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I am submitting herewith a dissertation written by Thomas Robert Sadler entitled "The Case for Risk-Based Environmental Taxation." I have examined the final copy of this dissertation for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Doctor of Philosophy, with a major in Economics.

Robert Bohm, Major Professor

We have read this dissertation
and recommend its acceptance:

Accepted for the Council:

Associate Vice Chancellor and
Dean of The Graduate School

The Case for Risk-Based Environmental Taxation

A Dissertation
Presented for the
Doctor of Philosophy
Degree
The University of Tennessee, Knoxville

Thomas Robert Sadler
August 1998

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A dissertation evolves as ideas progress. In my case, the dissertation began during the second year of study in the Ph.D. program. As a research assistant at the Energy, Environment, and Resources Center, I worked under the guidance of Professor Robert A. Bohm, who hired me to research the broad topic of environmental taxation. During the course of the next two and a half years, we read many papers, discussed the current literature and developed ideas. A dissertation on risk-based regulation evolved from many conversations on environmental taxation.

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circumstances in graduate school or as a new Assistant Professor, Holly has been my closest friend. Her encouragement has created a wonderful environment for learning and growth. I cherish our relationship, as it keeps the important aspects of life in perspective.

ABSTRACT

Risk-based environmental taxation offers a new method to regulate chemical emissions. The risk-based levy is an emissions tax that uses an approximation of the damage function. To formulate tax rates, an index of "risk values" first takes into account human health effects, environmental effects and the exposure potential for target pollutants. Then, by using a common tax base (the risk unit) that reflects relative risk values, and determining the number of risk units for each chemical emission, the risk-based system taxes according to approximate pollution damage. The use of the risk unit allows a policy maker to regulate a number of chemical emissions and implement a tax rate on one risk unit, which then translates a different rate to each pollutant. As a result, relatively higher tax rates provide industry with the incentive to reduce the emissions of the most harmful chemical pollutants.

In a first-best world, economic theory prescribes, for pollution reduction and efficiency gains, an environmental tax above the marginal cost of abatement (MCA). But this paper admits that the MCA is unknown for target firms. In addition, information restrictions limit the ability to prescribe fully informed policy. The inclusion of a number of pollutants, information restrictions with marginal abatement costs and damage functions, and the admission of pre-existing tax distortions in the economy create an nth best pollution problem. As a result, the risk-based regulation recognizes, under plausible circumstances, the inability of the policy to reach an optimal solution. To levy taxes, the risk-based regulatory framework focuses on the relative damage values of individual pollutants. The system uses the chemical ranking of the Center of Clean Products and Clean Technology at The University of Tennessee. The risk-based system taxes, at the state level in the United States, chemical emissions in the Chemical and Allied Products Industry Group.

The risk-based system encompasses a number of pollutants under the jurisdiction of one policy, levies one tax rate on a risk unit which translates to a different rate for each target emission, provides incentive for firms to reduce the emissions of the most harmful target emissions, focuses on approximate marginal damage, and establishes logic for environmental regulation.

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CHAPTER 1

INTRODUCTION TO RISK-BASED TAXATION

This dissertation develops a risk-based environmental tax policy for chemical emissions. Higher tax rates apply to the pollutants that pose a greater degree of risk to humans and the environment. This system assigns individual chemical emissions a certain number of "risk units," which reflect the pollutant's damage value. The damage value takes into account human health effects, environmental effects and exposure potential. The system levies a tax on one risk unit that translates, according to the quantity of risk units, a different rate per pound of each chemical emission.

Risk-based taxation serves as a proxy for a theoretical system of fully informed emissions taxes. The risk-based levy is an emissions tax that uses an approximation of the damage function. To set pollution prices, this system first looks to damage values to provide information concerning marginal damage for a number of chemical emissions. The tax then embeds an economic valuation of an approximation of marginal damage into production cost, creating a chemical emission price for polluting firms. The goal of the risk-based system is to narrow the gap between the marginal private and marginal social cost that results from the chemical emission externality.

The development of the risk-based policy stems from the problems associated with finding information about (1) the optimal level of pollution reduction for target emissions, and (2) a common measure of damage to compare the chemicals. The difficulty in finding the optimal level of abatement is well documented [Goulder, Parry and Burtraw (1996), Turner, Pearce, and Bateman (1993), Baumol and Oates (1988), and Fishelson (1976)]. With the second point,

researchers have not provided, in a damage function approach,[1] the link between the target[2] chemical emissions and environmental damage.[3] It would therefore be difficult, in the absence of an index of damage values, to tax the pollutants in a consistent manner. As a result, a relative index provides the framework for risk-based taxation:[4]

> *In reality, then, accurate calculation of an optimal pollution tax level is likely to be an unrealistic goal. The best we can hope for is to determine an acceptable compromise in the face of imperfect information. What may be more feasible is to calculate the relative magnitudes between a tax on one pollutant and a tax on a second by comparing the levels of damage which each causes.* (Turner, Pearce and Bateman, 1993, 171)

The identification of relative damage between pollutants, the incorporation of this information into a policy framework and the calculation of the tax rate to regulate target chemical emissions serve as the focal points of this paper. This dissertation concludes that, when a number of chemical emissions are regulated, marginal damages increase at a relatively stable rate and the impact of pollution synergy does not render the proposed damage values ineffective, the risk-based levy represents a viable proxy for a fully informed emissions tax.[5]

The development of the risk-based system contributes to the environmental tax literature in four ways. First, the use of the risk unit facilitates the regulation of different pollutants under the jurisdiction of one policy. As a result, the system encompasses a *set* of chemical emissions. Aside from important research and enforcement costs, this framework eases administrative duty.

Second, from the standpoint of the regulator, an underlying concern with environmental tax policy entails the inability to identify marginal abatement costs for individual firms (Adar and

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[1] The damage function approach uses a consistent method to value environmental damage. This approach is explained in chapter three.
[2] The risk-based tax policy targets 158 chemical emissions.
[3] The term environmental damage could have many different meanings. Within the framework of risk-based taxation, environmental damage includes the human health effects, environmental effects, and exposure potential of individual pollutants.
[4] The risk-based tax system incorporates the relative damage values of different chemical emissions from an index created by the Center for Clean Products and Clean Technology at The University of Tennessee.
[5] Information limitations restrict a full assessment of the welfare impact of a system of risk-based taxation. Chapters three and four use available data and information to evaluate the risk-based tax system.

Griffin, 1976). Economic theory, however, prescribes that tax rates exceed marginal abatement cost for emission reduction. In the absence of this information, the risk tax uses the index of relative damage values, levies taxes on risk units, and functions as a damages-based levy.

Third, this paper explains when the risk-based levy serves as a good proxy for a theoretical fully informed emissions tax and when it does not. In effect, when the link between emissions and damage is sufficiently understood, and an accurate economic valuation of pollution damage exists, a fully informed emissions tax would serve as the appropriate policy. In the absence of this information, risk-based taxation uses damage information from a toxicity index to approximate a fully informed levy. The degree of accuracy of the risk-based system depends on the size of the estimation error of marginal damage when compared to a fully informed tax. Whether the benefits of the risk-based system exceed the costs is an important issue considered below.

Fourth, this dissertation considers the choice of the risk-based environmental tax in the context of the portfolio problem. This problem occurs when regulators must choose between different evaluation criteria in order to implement policy. The application of environmental taxation, for instance, requires the creation of administrative and compliance costs. The economic value of these and other costs must be weighed against all benefits, including gains from improvements in environmental quality. Therefore, an evaluation of the "effectiveness" of tax implementation must include all potential policy costs and benefits.

Since taxation serves as the relevant policy instrument, chapter one discusses this form of regulation, evaluates its merits and drawbacks, and contrasts it to other environmental policies, particularly fully informed and "simple" emission levies. Chapter one also explains the role of risk in environmental management and discusses criteria to use to evaluate environmental taxes.

LARGE NUMBER POLLUTION PROBLEMS

One type of market failure occurs in the economy when private agents impose unintended or external costs, in the form of chemical emissions, on others. In the absence of public intervention, these negative externalities persist since prices reflect private costs to the firm, but do not include pollution's cost to society. A policy maker may implement regulation that accounts for the additional social cost by effectively pricing pollution in one of several ways.

Individuals, of course, may resolve certain externality problems without public intervention. The small-numbers case occurs, for example, when a person convinces a neighbor to stop playing loud music. If left alone, the agents in the externality dispute negotiate a suitable solution. The key in this instance is the ability of the parties to negotiate in the presence of low transaction costs (Coase, 1960).

In situations with a large number of polluters and/or victims, a collective decision-making body, i.e. a government, must intervene when information requirements prohibit individual agents from negotiating a solution. In the United States, many polluters in each state emit hundreds of chemical emissions. The relevant policy issue involves large-number-pollution-problems with multiple polluters of chemical emissions. As Davis *et al.* explain,

> *The hazardous waste and many of the persistent air and water pollution problems are primarily toxic chemical problems. Widespread use of toxic chemicals in all segments of industry and commerce has created the need to deal with burgeoning wastestreams containing toxic chemicals emitted into the air and water and buried in the soil. Two decades of pollution control regulations have not sufficiently reduced environmental releases of toxic chemicals to acceptable levels.* (1994a, ES-1)

Many firms use toxic chemicals in the manufacturing process. Market failure occurs when these companies do not take into account the additional social costs imposed on society through waste disposal. Different environmental policies could address this form of market failure in the United States.

ALTERNATIVE ENVIRONMENTAL POLICY INSTRUMENTS

In 1986 the EPA established legislation that requires individual firms to report quantities of chemical emissions to the government. Under the Emergency Planning and Community Right-To-Know Act (EPCRA), businesses must report the quantities and locations of on-site chemicals to local and state governments. Section 313 of EPCRA, in particular, requires manufacturers in Standard Industrial Classification (SIC) codes 20-39 to report the emissions of over 600 toxic chemicals into the air, water, and land. As a result of the legislation, individual firms are required by law to monitor the emissions of the target chemicals, which are tabulated by the EPA in the Toxic Release Inventory (TRI), and available to the public. The idea is to inform local citizens of the chemical emissions in the vicinity. With this information, concerned residents may apply pressure on local companies to reduce the emissions of harmful pollutants.

This paper addresses the regulation of current chemical emissions in the TRI, not previous releases that may persist in the environment. A question of the degree of regulatory control arises when considering the proper environmental policy to reduce the flow of these pollutants. Two general policy options exist that employ different levels of government control. First, direct (command-and-control) regulation forces firms to bear equal cost in pollution control, regardless of the relative costs of abatement. Performance standards establish a target pollution level, but do not prescribe the method to reach the target. Technology standards, in contrast, specify the method and/or technology that the firm must use to comply with regulation. Often direct policy requires the "best available technology," but does not specify abatement equipment. The 1990 Amendments to the Clean Air Act employ the best available technology.

Second, indirect (incentive) regulation encourages pollution reduction, but allows flexibility in the abatement process. This category of environmental policy makes the incentives of polluters correspond to society's best interest. Incentive policies include taxes, marketable permits, deposit-refund systems, performance bonding and liability systems.

When Direct Regulation Is Appropriate

In general, command-and-control regulation serves as an appropriate policy when (1) high monitoring costs exist, (2) society requires a very low level of emissions for a particular pollutant, and/or (3) some event creates a new relationship between emissions and damage, such as smog establishing transportation requirements (Kahn, 1998, 68). With certain highly toxic chemicals from the TRI, the optimal level of emissions may be close or equal to zero from society's standpoint. This prescription is based on the fact that the marginal damages per unit of certain chemical emissions may be high due to a tendency of many of these pollutants to persist for long periods of time in the environment. In addition, a small level of exposure from pollutants such as arsenic, terbufos, and formaldehyde, may kill small organisms, increase the risk of cancer in humans, or destroy exposed ecosystems.

