "The Welfare Cost of Capital Income Taxation," *Journal of Political Economy,* vol. 86, pp. 529-553.

Fisher, Anthony C. 1981. *Resource and Environmental Economics,* (Cambridge: Cambridge University Press).

Fortuna, Richard C., and David J. Lennett. 1986. Hazardous Waste Regulation: The New Era, (New York: McGraw-Hill).

Franklin Associates. 1983. *Management and Composition of Used Oil in the United States: Draft Report,* (Prairie Village, Kans.: Franklin Associates).

_____. 1984. *Management and Composition of Used Oil* in *the United States,* (Prairie Village, Kans.: Franklin Associates).

Friedman, Lee S. 1984. *Microeconomic Policy Analysis,* (New York: McGraw-Hill).

GAO 1983a. United States *Congress,* General Accounting Office. *Information on Disposal Practices of Generators of Small Quantities of Hazardous Waste,* (Washington, D.C.: USGAO, GAO/RCED-83-200).

_____. 1983b. United States Congress, General Accounting Office. *Wastewater Dischargers Are Not Complying With EPA Pollution Control Permits,* (Washington, D.C.: USGAO, GAO/RCED-84-53).

_____. 1985. United States Congress, General Accounting Office. *Illegal Disposal of Hazardous Waste: Difficult to Detect or Deter,* (Washington, D.C.: USGAO, GAO/RCED-85-2).

Garelick, Barry. 1987. "EPA's Definition of Solid Waste: Making Distinctions Between Shades of Gray," *Environmental Law Reporter,* vol. 17, pp. 10349-10352.

Goldman, Benjamin A., James A. Hulme, and Cameron Johnson. 1986. *Haz*ardous *Waste Management: Reducing the Risk,* (Washington, D.C.: Island Press for the Council on Economic Priorities).

Haigh, John A., David Harrison, Jr., and Albert L. Nichols. 1984. "Benefit-Cost Analysis of Environmental Regulation: Case Studies of Hazardous Air Pollutants," *Harvard Environmental Law Review,* vol. 8, no. 2, pp. 395-434.

Hammitt, James K., and Peter Reuter. *[forthcoming]. Deterring Illegal Disposal of Hazardous Wastes: Policy Lessons From Enforcement in Selected Jurisdictions,* (Santa Monica, Calif.: Rand Corporation).

Harford, John D. 1978. "Firm Behavior Under Imperfectly Enforceable Pollution Standards and Taxes, " *Journal of Environmental Economics and Management,* vol. 5, no. 1, pp. 26-43.

Harris, Christopher, William L. Want, and Morris A. Ward. 1987. *Hazardous Waste: Confronting the Challenge,* (Westport, Conn.: Quorum Books).

ICF Consulting Associates, Inc. *1985. Economic Incentives for the Reduction of Hazardous Wastes,* (Sacramento, Calif.: Alternative Technology and Policy Development Section, Toxic Substances Control Division, Department of Health Services, State of California).

Kneese, Allen V., and Charles L. Schultze. 1975. *Pollution, Prices, and Public Policy,* (Washington, D.C.: Brookings).

Law Enforcement Hearings 1984. United States Congress, House of Representatives, Committee on Energy and Commerce, Subcommittee on Oversight and Investigations. *EPA'S Law Enforcement Authority: Hearings, December 19, 1983,* (Washington, D.C.: USGPO).

Leonard, Herman B. 1986. *Checks Unbalanced: The Quiet Side of Public Spending,* (New York: Basic Books).

Leone, Robert A. 1986. *Who Profits: Winners, Losers, and Government Regulation,* (New York: Basic Books).

Linder, Stephen H., and Mark E. McBride. 1984. "Enforcement Costs and Regulatory Reform: The Agency and Firm Response," *Journal of Environmental Economics and Management,* vol. 11, no. 4, pp. 327-346.

Lindley, D. V., and W. F. Scott. 1984. *New Cambridge Elementary Statistical Tables,* (Cambridge: Cambridge University Press).