With these highly toxic emissions, marginal damages may exceed marginal abatement costs over the entire range of emissions. Even if marginal abatement costs are greater than marginal damages at very low emission levels, direct regulation may still be cost effective. In this latter situation social costs would accrue to society with emission reduction to zero when optimal emissions are positive. But these social costs may be lower than the administrative and enforcement costs of implementing a new policy for regulation.

The production and use of certain chemicals is already banned or severely restricted by law. According to the Office of Pollution Prevention and Toxics of the EPA,[6] a banned industrial or consumer chemical is one that has, "for health or environmental reasons, been prohibited for all uses by final governmental regulatory action or that has been refused first time use by final regulatory action, for health or environmental reasons." A severely restricted industrial or consumer chemical is a substance "which, for health or environmental reasons, virtually all uses have been prohibited nationally by final government regulatory action, but for which certain specific uses remain authorized...Also, severely restricted chemicals have been withdrawn by the industry...from the market...where there is clear evidence that such action has been taken for health or environmental reasons." Currently the only chemicals that are banned in the US, according to the Office for Pollution Prevention and Toxics, are polychlorinated biphenyls (PCBs). Many other chemicals, however, are severely restricted. Table 1 lists the thirty chemicals that are either banned or severely restricted.[7]

The TRI includes the following seven chemicals from Table 1: carbon tetrachloride, polychlorinated biphenyls, hexamethylphosphoramide, pentachloroethane, urethane, tris (2,3-dibromopropyl) phosphate and asbestos. Manufacturers in the United States emit some of these, and many other, highly toxic chemicals. Table 2 summarizes the pounds of releases in 1994 from the TRI of these seven chemicals and the number of companies that emitted the pollutants. The TRI does not report emissions of PCBs, hexamethylphosphoramide, or tris (2,3-dibromopropyl) phosphate in 1994. As a result of their potential harm to humans and the environment, and high levels of damage per unit, it may be cost effective to ban the most toxic chemical emissions in the TRI.

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[6] This office is on the internet at www.epa.gov.

[7] The 1990 and 1992 amendments to the Montreal Protocal severely restrict, but do not outright ban, the CFCs and halons in Table 1. See the website at www.nwi.org/Treaties/montreal.html.

Table 1
US banned or severely restricted industrial or consumer chemicals

Chemical	Chemical	Chemical
1. CFC-11	11. CFC-213	21. polychlorinated biphenyls
2. CFC-12	12. CFC-214	22. 1-chloro-2-bromoethane
3. CFC-13	13. CFC-215	23. hexamethylphosphoramide
4. CFC-111	14. CFC-216	24. methyl n-butyl ketone
5. CFC-112	15. CFC-217	25. pentachloroethane
6. CFC-113	16. halon 1211	26. urethane
7. CFC-114	17. halon 1301	27. tris(2,3-dibromopropyl) phosphate
8. CFC-115	18. halon 2402	28. polybrominated biphenyls
9. CFC-211	19. carbon tetrachloride	29. asbestos
10. CFC-212	20. methyl chloroform	30. isomers of chemicals 1-20, other than 1,1,2-trichloroethane

Source: Office of Pollution Prevention and Toxics, EPA

Table 2
Emissions of banned or severely restricted chemicals from 1994 in the TRI

Chemical	Emission pounds	Number of firms
carbon tetrachloride	632804	69
PCBs	0	0
hexamethylphosphoramide	0	0
pentachloroethane	1123	3
urethane	11050	4
tris (2,3-dibromopropyl) phosphate	0	0
asbestos	294368	85

Source: Toxics Release Inventory, EPA

Policy Options

If some of the most harmful TRI chemicals are to be banned, policy options for the entire set of pollutants include (1) banning a subset of the pollutants, (2) banning the entire set, (3) banning the most toxic and regulating the others with direct control, or (4) banning the most toxic and regulating other chemical emissions with incentive regulation. The first alternative seems plausible. In this situation the EPA could ban the use and manufacture of chemicals when the optimal level of emissions is zero or near zero. The policy could target the most harmful chemicals. This first option, however, would not encourage firms to reduce other chemical emissions.

The problem with the second option is that many chemicals in the TRI have low damage values: an outright ban would prevent the use of many substances among the 600 that pose little risk to humans and the environment. In addition, manufacturing industries in SIC 20-39 use the 600 chemicals as inputs and produce them as output (as is the case with SIC 28—Chemicals and Allied Products). Banning the *entire* set would create a situation where many industries would have to alter their production processes dramatically.

The third policy possibility raises three concerns about the cost of regulating a large number of chemical pollutants not banned with direct policy. First of all, direct controls do not encourage technological innovation beyond the target level of pollution. As Farmer, Kahn, and McDonald (1995) argue, firms may not choose to implement new abatement capital even if the investment leads to short-term profits. This hesitation occurs since other firms soon copy new technology. In this situation, the innovator's profits cease.

Second, with direct policy, performance and technology standards may force firms to implement expensive abatement measures. Table 3 shows, for example, that direct regulation increases costs more than incentive policy. The ratios of command-and-control cost to least-cost

policy exceed one, implying that direct regulation is more expensive. Since these studies do not evaluate chemical emissions, the research does not consider the benefits that would accrue to society from over-compliance. In the case of an incentive instrument such as environmental taxation, properly employed policy would encourage firms to reduce pollution to the point where the tax rate equals marginal abatement cost (Baumol and Oates, 1988). But since a per-unit tax would continue to price any remaining emissions, firms would have incentive for further pollution reduction. As a result, the polluters may want to continue to reduce emissions to avoid tax payment. In the context of the Carlin studies, this over-compliance resulting from incentive policy would lead to further benefits to society from a cleaner environment not accounted for in the research in Table 3. In addition, the installation and operating costs of abatement technology vary between firms. Firms, not the government, are in position to implement least-cost technology. Carlin (1992) mentions additional research on other forms of air and water pollution that pattern the results in Table 3.

Table 3
Quantitative studies of economic incentive savings—air pollution

Pollutants Controlled	Study, Year, and Source	Geographic Area	Command-and-Control (CAC) Approach	Ratio of CAC cost to least cost
Hydrocarbons	Maloney & Yandle (1984)	Domestic Dupont Plants	Uniform % reduction	4.15
Nitrogen dioxide	Seskin *et al.* (1983)	Chicago	Available control technologies	14.4
Nitrogen dioxide	Krupnick (1986)	Baltimore	Available control technologies	5.9
Sulfur dioxide	ICF Resources (1989)	United States	Uniform emission limit	5.0
Particulates	Spofford (1984)	Lower Delaware	Uniform % reduction	22.0
Airport noise	Harrison (1983)	United States	Mandatory abatement	1.72

Source: Carlin, Alan. 1992. *The United States Experience with Economic Incentives to Control Environmental Pollution*, USEPA, 230-R-92-001.

Third, when the government attempts to reduce pollution levels with direct regulation, high information costs prevail. The United States Environmental Protection Agency (1990) estimates that the government spends more than $350 million annually at all levels to comply with environmental regulations. In many cases governments must certify some form of abatement technology, which may be expensive.

A Role for Incentive Policy

As a result of the problems with the first three policies, a case may be made for the fourth policy option, with incentive policy regulating pollutants not in the subset of chemicals to be banned. Incentive instruments include taxes, marketable permits, deposit-refund systems, performance bonding and liability systems. Environmental taxes establish a pollution price. Levied on a per-unit basis according to emission levels, the increase in cost encourages firms to alter their production mix in order to stay competitive. In response to the policy, polluters may introduce more efficient, less polluting technology, reduce emissions, or substitute away from "dirty" inputs. In the absence of information for an optimal solution,[8] policy makers use taxes to create an intended response among polluters. Another appealing characteristic of environmental taxes is that they raise revenue over the life of the policy.

Since polluters emit large quantities of a number of chemicals, a tax system could be developed that takes into account the damage potential of each pollutant. The key is that polluters in SIC 20-39 are already required by law to monitor and report emission levels to the TRI every year. A fully informed tax policy would levy rates on each substance that reflect emission damage to humans and the environment. But, in the absence of information to establish definitive damage

[8] The "optimal" level of emission reduction occurs where the marginal damages of individual pollutants equal the marginal abatement costs.

values for each target pollutant, an approximation could be used. Per-unit emission taxes could be established to regulate the chemicals not banned through direct policy.

With marketable permits, the government defines property rights for environmental resources, creates a limited number of permits for the discharge of pollution, and distributes or sells them through some kind of auction. By limiting the number of permits to reflect a target level of waste, the government establishes a market for pollution. The market in turn creates an equilibrium price for all permits. Since polluters equalize marginal abatement costs, cost-minimization results (Baumol and Oates, 1988, 177). If the government or environmental groups want to reduce emissions in a target industry, they may purchase permits. Although it could raise revenue, a policy of marketable permits would not regulate a set of chemical emissions since it is designed for one pollutant. Taxation would serve as a better policy for a set of chemical emissions.

Deposit-refund programs give consumers money back when they return environmentally-harmful products. This regulation serves as a good incentive policy when high monitoring costs exist and the product causing the pollution externality is identifiable. Currently individuals in many states return soda cans to receive a deposit that they pay at purchase. Other products such as pesticides and batteries contain a variety of toxic substances that could be regulated with a deposit-refund system.

But in the case of high releases of chemicals, a deposit-refund program would be difficult to implement. A number of these substances are used in production processes that create different levels of economic output and waste output. There may not be a fixed relationship, however, between chemical inputs and output, as in the case with batteries or soda cans. A properly designed policy to regulate polluters with large emission levels could establish a payment per unit of each chemical input to act as the deposit. The refund returned to the firm could be based on a constant multiplied by the weight of waste output prepared for proper disposal. The refund would

not encourage the production of further waste to increase the refund. A problem with the implementation of this system would involve the cost of new administrative duty. Production function information from individual firms including chemical inputs, products, and waste output would be necessary. Even though waste output is identifiable in the TRI, a deposit-refund program is not conducive for large-numbers-pollution-problems with chemical emissions.

Bonding systems require potential polluters to place money in an escrow account initially as an incentive to maintain environmental quality. If environmental quality remains unchanged or is improved after production to its original condition over a specified period of time, the money is returned. If not, the money is forfeited. For policy effectiveness, the payment must be large enough to provide the incentive for businesses to include the environment in their planning. Polluters would receive money back after it is determined that environmental quality did not deteriorate as a result of the production process.

The bonding system would be difficult, but possible, to implement in the context of large-scale chemical emissions. A significant payment could be established for individual firms, but administrative costs would be high as government would have to keep track of the extent of environmental damage caused by hundreds of chemical emissions for polluters in SIC 20-39. This calculation would be necessary, however, to decide whether to keep or return the money in the escrow account. A question would arise concerning the extent to which the pollutants maintained, increased, or reduced environmental quality over a period of time. As the substances cross national boundaries, are affected by meteorological conditions, and impact humans and the environment differently when they interact, the answer would be difficult to determine in the context of large-scale emission generation. With tree cutting, in contrast, a bonding system serves as an appropriate policy as business and government may keep track of the number of trees chopped and

replanted in a forest. However, the problems of administration and damage information limit the applicability of a bonding system in the context of the chemical pollution problem.

Another possibility entails the establishment of legal liability, with respect to the damages from the chemical waste, for all manufacturing companies in SIC 20-39. This regulation would require the polluters to pay for the damages for which they were responsible, including cleanup. By requiring firms to pay for damages and be held accountable if they do not, a liability system could effectively help to reduce the emissions of chemical substances. Through the Comprehensive Environmental Response Compensation and Liability Act of 1980 (CERCLA), polluters are responsible for improper waste disposal. High fines ($250,000) and jail terms up to fifteen years encourage compliance (Kahn, 1998, 425).

To address the specific problem of toxic substances and provide an incentive to reduce a subset of harmful chemical emissions from the TRI, taxation and liability systems would both serve as effective environmental policies. The above discussion makes clear that, with an effective liability system, the individual polluter would be responsible for damage and cleanup. But airborne chemical emissions may cross national boundaries or impact waterways so that the details and costs of cleanup would be difficult to determine. With polluters held responsible, the liability policy would be strict in the context of the TRI substances.

As a result of these concerns, taxation may serve as a better form of regulation in the context of option four: to regulate the TRI chemical emissions not banned through direct regulation. Firms in SIC 20-39 already have to monitor and report emission quantities annually. This information would facilitate the implementation of a per-unit tax. Environmental taxation encourages continuous innovation (OECD, 1993, 20). Even with lower chemical emissions, regulated firms must pay the tax if positive emission levels persist. A tax system could encompass a number of chemical emissions. Different rates could price the damage of individual pollutants.

A tax system would raise revenue over the life of the policy. The money could finance lower rates on non-environmental taxes, fund environmental policies or awareness campaigns, or contribute to the general budget.

Most importantly, this incentive policy would encourage the implementation of least-cost abatement technology. Cost savings would occur from policy flexibility. The regulator would tax chemical emissions, but the firm would decide how to alter its production process or selling strategy to reduce pollution. With environmental taxation, firms with lower abatement costs have the incentive to decrease their emission levels more than firms having higher costs (Baumol and Oates, 1988). While all incentive policies share the advantage of allowing firms to react individually, direct regulation does not. With this latter form of regulation, all businesses have to restrict their emissions to a certain level or implement specific abatement technology. As a result, the firms may not implement least-cost abatement technology.

This tax policy has drawbacks. Policy regulation of chemical emissions involves uncertainty and enforcement. With direct policy, regulators could mandate a specific level of chemical-emission reduction or abatement technology and know how firms would react. But it is not clear how tax implementation would alter emissions. Firms could choose to pay the tax and continue emitting the same emission levels.[9] In addition, the use of an emissions tax requires micro-level data, which leads to enforcement costs. Firms would have the incentive to under-report emissions to avoid tax payment. Finally, since Pigouvian[10] taxes reflect marginal damages from individual pollutants, damage information is needed to price the emissions. In the absence of information to levy a fully informed tax on each pollutant, an approximation must be used.

[9] The key in this instance is that the tax rate exceeds the marginal cost of abatement. Chapter two addresses this issue.

[10] These taxes are explained in the next section.

THE PROPER ROLE FOR TAXES IN ENVIRONMENTAL POLICY

Barthold (1994) explains that economic theory offers three rationales for using the tax code as a tool of environmental regulation. First, the government may levy environmental taxes as user fees. In the United States, this benefits principle applies to the motor fuels tax. Second, taxes may serve as insurance premia in a program of risk pooling. The 1980 excise tax on hazardous waste in the United States, for example, funds the Superfund cleanup program. Since it provides for the cleanup of future hazardous waste sites, the tax acts as an insurance premium. The Oil Spill Liability Trust Fund, moreover, pays for unrecovered claims that result when the owner of an amount of oil assumes first liability. Derived from a tax of five cents per barrel of oil produced domestically or imported into the country, this fund applies to cases when an oil tanker spills at sea or an oil pipeline bursts. Since it spreads risk across all oil consumers, this fund represents an environmental social insurance program (Barthold, 1994, 135).

Third, the government may regulate polluting emissions with a Pigouvian tax. Properly designed, this is the type of environmental levy that could regulate chemical emissions. In his famous book, *The Economics of Welfare*, Pigou explained how the self-adjusting market mechanism fails to maximize economic welfare and why government must intervene: "Private enterprise left to itself, even when it operates under conditions of simple competition, often leads to a distribution of resources less favourable to the national dividend..." (Pigou, 1932, 381.) Pigou believed that, since the size of the national dividend largely influenced economic welfare, unregulated firms decrease welfare through their unintended actions. According to Pigou, economic welfare is not maximized when net private product diverges from net social product.

> *It might happen... that costs are thrown upon people not directly concerned, through, say, uncompensated damage done to surrounding woods by sparks from railway engines. All such effects must be included—some of them will be positive, others negative elements—in reckoning up the social net product.* (1932, 134)

Pigou concluded that, when social and private net products diverged at the margin, the government may "put matters right by the imposition of a tax" (Pigou, 1932, 381). In contemporary terminology, the Pigouvian tax internalizes social cost by pricing pollution and raising the cost of production. In other words, the tax causes the market price to reflect social cost, including the pollution externality. A Pigouvian tax is imposed at such a rate that no additional net benefits are possible with further pollution reduction.

WHEN TAXES ARE APPROPRIATE

From an economics standpoint, an environmental tax serves as an appropriate policy when (1) the tax base is linked to the pollution source, so the firm's tax payment reflects the level of pollution, and (2) the benefits of improvements in environmental quality outweigh any administrative, compliance, enforcement, and efficiency costs of the tax, and distribution consequences are favorable (OECD, 1993, 50-53). With the first factor, an effectively designed levy creates the incentive to change behavior when the tax reflects actual polluting behavior. In the case of business activity, the higher production cost encourages the polluter to decrease the flow of harmful emissions and/or switch to cleaner production alternatives. According to the OECD, "The effectiveness of environmental taxes in achieving an efficient reduction in environmental damage will depend on the degree to which the taxation is closely linked[11] to the environmental damage which it aims to control" (OECD, 1993, 53). With the second factor, policy makers do not want a situation where costs outweigh environmental benefits to society.

[11] The concept of "linkage" between tax and source of pollution is discussed below.

THE QUESTION OF THE APPROPRIATE DEFINITION OF THE BASE

The term "environmental tax" can have several different interpretations. Authorities levy these instruments on output, inputs, and emissions.[12] The following definitions, from OECD (1995b, 11), define different environmental taxes available to policy makers.[13]

> ***Emission taxes*** involve tax payments that are directly related to the measurement (or estimation) of the pollution caused, whether emitted into air, water or the soil, or due to the generation of noise...They...are usually only suitable for stationary sources because of their high monitoring and administrative costs.
>
> ***Product charges or taxes*** can be a substitute for emission taxes when direct measurement of emissions is not possible. More generally, product taxes may be levied to price environmental effects correctly, and could be used to correct externalities other than pollution. A product tax may be levied on the units of harmful substance contained in products.

The production of output in any economic process requires the use of different inputs: capital, labor, land, raw materials and chemicals. Environmental releases occur at every step of the economic process and may also result from disposal (Figure 1). The production of batteries, for example, involves the use of cadmium, lead, mercury, nickel and other chemicals. Environmental releases of these and other metals occur from the production of metal inputs, the battery manufacturing process, and the recycling and disposal of spent batteries. Specifically, during these different stages, chemical emissions are released into the air, water, and land above and below ground (Davis, 1994a, 87-92).

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[12] The environmental economics literature explains the efficiency implications of using each of these tax bases. Chapter two contrasts the bases and explores the implications of using each for different pollution situations. Schmutzler and Goulder conclude that "(C)ombinations of emissions and output taxes are often superior to pure tax policies" using one base (Schmutzler and Goulder, 1997, 63).

[13] The literature uses the terms fees and charges to signify that the revenue is designated for specific purposes and put in a trust fund; however, a "tax" means the revenue is directed to the general budget.

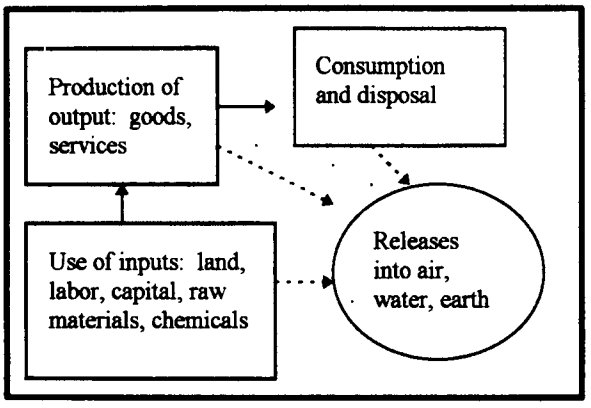

Figure 1
The relationship between inputs, output, consumption, disposal and releases

If the tax source (inputs, output or emissions) for a particular environmental levy is not linked to the pollution problem, the instrument will not directly enhance environmental quality (OECD, 1993). The carbon tax, for instance, is an appropriate instrument since carbon can be linked directly to the emission of CO_2. When one ton of carbon burns, approximately 3.667 tons of CO_2 are emitted if all of the carbon is oxidized (Dower and Zimmerman, 1992). In contrast, certain input and output taxes (e.g., on batteries, lubricant oil, and disposable containers) do not provide the incentive to lower emissions directly, but may discourage the consumption or use of the "dirty" products, which lead to environmental degradation (OECD, 1994a).

In an environmental policy setting, the choice of tax base depends on the extent of the problem and the objective of the regulator. What do policy makers attempt to accomplish? An environmental tax rate high enough to impact behavior will encourage agents to substitute away from consumption or production that degrades the environment. When taxes change relative prices substitution effects occur as producers and consumers respond. With an emissions tax, firms may adopt abatement technology to lower their emissions. This emissions levy distinguishes between a change in the production of economic output and waste output. The emissions tax discourages

waste output directly. An input tax will encourage the substitution of cleaner resources by making dirty ones more expensive.[14] An output tax will raise the price of dirty products for consumers and change the consumption mix. If the relationship between output and damage is known, an optimal environmental output tax may take the information into account. It is often the case, however, that the emissions from many sources are expensive to monitor, so output or input taxes are used in their stead. An example of this latter case is the gas-guzzler tax in the US.

The difference between an optimal output and emissions tax, if full damage information is known, is the base. Relative price changes discourage environmentally harmful activity. If a levy creates a behavioral response, a smaller tax base and cleaner environment result. But the smaller tax base will decrease government tax revenue over time. A tradeoff therefore exists between improvements in environmental quality and revenue stability (Oates, 1991).

As with all policies, environmental taxation entails cost. Roughly speaking, a tradeoff exists between administrative and compliance[15] costs on one hand and the tax link to the polluting activity on the other (OECD, 1993). Compared to output and input taxes, emission taxes have higher compliance and administrative costs (OECD, 1995b). But these levies are linked directly to the pollution source. In the absence of a method to tax emissions, output and input taxes are linked indirectly to pollution damage. Since an established taxing system exists, administrative and compliance costs are low. The cost differential occurs because the emissions tax involves monitoring and enforcement. Monitoring requires firms to alter their compliance behavior.

[14] Bovenberg and Goulder (1996) explain that the environmental tax rate on "clean" inputs should equal zero.

[15] Chapter two discusses these costs in the context of different environmental tax bases.

An important drawback occurs with incentive policy. Pigouvian levies account for pollution damage[16] (Baumol and Oates, 1988); however, problems exist with the techniques used to estimate damage functions (ORNL/RFF, 1994). In the case of air pollution, damage is widespread, accumulates over time, and is not easy to quantify. The robustness of damage estimation depends on the availability and reliability of data. As a result of estimation problems, regulators often do not formulate rates on the basis of pollution damage (OECD, 1995b).