Lipsey, Richard G., and Kelvin Lancaster. 1956. "The General Theory of the Second Best," *Review of Economic Studies,* vol. 24, pp. 11-32.

Massachusetts Department of Environmental Management 1987. Commonwealth of Massachusetts, Department of Environmental Management, Office of Safe Waste Management. *Hazardous Waste Management in Massachusetts: 1987 Statewide Environmental Impact Report,* (Boston: Massachusetts Department of Environmental Management).

National Research Council 1985. Committee on Institutional Considerations in Reducing the Generation of Hazardous Wastes, National Research Council. *Reducing Hazardous Waste Generation,* (Washington, D.C.: National Academy Press).

New England Waste Management Officials' Association. 1988. *Used Oil in New England,* (Boston, Mass.: NEWMOA).

New York State Legislative Commission 1986. New York State Legislative Commission on Toxic Substances and Hazardous Wastes. *Used -- But Useful: A Review of the Used Oil Management Program in New York State,* (Albany, N.Y.: New York State Legislative Commission on Toxic Substances and Hazardous Wastes).

_____ . 1987. New York State Legislative Commission on Toxic Substances and Hazardous Wastes. *Hazardous Waste Reduction: Obstacles and Incentives,* (Albany, N.Y.: New York State Legislative Commission on Toxic Substances and Hazardous Wastes).

Nichols, Albert L. 1984. *Targeting Economic Incentives for Environmental Protection,* (Cambridge, Mass.: MIT Press).

Nichols, Albert L., and Richard J. Zeckhauser. 1986. "The Perils of Prudence: How Conservative Risk Assessments Distort Regulation," *Regulation,* vol. 10, no. 2, (Nov/Dec), pp. 13-24.

Nichols, Elizabeth, and Aaron Wildavsky. 1987. "Nuclear Power Regulation: Seeking Safety, Doing Harm?," *Regulation,* vol. 11, no. 1, pp. 45-53.

OTA 1983. United States Congress, Office of Technology Assessment, *Technologies and Management Strategies for Hazardous Waste Control,* (Washington, D.C.: USGPO, OTA-M-196).

_____. 1986. United States Congress, Office of Technology Assessment, *Serious Reduction of Hazardous Waste: For Pollution Prevention and Industrial Efficiency,* (Washington, D.C.: USGPO, OTA-ITE-317).

_____. 1987. United States Congress Office of Technology Assessment. *From Pollution to Prevention: A Progress Report on Waste Reduction,* (Washington, D.C.: USGPO, OTA-ITE-347).

Pashigian, B. Peter. 1984. "The Effect of Environmental Regulation on Optimal Plant Size and Factor Shares," *Journal of Law and Economics,* vol. 26, p. 1.

Reuter, Peter. 1984. "Regulating Rackets," *Regulation,* (Sept/Dec), pp. 29-36.

Riesel, Daniel. 1985. "Criminal Prosecution and Defense of Environmental Wrongs," *Environmental Law Reporter,* vol. 15, pp. 10065-10081.

Russell, Clifford S., Winston Harrington, and William J. Vaughan. 1986. *Enforcing Pollution Control Laws,* (Washington, D.C.: Resources for the Future).

Schelling, Thomas C. (ed.) 1983. *Incentives for Environmental Protection,* (Cambridge, Mass.: MIT Press).

Schwartz, Seymour I., Daniel R. Deonegan, Nancy Steffenson Ostrom, Tony Emmert, and Debbie Sivas. 1985. *Managing the Electronics Industry's Hazardous Wastes: Technology and Economics of Alternatives to Land Disposal,* (Davis, Calif.: Division of Environmental Studies, University of California at Davis).

Schwartz, Seymour I., Wendy Pratt Cuckovich, and Nancy Steffenson Ostrom. 1987. *Managing Hazardous Wastes Produced by Small Quantity Generators,* (Sacramento, Calif.: Senate Office of Research, California State Legislature).