A PLACE FOR RISK IN ENVIRONMENTAL POLICY

Every form of pollution poses a degree of harm to environmental health, the quality of human life, or the economic system. In other words, each problem presents some level of risk: "If hazard is defined as the potential for adverse consequences of some event, then 'risk' may be defined as the chance of a particular hazard occurring" (McKenzie, 1994, 89). The "risk values" used in the environmental tax framework in chapter three approximate damage from individual pollutants.[17] In terms of the familiar cost/benefit graph from economic theory, risk may mean the area under the marginal benefits function from the origin to the point of emissions (see Baumol and Oates, 1988, 58-59, for a demonstration of the graph.)

From an environmental perspective, risk means the exposure to potential harm from pollution. Since regulatory resources are scarce, it makes sense for environmental authorities to use risk as a common denominator. In doing so, policy makers may reduce human exposure of the most harmful pollutants (Reilly, 1991, 3).

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[16] Pollution may impact human health and the environment. Sulfur dioxide emissions, for example, may increase in adult's chest discomfort days and/or children's cough days (ORNL/RFF, 1994, 10-32). Moreover, SO_2 corrodes zinc. This impact could corrode zinc-coated steel (Hershaft, 1976, 995).

[17] The chemical ranking methodology of Davis *et al.* (1994c) does not express the relative risk values in terms of confidence intervals or probabilities. The methodology compares a unit of the different emissions on the basis of the potential adverse effects on human health and the environment.

With environmental regulation, the concept of risk helps policy makers and scientists discuss different pollution problems in common terms. Air, water, and ground contaminants, for instance, pose a different degree of risk to individuals, depending on location, dispersion, and many other factors. Risk helps to establish policy priorities and direction in addressing these problems, allocate resources, and mobilize and deploy expert judgment (Reilly, 1991, 4).

As the concept of risk provides a logical basis for comparison between pollution problems in general, risk could play an important role for environmental taxation in particular. If regulators could identify the damage of particular pollutants, they could levy taxes accordingly. Unfortunately, efforts to calculate relative risk values of chemical emissions often lead to incomplete analyses (Davis *et al.*, 1994b). Data on human exposure potential and toxicity, for example, are often deficient. Uncertainty often exists, moreover, with current data (Davis, *et al.*, 1994c). Toxic-response models depend on key parameters which are simulated by computers or extrapolated from experiments on laboratory animals (ORNL/RFF, 1994).

Different methods take into account all available information and develop scientific rankings for chemical emissions. These systems target pollutants in terms of toxicity and other risks to humans and the environment. Davis *et al.* (1994b) contrast fifty-one ranking methodologies on the basis of human health criteria, data selection approach, the handling of missing data, and many other factors.[18] The risk-based tax system in this dissertation uses, as a basis for tax policy, the ranking index of the Center for Clean Products and Clean Technologies (CCPCT) of The University of Tennessee. The CCPCT method ranks 158 chemicals with relative hazard values.[19]

[18] Chapter three discusses some of these measurement techniques.

[19] As chapter three explains, the hazard values take into account the health effects, environmental impact, and exposure potential of a chemical emission.

The Marginal Damage Function and Tax Policy

Pollution problems common to specific industries could be regulated with a tax system that encompasses a set of emissions. The chemical industry, for example, emits harmful substances such as formaldehyde, ammonia, sulfuric acid, nitric acid, and hydrochloric acid. To implement a risk-based tax to regulate this industry, policy makers could incorporate information concerning the relationship between emissions and environmental damage. Chemical emissions may or may not pose risk to humans and the environment. The impact depends on the toxic content of the chemical emissions, dispersion, proximity to population centers and animal life, the effect of emissions on humans, animals, and the environment, and many other factors. Damages include all of the negative impacts that users of the environment experience as a result of pollution. Contingent valuation, conjoint analysis, or another method may quantify the dollar value of damage.

A marginal damage function models this relationship (Figure 2). Damage (D) is measured on the vertical axis. The emissions of a pollutant are measured on the horizontal axis. The function demonstrates the change in damage that results from a unit change in emissions. If emissions equal E_o, a one-unit increase in emissions would lead to damage equal to D_o. At a particular level of emissions (E_o), total damages represent the area under the curve (a).

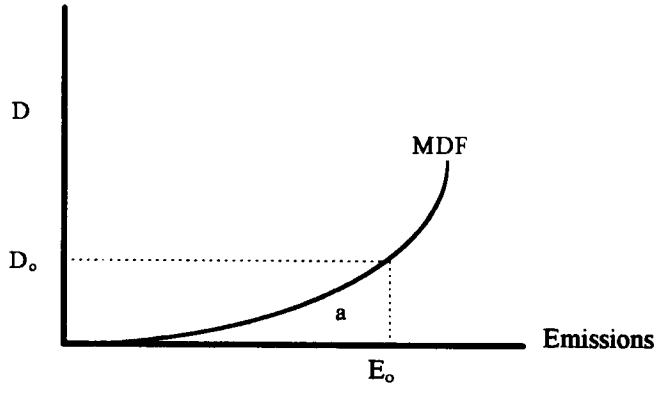

Figure 2
Marginal Damage Function

Importantly, if a marginal damage function is known for a target pollutant[20] a policy maker could levy a fully informed emissions tax. An emissions tax is a per-unit charge on an individual pollutant. In particular, the policy maker could set the rate so that tax payment would equal damage, or the area under the marginal damage function at the particular level of emissions. But, from a policy perspective, problems with tax implementation stem from the facts that marginal damage functions take many shapes, and different measures of damage are used for estimation. In terms of the shape, a higher level of emissions usually means a greater level of damage. But the relationship between emissions and damage remains difficult to define (ORNL/RFF, 1994). Environmental damage, for instance, may rise, depending on wind patterns and exposure potential, as an increasing or decreasing function of emissions. A positive relationship is not always the case, however, since the environment could be so saturated with pollution that a marginal increase in emissions would not increase total damage. A small, polluted lake serves as an example of this possibility. In general, scientists posit a direct relationship between emissions and damage, but concede that many meteorological, experimental, and human factors make the research a difficult task (Kahn, 1998, 53-56).

In terms of the different measures of damage, researchers may calculate the impact of chemical A on the number of cough days of adults, chemical B on asthma days for children, and chemical C on the corrosion rate of a certain metal. It is difficult to generalize the damage of a particular chemical emission on humans and the environment on the basis of such research. However, these measures for human and environmental damages are common in the literature. (The damage effects of different chemical emissions also may not be formulated in dollar values.) As a result of these information problems, it would be difficult, according to some consistent

[20] Chapter three reviews some estimates of marginal damage functions. The risk-based system uses a chemical-emission ranking methodology in lieu of the information that a group of marginal damage functions would provide.

measure of damage found in the literature, to levy a separate emissions tax for many of the chemical pollutants in the TRI.

The Risk-Based Emissions Tax

The damage values from the CCPCT provide an index to regulate a number of chemical emissions. By approximating the damage function, risk-based taxation sets rates that take into account relative damage values of target emissions. The risk tax is based on two assumptions: marginal abatement costs for individual firms and full information on marginal damage functions are both unknown. Given these assumptions, the risk-based system incorporates the risk unit as tax base. The risk unit equals the damage value for one pound of a numeraire chemical emission. The numeraire pollutant has a damage value and risk unit value defined to equal one. The number of risk units of individual chemical emissions equals the damage value, relative to the numeraire, of one pound of the pollutant. If one emission pound has a damage value of 1.5, relative to the numeraire, the pound would equal 1.5 risk units. Thus, one pound of each chemical emission equals a certain number of risk units. The damage values are determined by the CCPCT index. These values provide an approximation of marginal damage.

The regulator levies a tax rate on one risk unit, which translates a separate rate for each pollutant. Higher tax rates apply to the pollutants that have more risk units. The tax rate on one risk unit may be formulated to raise a target level of revenue or influence industry behavior at the margin.[21] By using a relative estimation of damage to form tax policy, the risk tax focuses on an approximate damage function of individual pollutants.

[21] This latter issue is discussed in the "Setting the Tax Rate" section in chapter four.

Other Ways to Formulate an Emissions Tax

It is necessary to consider two other possibilities for the implementation of an emissions tax. If the information were available, a fully informed emissions tax would take into account the value of actual environmental damage. (The risk-based tax above, by contrast, uses an approximation of true damage.) In the context of Figure 2, the tax payment for the firm with the fully informed levy would equal the dollar value[22] of area a if emissions were equal to E_o. For each chemical emission under regulation, policy makers would levy a different tax rate to take into account varying levels of environmental damage. As more accurate damage information became available, the rates would be adjusted so tax payment would reflect pollution damage in a Pigouvian sense. In the context of a fully informed system of emissions taxes, the measure of damage would have to be consistent across pollutants.

Using a naïve, "simple" emissions tax with the same per-unit rate on each pollutant, a regulator could ease administrative duty, reduce compliance costs and regulate a number of chemical emissions. Lacking a method to link the tax rate to marginal damage, this tax would not take into account the fact that a pound of each effluent may impact humans and the environment in different ways. With this framework, the simple emissions tax system is not Pigouvian in nature, but would allow firms to estimate their tax payment ahead of time with emission projections. The firms would not have to face a different rate for each pollutant. Currently, no such system exists.

With the simple emissions tax, policy makers could attempt to raise a certain level of revenue. Suppose a tax rate of t_1 on each target pollutant would achieve this goal (Figure 3). The firm would pay t_1E_o dollars, which is greater than the amount of total damage at emissions E_o, for the target pollutant. With other chemical emissions, the tax rate of t_1 would necessarily understate

[22] As they apply to damage functions and tax policy, economic valuation techniques are discussed in chapter three.

the level of total damage. This policy system eases administrative duty, but does not accurately incorporate the damage of target pollutants.

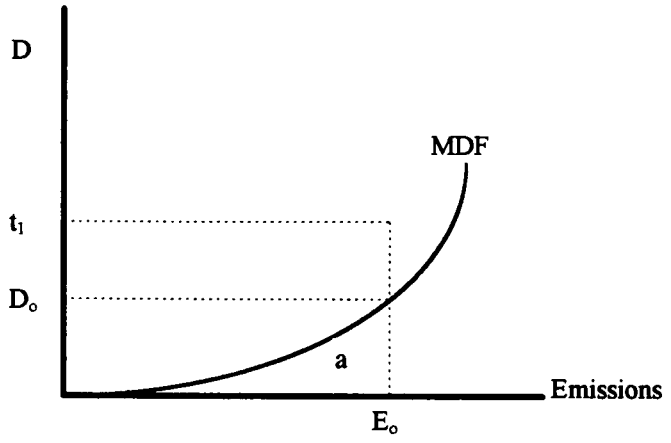

Figure 3
Marginal damage function and the simple emissions tax

CRITERIA FOR EVALUATION OF ENVIRONMENTAL TAXES

Before policy implementation, regulators must address a number of important design issues: the appropriate levy, taxable base, interaction between old taxes and new, interaction between new taxes and existing regulation, and the interaction among pollutants. These issues must be evaluated within the overall policy scheme. For example, specific environmental problems may dictate potential tax bases. These bases make clear potential environmental levies. But regulation may operate with other policy; this result would cloud the estimate of the levy's impact.

The policy maker must compare policy benefits with regulation costs. From society's perspective, the governing authority must consider whether tax implementation creates more value than is lost. Will the benefits of a cleaner environment outweigh the costs of administration and compliance? The relevant criteria for cost/benefit analysis are efficiency, equity, administration,

compliance, revenue, and the impact on environmental quality over time. These criteria also include sub-categories. If the benefits exceed the costs, welfare gains accrue to society. These criteria will be used to evaluate the risk-based tax.

Because the taxation of a particular base involves tradeoffs, the choice of policy instrument becomes a "portfolio problem." The implementation of an emissions tax may discourage waste output directly but entail high monitoring and enforcement costs. This policy may also discourage the use of complementary items. The policy maker must consider criteria tradeoffs such as these. To address the chemical emission problem, different bases may also be targeted. Table 4 summarizes the evaluation criteria, which are explained below.

Goulder explains the first criterion by saying that "the notion of efficiency encompasses all of the aggregate welfare impacts of policy changes, including those that arise from policy-induced environmental impacts" (Goulder, 1994, 110). The literature normally separates environmental and nonenvironmental effects. If an environmental tax connects directly to the pollution source, internalizes the negative externality at the optimal level of emissions, and does not discourage the production of economic output, efficiency gains occur. In addition, when a policy reduces pollution *toward* the optimal level, ceteris paribus, it increases efficiency. If, however, an environmental tax reduces pollution, but also discourages "related" economic activity,[23] efficiency losses result. The net welfare effect of a particular environmental policy depends on the magnitudes of the benefit of pollution reduction and the cost of efficiency losses.

[23] From an environmental perspective, "effective" pollution taxes discourage waste output but not economic output (the production of goods and services), unless the consumption of the product itself pollutes the environment. "Related" economic activity is discussed in the next paragraph.

Table 4
Evaluation criteria

Criteria	Fully informed Emissions Tax	Risk Tax	Output Tax	Input Tax
Efficiency				
Tax interaction effect				
Revenue recycling effect				
Equity				
Administration				
Adaptability				
Collection				
Enforcement				
Source/destination				
Compliance				
Monitoring				
Evasion				
Avoidance				
Revenue				
Yield				
Adequacy				
Short-run stability				
Elasticity				
Environmental impact				

This second efficiency consideration concerns related efficiency effects. An environmental tax creates efficiency costs if it distorts[24] the labor-leisure, intertemporal,[25] intermediate resource, or consumer product choice (Goulder, 1994, 110). The former two choices are components of the factor market while the latter two exist in the domain of the product market.

To maintain or increase efficiency, environmental taxes should minimize any resulting burden. Bovenberg and de Mooij (1994) explain that the implementation of an environmental output tax may distort the individual's labor-leisure choice: higher production costs from environmental taxation lower the after-tax wage, which causes workers to substitute leisure for labor. This distortion occurs if the government relies on revenue substitution[26] with income taxes.[27] (Revenue substitution occurs when a government finances a lower rate on one tax with a new tax or a higher rate on an existing levy.) According to Goulder (1995a), two efficiency conditions result from revenue substitution with environmental taxes: the tax-interaction and revenue-recycling effects. The first effect represents the welfare costs of introducing new environmental taxes. The second explains the decrease in tax-system distortions from a lower pre-existing tax.

When a wide tax base achieves a target level of revenue, efficiency costs will be small[28,29] (Rosen, 1992). In addition, the efficiency cost will decrease the more inelastic is compensated demand or supply.[30] To evaluate the full impact of a tax, economists use a general-equilibrium framework to capture the efficiency effects in related markets.

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[24] A tax distorts an economic choice if it causes the individual to make a different decision than would occur in the absence of the tax.

[25] This is the choice between consuming today or tomorrow by saving and investing.

[26] To improve environmental quality, policy makers must set rates at sufficient levels to decrease the taxable base. If the government relies on a specific environmental tax for a certain amount of revenue, a smaller tax base proposes a problem.

[27] But Goulder, Parry, and Burtraw (1996) show that revenue substitution leads to the lowest efficiency losses when compared to other policy instruments (grandfathered permits, command-and-control regulation) that do not raise revenue.

[28] With respect to the Bovenberg/de Mooij framework, a wide base facilitates revenue substitution.

[29] But, from an environmental standpoint, society wants the tax base (pollution) to decrease over time.

[30] The "Ramsey" condition endorses tax rates in inverse relation to demand elasticity.

To evaluate efficiency effects, this dissertation considers whether an environmental tax satisfies the Pareto conditions of microeconomic theory, provides incentive to reduce pollution, distorts related economic decisions, and leads to tax-interaction and revenue-recycling effects. Chapter two addresses the concept of efficiency in the context of these issues.

The second criterion involves equity. An environmental tax that burdens a particular income group or region limits the effectiveness of the policy. Baumol and Oates suggest that

> *... (S)trong measures to improve environmental quality may indeed have a very uneven pattern of incidence, particularly during the period of adjustment to a new composition of output and employment. Moreover, the evidence suggests that we can typically expect a somewhat regressive pattern of distribution of the benefits and costs from environmental programs.* (Baumol and Oates, 1988, 236)

For horizontal equity to hold, individuals at similar income levels must bear the same tax burden. With vertical equity, individuals with higher incomes bear greater tax burden. Equity analysis determines whether a tax is progressive or regressive with respect to income groups. To address the equity issue, this dissertation considers whether a particular income group or region receives undue burden. Equity also depends on who benefits from environmental change. People living in highly polluted areas may benefit more from pollution reduction.

Third, policy makers must evaluate the costs of administration. To benefit society, policies with high public costs must significantly enhance environmental quality. Administrative costs include information gathering on pollution sources and destination, policy adaptability, the implementation of different tax rates, the ease of applicability and enforcement. With the latter concept, certain environmental problems transcend the scope of policy jurisdiction: pollution may flow beyond state or local borders. The extent that an environmental tax helps to reduce pollution signals the effectiveness of the policy. As Alm concludes, "there is little information on how these (administrative) costs vary with various policy tools; that is, it seems likely that the administrative costs change in large and discrete amounts with the scale of collections...and that they may also

display economies of scale in their collections, but these aspects of the collection cost technology are not known" (Alm, 1996, 121).

Fourth, compliance costs include standard responses for tax payment, comprehending tax law, seeking professional advice, and filling out returns. Compliance costs also include evasion and avoidance (Alm *et al.*, 1990, 603). Evasion entails the illegal game of tax reduction or non-payment. Avoidance, in contrast, includes the legal methods of tax reduction. Alm explains that "Compliance costs of taxation are significant, often of comparable or even larger values than the more traditional calculations of the excess burden of taxation" (Alm, 1996, 121).

For environmental taxes, monitoring costs represent an important aspect of compliance. In particular, monitoring systems are essential for the administration of emission levies. As a result of the monitoring requirement, companies may attempt to avoid tax payment. The next chapter explains that monitoring costs for emissions taxes exceed the costs for output and input levies. In sum, administration and compliance costs vary considerably between policies.

Fifth, environmental taxation creates revenue. The policy maker must consider different revenue issues including yield, buoyancy (elasticity), rate and base behavior, long-tern adequacy, and short-term stability. As discussed previously, a tradeoff exists between revenue stability/adequacy and improvements in environmental quality (Oates, 1991). Tax regulation could rest in the hands of an environmental or a revenue authority. An environmental regulator would set tax rates to impact behavior and reduce pollution. A revenue authority, however, may set rates at lower levels to maximize revenue over time, but not decrease the base.

The sixth criterion relates to the first. While efficiency analysis encompasses pollution reduction and related effects (discussed above), the environmental impact constitutes the most important criterion from a regulator's perspective. Do taxes create an incentive for pollution

reduction? If a levy reduces the pollution from individuals and/or firms, then it helps to enhance environmental quality. Appendix 1 lists several taxes that elicit an incentive response.

OUTLINE OF THE STUDY

Chapter two explains that, for policy makers, the implementation of an environmental tax system means the comparison of costs to benefits. Using the judgment criteria from chapter one, chapter two contrasts environmental output, input, and emissions taxes. Chapter two also addresses the fact that practical environmental policy does not exist in a "first-best" world. On the contrary, new regulation must consider non-optimal solutions, information problems and pre-existing regulatory distortions. Chapter two concludes by making a case for risk-based taxation.

Chapter three addresses the important design issues of the risk-based tax system. In particular, this chapter discusses research on the estimation of marginal damage functions, contrasts these studies to the CCPCT ranking system, and explains the latter methodology. A discussion of the key issues of the risk-based system follows this material. Chapter three explains that the risk-based framework adopts an indirect measure of damages as its basis for taxation. In addition, the use of the risk unit allows a policy maker to include a number of chemical emissions within the framework of the policy. Chapter three explores the strengths and weaknesses of risk-based taxation and discusses when this instrument serves as an appropriate environmental policy.

Chapter four considers, in terms of the welfare impact on society, the implications of shifting to risk-based taxation. To put the risk-based system in perspective, chapter four first analyzes different environmental taxes in a mathematical framework. Chapter four then identifies target chemical emissions and develops a simulation experiment that contrasts, with welfare

analysis, alternative environmental taxes. To calculate efficiency effects, the model in chapter four is adapted from the theoretical framework of Parry (1995).

Chapter five concludes this dissertation. Chapter five summarizes the merits of risk-based taxation, contrasts indirect and direct risk-based taxation, and discusses the costs and benefits of the policy in terms of the portfolio problem. Chapter five also outlines a proposal for the adoption of the system. The latter section addresses implementation problems and sequence. Chapter five concludes with a final commentary.

CHAPTER 2

PROPERTIES OF ALTERNATIVE ENVIRONMENTAL TAX BASES

The regulator faces a difficult task in matching policy with environmental problem. The authority must evaluate pollution characteristics, potential effectiveness of individual policy instruments, the interaction of new regulation with old policy, synergy between individual pollutants, the availability of public funds and many other concepts. With environmental taxation, in particular, the policy maker must decide when to use a certain base. Chapter one explains that regulators tax emissions, output and inputs.

Chapter two discusses the possibility of levying different environmental taxes to address the chemical emission problem. With chemical emissions, a fully informed Pigouvian levy would tax each pollutant according to a calculated measure of marginal environmental damage. A risk-based levy, in contrast, would serve as a proxy for a fully informed policy. Output and input levies could price products and resources, but would not tax the pollution source directly. To put the risk tax in perspective, chapter two first applies the evaluation criteria of chapter one to the fully informed emissions, output, input and risk-based levies. Chapter two then brings all of the criteria together in the context of the portfolio problem of environmental taxation. The purpose of chapter two is to show that, while emissions-based levies are preferable to output or input taxes in the context of the chemical emission problem, these former policies entail important administrative and compliance costs. Policy tradeoffs occur, in other words, when evaluating the overall impact of a specific environmental tax.

THE PIGOUVIAN EMISSIONS TAX: EFFICIENCY

The Pigouvian emissions tax establishes the textbook case when an environmental levy internalizes the pollution externality. As a result, this section begins by evaluating its characteristics. This initial policy example incorporates a first-best world without pre-existing tax distortions and finds the optimal solution, i.e. optimal tax rate. In addition, identifiable abatement and damage functions provide accurate information for the administrator while firms exhibit perfect compliance.

Baumol and Oates (1988) explain that emission taxes internalize the pollution externality at the optimal emissions level in a "first-best"[31] world. In this optimal setting, emission taxes satisfy the three Pareto efficiency conditions established by welfare economics. For the sake of simplicity, a first-best mathematical presentation narrows the focus to a two-by-two world with outputs X and Y, inputs K (capital) and L (labor), output prices P_X and P_Y, input prices W_K and W_L and persons a and b.[32] The production process which creates the two outputs leads to a polluting emission called externality Z.