Schultze,,Charles L. 1977. *The Public Use of Private Interest,* (Washington, D.C.: Brookings).

Siegel, Andrew, and David Salvesen. 1987. "'Sterling v. Velsicol': The Case for a New Increased Risk Rule," *Environmental Law Reporter,* vol. 17 (May), pp. 10155-10161.

Sobotka and Co., Inc. 1981. *An Assessment of Nonregulatory Approaches for Encouraging the Recycling and Proper Use of Waste Lube Oil,* (Washington, D.C.: Sobotka and Co.)

SQG Hearings 1983. United States Congress, House of Representatives, Committee on Small Business, Subcommittee on Energy, Environment, and Safety Issues Affecting Small Business. *Hazardous Waste Control and Enforcement Act, H.R. 2867 (Part 1--Small Quantity Generators): Hearings, July 26, 1983,* (Washington, D.C.: USGPO).

Stigler, George J. 1970. "The Optimum Enforcement of Laws," *Journal of Political Economy,* vol. 78, pp. 526-536.

Storey, D. J., and P. J. McCabe. 1980: "The Criminal Waste Discharger," *Scottish Journal of Political Economy,* vol. 27, no. 1, pp. 30-40.

Sullivan, Arthur H. 1987. "Policy Options for Toxics Disposal: Laissez-Faire, Subsidization, and Enforcement," *Journal of Environmental Economics and Management,* vol. 14, pp. 58-71.

Temple, Barker and Sloane. 1985a. *Regulatory Impact Analysis* of *Proposed Standards for the Management of Used Oil,* (Lexington, Mass.: Temple, Barker and Sloane).

_____. 1985b. *Regulatory Impact Analysis of Proposed Standards for the Management of Used Oil: Background Document,* (Lexington, Mass.: Temple, Barker and Sloane).

_____. 1987a. *DIY Issue Analysis,* (Washington, D.C.: Temple, Barker and Sloane).

_____. 1987b. *Revised Baseline Flow Data for Used Oil Modeling,* (Washington, D.C.: Temple, Barker and Sloane).

Tenusak, Jackie, and Paul E. Bailey. 1987. "An Overview of Liability Insurance Under, RCRA," *Proceedings of the National Conference on Hazardous Wastes and Hazardous Materials, March 16-18, 1987,* (Silver Spring, Md.: Hazardous Materials Control Research Institute).

Terkla, D. G. 1979. *The Revenue Capacity of Effluent Charges,* Ph.D. Dissertation, University of California at Berkeley.

Tietenberg, T. H. 1985. *Emissions Trading,* (Washington, D.C.: Resources for the Future).

Vaughan, William J., and Clifford S. Russell. 1983. "Monitoring Point Sources of Pollution: Answers and More Questions from Statistical Quality Control;" *The American Statistician,* vol. 37, pp. 476-487.

Versar, Inc. 1986a. *Do-It-Yourself Used Oil Recycling Programs in the United States,* (Springfield, Va.: Versar).

_____.1986b. *Waste Minimization Issues and Options: Volume I,* (Washington, D.C.: USEPA, EPA/530-SW-86-041)

_____.1986c. *Waste Minimization Issues and Options: Volume III,* (Washington, D.C.: USEPA, EPA/530-SW-86-042)

_____ .1986d. *Waste Minimization* Issues *and Options: Volume III,* (Washington, D.C .: USEPA, EPA/530-SW-86-043)

Viscusi., W. Kip, and Richard J. Zeckhauser. 1979. "Optimal Standards with Incomplete Enforcement," *Public Policy,* vol. 27 (Fall), pp. 437-456.

Ward, Bud. 1983. "Criminal Enforcement: Is There Any?" *Environmental Forum,* (October), pp. 6-9.

Watson, William D., Jr., and Paul B. Downing. 1976. "Enforcement of Environmental Standards and the Central Limit Theorem," *Journal of the American Statistical Association,* vol. 71, no. 355 (September), pp. 567-574.