In this two-by-two world, the government must maximize the utility of one individual (U^a) subject to the conditions that the other person's utility (U^b) does not decrease, while maintaining productive efficiency in both product and factor markets. The production of X creates emissions S^X while the production of Y creates S^Y where $Z = S^X + S^Y$. The derivation of efficiency conditions one through three appears in Appendix 2 along with Figure 10, which depicts the optimal level of emissions in a first-best world. The following equations result.

***First efficiency condition*:** for person a the MRS_{XY} = the MRS_{XY} for person b. The marginal rate of substitution (MRS) is equal to the ratio of the individuals' marginal utilities.

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[31] A first-best world means a tax implemented at the optimal economic solution with no pre-existing tax distortions (see Figure 4).

[32] This presentation is an adaptation of the general discussion of this concept in Baumol and Oates (1988).

***Second efficiency condition*:** for good X the $MRTS_{KL}$ = $MRTS_{KL}$ for good Y, where MRTS = marginal rate of technical substitution.

***Third efficiency condition*:** $-\Sigma MRS_{YZ} = f_S$ for good Y. (The same holds true for X.)

The pollution problem occurs in this context as private prices do not reflect the externality and government must intervene to set a tax equal to the difference between marginal private and marginal social costs.

PROBLEMS WITH A "FIRST-BEST" ANALYSIS

The optimal-tax literature does not provide an analysis of the link between emissions and environmental damage, which is crucial for the application of environmental taxation. Large information costs and the presence of uncertainty prohibit the implementation of a first-best tax. Government regulators cannot "observe" the abatement cost functions of all polluting firms and the damage functions of individual chemical emissions. To find the optimal level of emissions, policy makers would have to estimate the expected value of benefits or damages avoided. This procedure requires the calculation of the value of damage reduction and the assignment of a probability associated to a particular outcome (OECD, 1995b). The valuation of social costs, moreover, requires additional information beyond the impact of a tax that alters production. The problem is further complicated by the existence of many pollutants. Pollutants interact, which clouds the estimate of the impact of any particular emission.

A first-best analysis also assumes away other taxes in the market place. But pre-existing taxes interact with environmental levies and obscure the impact of the latter instruments. In fact, environmental taxes may exacerbate pre-existing tax distortions (Bovenberg and de Mooij, 1994; Oates, 1995; and Parry, 1995). Even more, the interaction of new environmental taxation with

pre-existing direct control complicates the evaluation of the effects of the former policy. These three issues—information limitations in the regulation of many pollutants, the problem of pre-existing tax distortions and the problem of direct control—create the need for an "n^{th}-best" analysis in the development of a risk-based environmental tax system.

THE WORLD OF THE "N^{th} BEST"

In the face of uncertainty concerning the shapes of marginal abatement cost and damage functions, policy makers may levy environmental taxes to impact behavior at the margin. The regulators may also raise revenue for the general budget or an environmental program. In these situations the policy tries to create a framework conducive for pollution reduction without achieving an optimal solution. If governing authorities look to the long term, firms may adjust their production processes over time. As polluters implement abatement technology and change the capital stock, they will establish an economy with a lower level of pollution.

To briefly state the difference between, first-, second-, and n^{th}-best solutions, the former situation exists at the optimal point of pollution abatement where marginal abatement cost equals the marginal benefit of pollution reduction. A second-best framework attempts to find the optimal solution in the face of one or more constraints that limit optimality. An nth best policy creates a movement toward the optimal solution (optimal level of pollution with environmental taxes), given a non-optimal position and information limitations (Harberger, 1964, 58).

The risk-based tax system, developed in chapter three, addresses these information and pre-existing tax problems. The risk-based system takes a practical approach in the formulation of a policy that encompasses many pollutants in an nth-best world. The risk-based system uses damage values for individual, target pollutants in the absence of full information on damage

functions and abatement cost information. The application of the risk-based system in a simulation experiment in chapter four incorporates pre-existing levies in the marketplace.

Non-Optimal Solutions

Economists prefer emissions taxes (see, e.g., Baumol and Oates, 1988); however, a policy maker may find it desirable to tax "dirty" output or inputs in the presence of information restrictions, high monitoring costs and/or many pollution sources. Taxes on output and inputs could enhance environmental quality in particular situations. When the relationship between output/inputs and emissions remains stable, for instance, a product levy may encourage pollution abatement (OECD, 1995b). In addition, when the consumption of a commodity leads to pollution damage, an output tax will help enhance environmental quality. On the other hand, if properly employed, an emissions tax reduces pollution toward the optimal level. But information restrictions eliminate the possibility of achieving optimal pollution reduction. Non-optimal solutions are the reality of environmental tax policy.

The Impact of Pre-existing Tax Distortions

When formulating new regulation, policy makers must consider the interaction of new environmental taxes with pre-existing levies on income, sales, payroll, etc. The reason is that economists have recently shown that the implementation of new environmental levies may exacerbate tax distortions already present in the marketplace (Bovenberg and de Mooij, 1994; Parry, 1995; Bovenberg and Goulder, 1996). For this reason, and to begin to put the empirical application in chapter four in context, this subsection explores some recent literature which discusses the implementation of environmental taxation in a world with pre-existing levies. This

analysis applies to the development and explanation of the risk-based system since this policy is a new form of environmental regulation. From the perspective of society as a whole, the costs of policy implementation must include interaction with taxes that already exist and any resulting efficiency losses.

The issue of pre-existing tax distortions became an important topic with the progression of the "double dividend" literature (Goulder, 1995a). Theoretically, environmental taxes reduce excess burden over some range of implementation; that is, they improve resource allocation since they discourage "dirty" production processes for firms. For government, environmental tax revenues present an appealing source of funds. If authorities use the money to finance lower rates on other taxes that distort economic activity, the excess burden of the existing tax system could decrease (Oates, 1991). Environmental taxes could therefore confer two dividends: enhanced environmental quality and increased tax-system efficiency.

But with their model and theoretical application, Bovenberg and de Mooij (1994) do not affirm the double dividend hypothesis. In a general-equilibrium framework, the authors create a model in which environmental tax revenues finance a lower rate on an income tax. Specifically, a tax on "dirty" commodities increases product price. As a result, individuals consume more clean products, which reduces the environmental tax base.[33] This smaller base lowers employment after a decrease in wages. Wages fall since the lower tax rate on labor income does not fully compensate individuals for the negative effect of the decreasing environmental tax base on their real wage. This wage effect distorts the labor-leisure choice. Labor supply decreases as a result. This loss represents the tax-interaction effect. In the context of the Bovenberg/de Mooij model, the revenue-recycling effect explains the gains to society from having a lower tax rate on income.

[33] But also, I argue, improve the environment.

In general, it would be difficult for government to find the "correct" prices for environmental levies when they exist alongside sales, income and payroll taxes. In this second-best world, the cost to society of introducing new environmental taxes depends on the marginal rates of sales, income, payroll and other levies. As Bovenberg and de Mooij argue, the introduction of environmental taxes in a second-best world may increase the distortions that already exist.[34]

Bovenberg and de Mooij raise some important issues. First, they show why an analysis of environmental taxation must consider interaction effects with other levies. Second, they show that efficiency gains must occur if policy makers use environmental tax revenues to finance lower rates on levies set at high levels. The national government of Sweden, for example, reduced the marginal income tax rates to 30 percent for more than 80 percent of income earners and to 50 percent for the remaining high-income earners after the implementation of some environmental taxes in the 1990s. (The rate for the highest income bracket was 73 percent in 1989.) Surely this policy of revenue substitution resulted in efficiency gains (Oates, 1995).

The influential double dividend models (Bovenberg and de Mooij, 1994; Bovenberg and Goulder, 1996; Parry, 1995; and Goulder, 1995b) assume one pollutant (tax source) for the process of revenue substitution. These analyses conform to the environmental tax literature by examining one base. What is missing in the literature is an analysis of a *system* of environmental taxes on bases other than output and inputs. For instance, a governing authority could regulate a *set* of emissions and implement a policy of revenue substitution. A tax rate on each pollutant—

[34] However, the Bovenberg/de Mooij analysis does not include the impact of environmental quality on production. The welfare effect that results from the introduction of a revenue-neutral environmental tax must weigh the gross benefit (cost) of a cleaner environment and the gross benefit (cost) of efficiency changes in the tax system. Why did they omit this impact? As Goulder (1995a) notes, measuring the monetary value of a healthier environmental is a difficult task. In the Bovenberg/de Mooij paper, environmental quality enters the utility function as a non-rival consumption good. Individuals can separate the consumption of ordinary goods from the utility gained from environmental quality. In effect, Bovenberg and de Mooij separate environmental benefits from the costs of revenue substitution. Since it is difficult to measure, the authors' final welfare analysis does not consider that a healthier environment may increase the marginal productivity of pollution-sensitive inputs.

significant enough to enhance environmental quality—would target a wider base than a single pollutant, but would still decrease over time.[35] The development of the risk-based tax policy in chapter three and subsequent welfare analysis in chapter four addresses these issues. Chapter four also considers a related question: could the larger revenue stream facilitate a policy of revenue substitution? Bovenberg, Goulder, and Burtraw (1996) argue that environmental taxation and revenue substitution entail the lowest efficiency cost when compared to other regulations that do not raise revenue.[36]

The Impact of Pre-existing Direct Control

Another factor to consider with new environmental tax policy is the presence of direct control. In most industrialized countries direct and incentive regulation coexist at the national and sub-national levels of government (OECD, 1994a). In many places, taxation enforces direct regulation by financing recycling programs and waste treatment centers (OECD, 1994a). In the United States, the 1990 Amendments to the Clean Air Act established a national SO_2 marketable permit system for power plants (in Title IV) in addition to standards for emission reduction (ORNL/RFF, 1994, 10-38). Historically, however, many countries have implemented direct measures in lieu of incentive policy. Regulators have often chosen the reliability of direct control (OECD, 1994a).

The combination of incentive policies and direct regulation varies between countries. The United States relies on direct regulation at the national level, although many incentive instruments exist at the national, state, and local levels, including charges on hazardous waste, water effluent, and air pollution (OECD, 1994a). In contrast, the cornerstone of environmental policy in the

[35] The tax base would decrease if we assume a behavioral response among polluters and an improvement in environmental quality.
[36] The following subsection addresses this paper and point.

Netherlands and France is the water effluent charge (Andersen, 1994). In the future, many countries such as the Netherlands, Sweden, Norway, and Denmark plan to change their mix of policy instruments to give a preeminent role to incentive instruments (OECD, 1994a)

The presence of pre-existing direct control could make it difficult to observe the effects of a new tax policy. Existing controls could cloud the estimate of a new tax's environmental impact. In the presence of direct regulation, the implementation of new incentive policy could raise production costs unnecessarily. It may be that costs to the firm would decrease if incentive policy *replaced* direct regulation.

Direct regulation may limit pollution flow and create a situation where an additional environmental tax would not lead to further reductions below the amount mandated by existing policy. As an example, consider sulfur dioxide emissions from electric power plants that use coal. The 1990 Amendments to the Clean Air Act regulate this effluent. An additional energy tax would reduce the demand for electricity, lower power plants' output and allow the power plants to comply with the mandated standards with higher ratios of sulfur dioxide per unit of output. In this situation, the power plants do not have to reduce the level of emissions per output within the framework of the two policies (Goulder, 1994, 117). If requirements for mandatory abatement equipment are already in place, the implementation of the risk-based tax system could lead to similar results.

ALTERNATIVE TO THE PIGOUVIAN EMISSIONS TAX

In various countries, environmental taxation, other incentive measures and direct regulation offer different degrees of control (OECD, 1995a). Most often, as a result of the certainty of the resulting standards, regulators employ direct policy. Environmental taxation,

however, helps to enhance environmental quality, raises revenue, and allows the flexibility of choosing output, inputs or emissions as base. Weitzman concludes that, "generally speaking it is neither easier nor harder to name the right prices than the right quantities because in principle exactly the same information is needed to correctly specify either" (Weitzman, 1974, 478). In his influential article, Weitzman shows that, as the marginal abatement cost (MAC) function becomes flatter, the actual level of pollution reduction that results from a pollution tax moves further away from the target level of pollution reduction. When the marginal damage (MD) function becomes steeper, greater social losses occur when actual pollution reduction deviates from target pollution changes. With information on the slopes of the MAC and MD functions, direct quantity restrictions may be preferable to incentive instruments that focus on price (Weitzman, 1974).

An Evaluation of Risk-Based Taxation: Efficiency

Consider the difference between a fully informed emission tax and a risk-based levy. The tax rates for the former policy would incorporate a proven scientific link between emissions and marginal damage. The regulator prices pollution damage using emissions as the base. But, as a result of an information problem, it is difficult to implement, according to marginal damage functions, fully informed emissions taxes on the target chemical pollutants.

In contrast, to approximate fully informed regulation, the risk-based tax incorporates damage information from the CCPCT ranking and scoring index. This policy levies a rate that applies to the number of risk units of each pollutant, uses emissions as the vehicle to apply the tax and attempts to raise the cost of the most harmful chemical emissions under regulation. As a proxy for a fully informed levy, risk-based taxation incorporates available information to distinguish between the damage potential of individual pollutants.

Chapter three explains, for example, that chlorobenzene represents the numeraire pollutant in the risk-based tax system. As a result, one pound of chlorobenzene equals one risk unit (RU). All other pollutants are assigned a quantity of risk units relative to this numeraire, according to the hazard value in the CCPCT index. For example, one pound of styrene emissions equals 1.61 risk units while one pound of trichlorobenzene equals two RU. With a tax rate of $1 on one risk unit, the polluter would pay $1 for the pound of chlorobenzene emissions, $1.61 for the styrene, and $2 for the pound of trichlorobenzene. This tax system uses the risk unit as base and alters relative prices according to damage values.

The key in comparing the risk-based levy to the fully informed emissions tax concerns how well the former policy approximates the latter. The degree of approximation depends on the relationship between damage and emissions for each target chemical. The index values incorporated in risk-based taxation from the CCPCT imply an increasing level of damage as emissions increase. No substance, however, will assume this pattern exactly.

For efficiency gains in the aggregate, benefits from improvements in environmental quality have to outweigh any efficiency costs that result from the policy. To enhance environmental quality, the risk-based levy must effectively price different chemical emissions according to their relative impacts. The extent to which the risk-based levy achieves this goal and creates an overall efficiency gain is the subject of chapters three and four.

ALTERNATIVES TO EMISSIONS-BASED TAXES

With the chemical emissions problem, regulators could target an alternative base. To address this possibility, environmental output and input taxes are discussed as policy options. A policy maker could levy a tax on products from target manufacturing industry or "dirty" chemical inputs. The presentation below patterns the above discussion and incorporates the efficiency

effects from the beginning of the chapter. The environmental levies in this section discourage the use of dirty inputs and output, but do not discourage waste output directly.

An Evaluation of Output and Input Taxes

In a recent paper, Schmutzler and Goulder (1997) consider the regulator's choice between output taxes,[37] emissions taxes or a combination of both. They show graphically and mathematically that, when monitoring costs are low, effective end-of-pipe technologies exist, and the sources of emissions are identifiable, an emissions tax serves as the better instrument. But the conditions of high monitoring costs, no options for pollution reduction except a decrease in output and a high substitutability of the final product favor output taxes. A combination of output and emissions taxes is optimal, moreover, with low fixed monitoring costs and low initial marginal costs of monitoring that increase rapidly. Schmutzler and Goulder effectively show that policy depends on circumstance.

Environmental taxes on inputs increase resource prices and encourage the substitution to cleaner alternatives. When firms use cleaner inputs, environmental quality improves. The probability of substitution in the production process depends on technology, uniqueness of the product, resource prices and the tax rate. In some industries, flexible production methods may incorporate different inputs at low cost. But certain resources must serve as inputs in the production of final goods, such as chromium in the production of steel and cast iron (Davis *et al.*, 1994a). The governing authority will create a behavioral response among polluters with input levies by targeting those resources that have feasible substitutes.

[37] The authors state that, in their model, an input tax would serve the same purpose as an output levy.

Efficiency

A textbook emissions tax satisfies the three Pareto conditions in a first-best world. How do environmental taxes on output compare? The individual's first-order condition, with the tax levied on the dirty output X, from the beginning of the chapter is

$$MRS_{YX} = P_X(1+t_x)/P_Y.$$

With an environmental externality, the prices of commodities do not reflect the true MRS. Optimal environmental taxes therefore improve efficiency by pricing pollution damage. From an environmental standpoint, tax differentiation on final products should reflect the difference between damage potential. The environmental output tax makes the dirtier product X relatively more expensive than the cleaner Y.

Do environmental output taxes distort related economic activity? Goulder (1994) finds that a gasoline tax distorts the labor-leisure and consumer good choices. The latter distortion results when the environmental levy makes commodity taxation less uniform. But, from an environmental perspective, the levy prices the "dirty" commodity. The overall welfare effect of the gasoline tax depends on the benefit of environmental improvement, the cost of regulation and any efficiency losses.

If a link between a dirty input and pollution can be established, a resource levy would price the pollution externality. In the two-by-two framework established at the beginning of this chapter, an input levy on one resource raises the price of the dirty substance to the manufacturer. If a tax is levied on the dirty input K, but not L, the following first-order condition results:

$$MRT_{LK} = W_K(1+t)/W_L.$$

As a result of the change in relative prices, the levy prices the externality. With a high substitution elasticity between the factor inputs, a higher price for K encourages the use of the cleaner alternative. Bovenberg and Goulder (1996) show that, in a general-equilibrium model with

pre-existing taxes that distort economic activity (income taxes), taxes on "clean" inputs should equal zero while levies on dirty inputs should be positive.[38] In the presence of the externality, the tax prices the external cost of pollution.

Other efficiency considerations include the extent of the taxable base and potential for impacting related economic activity. The former concept depends on the particular resource. Energy inputs, for instance, are used widely; however, particular chemical resources may exist in the production processes of few firms. With the latter issue, Goulder (1994) examines the efficiency costs of the Btu levy in related markets. He finds that this tax distorts the labor-leisure, intertemporal and intermediate choices. But the levy's impact also depends on the value of an increase in environmental quality. Future research must consider the overall impact of this levy when incorporating the benefits of enhanced environmental quality.

Second-Best Efficiency Effects: Setting the Tax Rate[39]

The environmental tax literature has addressed the problem of finding the rate when market distortions exist. The difference between optimal and non-optimal solutions in a second-best world concerns the additional efficiency losses that occur with the interaction of an environmental levy and pre-existing taxes.[40] Optimal solutions take into account the efficiency consequences of this interaction and show how the second-best tax rate differs from the Pigouvian

[38] The two-by-two framework does not include previously existing tax distortions, but the results are similar to the Bovenberg/Goulder assertion when the two-by-two case includes one dirty and one clean input.

[39] Optimal tax models attempt to minimize the excess burden of the tax system subject to some revenue constraint, but also include an analysis of other tax considerations, including equity, administration and compliance. This area of literature evaluates the tax criteria in a cost-benefit framework. Since an individual tax or system of taxes will lead to different effects in terms of the judgment criteria of chapter one, environmental tax analysis becomes a "portfolio" problem. For a helpful discussion, see Alm (1996).

[40] Two possibilities create a non-optimal situation: a lack of marginal damage or abatement cost information on the part of the environmental regulator, or pre-existing tax distortions. The literature incorporates the latter issue in theoretical double dividend models (Oates, 1995).

rate. Non-optimal solutions have lower efficiency costs, but do not attempt to find the optimal tax rate. The literature has addressed these issues with input and output levies, but not emission taxes.

The reason for this subsection is twofold: to show the complications involved with finding the "proper" environmental tax rate in a world with pre-existing market distortions, and to show the information necessary to levy the "proper" rates in an optimal tax framework. The discussion in this section makes it clear that optimal tax literature does not necessarily aid the policy maker in the application of new policy. To formulate the tax rate for risk-based taxes in chapter four, different criteria are explored.

Consider an analysis of "optimal" solutions in a second-best world. In the presence of an income tax, the public authority could maintain revenue neutrality with the introduction of environmental taxation. From a tax standpoint, financing a lower income tax with environmental taxation (revenue substitution) leads to two results: the revenue-recycling and tax-interaction effects [Goulder (1995a) and Parry (1995)]. The former effect means the benefit of a lower income tax and smaller tax distortions. The latter effect explains the exacerbation of pre-existing tax distortions (Bovenberg and de Mooij, 1994.) In a context of revenue substitution, these results are additional efficiency considerations.[41] Environmental tax research concludes that, under plausible circumstances, the tax-interaction effect outweighs revenue-recycling (Oates, 1995).

If this interaction holds, the optimal environmental tax rate in a second-best setting lies below the Pigouvian rate. Bovenberg and van der Ploeg (1994) conclude that

$$t^* = MED/\mu ,$$

where t^* is the optimal tax, MED is the marginal environmental damage and μ is the marginal cost of public funds (any efficiency cost of taxation). In a first-best world, the environmental tax, t^*, equals MED. But in the world of the second best, the tax must take into account μ. Specifically, a

[41] Chapter four discusses the revenue-recycling and tax-interaction effects in detail.

high value of μ implies that it is more costly for government to provide environmental quality; therefore, the higher the value of μ, the lower the amount of environmental quality, and a lower tax rate t*, ceteris paribus.[42]

Extending the analysis, Bovenberg and Goulder (1996) derive optimal environmental tax rates for dirty intermediate inputs and output in a second-best setting. In a general-equilibrium framework, the model assumes that the public authority may levy taxes on labor income, clean and dirty intermediate inputs, and dirty output. While the tax on clean inputs equals zero, the optimal tax on dirty intermediate inputs is

$$t^* = \frac{(\delta U/\delta Q)(-\delta q/\delta x_d)}{\delta U/\delta C_C} (1/\mu),$$

where $\delta U/\delta Q$ is the partial derivative of the utility function with respect to environmental quality, Q; $\delta q/\delta x_d$ is the partial derivative of environmental quality with respect to the dirty intermediate input; $\delta U/\delta C_C$ is the partial derivative of the utility function with respect to the clean consumption good; and μ again represents the marginal cost of public funds. The first term on the right-hand-side of the equation represents the MED. Analogously, the optimal tax on the dirty consumption good is the MED from the use of the product divided by μ:

$$t^* = \frac{(\delta U/\delta Q)(-\delta q/\delta C_D)}{\delta U/\delta C_C} (1/\mu).$$

These two equations explain how pre-existing tax distortions affect the optimal environmental tax. The first term (numerator and denominator) on the right-hand-side of each equation is the Pigouvian rate in a first-best world. In this situation, a reduction in emissions of one unit corresponds to the welfare cost of a decrease in tax revenue due to a smaller tax base. Here, the marginal cost of public funds equals one (public funds cost the same as private funds.) With pre-existing tax distortions, however, public funds are more costly than private funds and μ is

[42] Parry (1995) supports this conclusion.

greater than one; therefore, Bovenberg and Goulder conclude that "the government finds it optimal to cut down on public consumption of the environment by reducing the pollution tax" (Bovenberg and Goulder, 1996, 987).

CONCLUSION: RELATIVE EFFICIENCY EFFECTS

To address the chemical pollution problem, a policy maker could choose between output, input and emission levies. The question of the appropriate base depends on the link between tax source and pollution damage. At the (first-best) optimal level of emission reduction with the Pigouvian rate, a tax would internalize the externality. The fully informed emission levy, in other words, discourages waste output directly. In an nth-best world with imperfect information, a risk-based tax uses a proxy for marginal damage to price chemical emissions. Output taxes, in contrast, would raise product prices for consumers, but would not link directly to pollution damage. The output tax would discourage economic output directly and waste output indirectly. An input levy would change relative prices to favor the use of cleaner resources. If the change were possible, this resource reallocation would encourage the substitution of less-costly inputs for their more-expensive counterparts. Along with the manufacture of chemical output, the use of harmful chemical inputs helps to degrade the environment. A regulator may establish a direct link between emissions and environmental damage. The link between inputs and damage is not as well defined.

ADDITIONAL TAX CRITERIA AND THE PORTFOLIO PROBLEM

Environmental tax policy addresses the portfolio problem. By pricing the externality, the tax is intended to increase efficiency and reduce pollution damage. But, in evaluating the appropriateness and effectiveness of tax regulation, public authorities must address additional criteria. From society's perspective, the benefits of creating pollution prices and reducing waste output may be offset by new administrative, compliance and distribution costs. The portfolio problem means that the regulator must address all policy impacts and weigh the overall benefits of regulation against the costs.

The administrator must also consider the evaluation criteria in terms of different tax bases. If input taxation would achieve the same environmental gains as an emission levy, but with lower costs, the input tax would be more appropriate. Large-scale chemical emissions release high levels of waste output into the air, water and earth. A policy maker could raise the price of chemical resources with an input tax or output with a product tax. But, with large-scale chemical releases, an emissions levy would tax the waste output directly.

Pollution taxes are unique when compared to other levies (such as those on sales and income). The goal of the former policy is to reduce the base in order to improve environmental quality. In other words, an environmental tax rate set to encourage pollution reduction will make the base insufficient for revenue generation. Over the long term, environmental taxes should not be good revenue-generating devices.

As chapter one explains, the additional tax criteria are equity, administration, compliance, revenue and environmental impact. In different pollution situations, a policy maker may assign higher weights to certain criteria. If a state or region is highly polluted, for example, the governing authority may be willing to assume high administrative and enforcement costs to ensure a steady

cleanup and stable revenue stream. In other instances, a tax policy could establish a regulatory mechanism to encourage the implementation of abatement equipment and pollution reduction over time, but not entail high implementation costs. While these tradeoffs are a part of the portfolio problem, the weight given to each criterion depends on circumstance. Chapter five argues that the risk-based policy could be implemented at the state level. In different regions, the chemical emissions and implementation costs would vary.

Equity

Many OECD countries conclude that the distribution consequences of environmental taxation leads to political controversy (OECD, 1995b, 75). Specific environmental policies may impact different parts of the same population unevenly (industries, income groups or regions.) It is possible to argue, for example, that the carbon tax may burden low-income households more than high-income households when measured as a percentage of income but not expenditures (Poterba, 1991). If it were implemented at the national level, moreover, different regions would bear a higher carbon tax burden in the United States (DeWitt *et al.*, 1991). The U.S. Department of Energy (1991) has shown that an increase in the gasoline tax of $0.25 per gallon would raise annual energy expenses more for upper income consumers than lower-income.

It is necessary to establish a framework for analysis to understand how environmental taxes designed to reduce chemical emissions would impact different groups. Four issues address the distribution consequences: (1) relevant costs and benefits, (2) formal and effective incidence, (3) baseline for comparison, and (4) relevant distribution dimensions (OECD, 1995b, 76-77). With the first issue, long-term distribution shifts occur that reflect changes in economic activity. In addition, short-run changes happen during the period of adjustment to the policy. The second issue involves the actual identification of those groups that bear the burden of the policy. Formal

incidence focuses on the one who makes the initial tax payment. Effective incidence focuses on the one who bears the ultimate financial burden. With the third issue, the effects on different groups will depend on the baseline for comparison. The fourth issue explains that different notions of distribution may be relevant, including the distribution effects on income groups, household types, businesses and households, types of firms or industries and regions (OECD, 1995b, 76-77). In the context of these equity criteria, the following discussion considers the possibility of using different environmental tax bases to address the chemical emission problem.

Normally, the poor and rich occupy separate neighborhoods. An emissions tax policy creates benefits that may flow to both parties. A fully informed emission or risk-based tax, for instance, would improve air and water quality everywhere when compared to a point in time that does not entail environmental taxation (the baseline). But pollution tends to be more serious in poorer neighborhoods. This situation results when heavy industry locates away from high-income areas and wealthier residents locate away from heavy industry. An emissions tax policy may therefore generate benefits that are more critical to the poor (Baumol and Oates, 1988, 246).

But equity analysis also entails costs. Although the physical quantity of pollution reduction may benefit poorer neighborhoods, the poor may not value these improvements more than others. While environmental policy could create short-term health benefits, higher levels of environmental quality could increase land values and drive up rents over time. As a result, higher living costs for poorer residents could offset the benefits of environmental improvement. Even though this price change could take years, changes in rent patterns, in the long term, may reduce the net benefits of the environmental policy (Baumol and Oates, 1988, 247).

From another perspective, emissions taxation could make communities pay the bulk of the cost of local policy in the short term. Firms may pass the tax on to consumers in the form of higher prices. But, if left alone, many individuals may achieve a desired level of environmental

quality given improvement costs. These individuals, for example, may insulate their houses from air pollution. From the point of view of these residents, regulation therefore imposes additional costs (Baumol and Oates, 1988, 249). These residents, however, would tend to have higher incomes. Those with lower incomes would not allocate money for individual environmental protection.

A critical feature of an emissions policy involves a likely uneven pattern of incidence. In the long term, fully informed emissions or risk-based taxes would impact certain industries more than others. For example, heavy polluters in a chemical industry could be forced to reduce output. In towns with heavy industry, policy cost could include job loss. Blue-collar households would suffer more than white-collar households. Even more, this loss could be exacerbated by the fact that emissions taxes cover regions unevenly. Industrial regions would feel the burden of higher unemployment more than unregulated areas (Baumol and Oates, 1988, 250).

Emissions-based taxation that regulates chemical emissions may enhance environmental quality for all, but may also impose relatively higher costs on the poor than the rich. In this sense, the fully informed emissions and risk-based taxes would have similar effects. Both policies would impose higher pollution prices on the emissions that pose a higher degree of risk to humans and the environment. In the short term, this would raise prices as firms pass the cost to consumers. In the long run, the policy could reduce employment in industrial areas.

Relative to an emissions-based tax, an input tax could lead to similar effects in terms of benefits from pollution reduction. Firms that use toxic chemical inputs would face higher input prices. In the short run, input taxation could lead to higher output prices. The communities with chemical manufacturing firms and industries that use regulated chemical inputs would feel a higher burden as the businesses pass along the costs. In the long term, industries that use a high percentage of toxic chemical inputs would have to change their production methods in adjusting to

the policy. The distribution of abatement costs, therefore, would tend to impact these industries the most. Ceteris paribus, the return on investment would be lower for these industries during adjustment to the policy, relative to the baseline in the absence of the tax, which may result in a loss of competitive position. (Of course, if the environmental tax replaced existing direct control, abatement costs could actually decrease.)

An output tax, in contrast, could impact different income groups throughout the entire population, depending on the extent of policy coverage. If the output tax covered all of the products that are deemed environmental harmful in SIC 20-39, for example, higher prices would affect a wide range of consumers. In addition, the tax could impact different regions more than others. But the policy would impact households more dependent on blue-collar workers in the long term. The benefit of the output tax is that it would discourage the consumption of dirty products, while the price would address the externality.

The degree to which the burden of the tax policy is borne by business or households depends on elasticities of supply and demand for the products coming from target industry. If supply is relatively elastic, but demand is relatively inelastic, the consumer will bear a higher burden. In contrast, if supply is relatively inelastic and demand is relatively elastic, the supplier will bear the burden. If the tax burdens the seller, the incidence may be shared between capital, labor, resource owners and other factors of production (OECD, 1995b, 76).

In sum, to reduce chemical emissions, environmental taxes on emissions, output or inputs would tend to burden blue-collar households more than other households. The benefits of environmental taxation would accrue to all those affected by environmental degradation, especially in the context of emissions-based policy. But, in the long term, the taxation of any base would impact industrial regions more than service sector areas.

If the pattern of tax incidence is not even, environmental policy should include means for redistribution. Environmental taxation, for instance, may be coupled with a policy of revenue substitution for industry or individuals to lower the burden. The revenues from environmental taxation could finance lower rates on income, property, payroll, capital or other taxes. In addition, the revenue could subsidize abatement technology or finance environmental programs such as recycling centers.

Administration

Different levels of government may administer environmental taxes. A main factor in the extent of jurisdiction is the dispersion of the pollutant.[43] A water effluent charge, for instance, may conveniently fall under local control. But the appropriate level of government to regulate a pollutant that transcends local boundaries is not clear. Should a state or national government operate a tax on particulates, for example? The answer for this pollutant and others depends on a number of other administrative issues: (1) pollution identification, (2) the ability to administer, oversee and adapt the policy, (3) revenue collection mechanisms, (4) enforcement, and (5) the pollution source as opposed to its destination. Graetz *et al.* (1989) note that, for taxation in the United States, multiple resources are allocated to tax collection, but there is a lack of coordination between different levels of government. With some tax policy coordination, "Substantial opportunities exist for significant efficiency gains for taxpayers and government alike" (Graetz *et al.*, 1989, 311). While different levels of government may oversee an environmental policy, national, state, and local governments are not in competition with one another for jurisdiction.

[43] The determination of dispersion alludes to the importance of the ability to observe pollution. If regulators identify pollution dispersion, the level of government jurisdiction follows; however, this identification may be difficult for regulators.

The first administrative issue involves pollution identification. With chemical emissions, the governing authority must identify target pollutants by determining that they are harmful and worth regulating. The idea is to regulate the emissions for which the benefits of pollution reduction exceed abatement costs. Chemical emissions in the TRI that pose a low degree of risk to humans and the environment are not worth regulating. In other words, the costs of pollution abatement could outweigh any gains from pollution reduction.

With the second concept, the implementation of an emissions tax entails research costs. Specifically, the policy authority must have an understanding of the relationship between emissions and environmental damage if a tax is to influence behavior. The benefit of an emissions tax system entails the ability to price according to approximate environmental damage from individual pollutants. But the cost includes the research expense of obtaining damage information and the calculation of an appropriate tax rate for each pollutant.

The risk-based tax system establishes logic for policy using available damage research. This system eases the process of tax implementation in the face of information limitations on marginal damage functions. By assigning risk units to target chemical emissions, the system taxes each effluent according to a relative hazard value. Since the policy uses the risk unit as base, the administrator levies one tax rate. Additional work for the administrator involves the calculation of the risk units associated with one unit of each chemical after choosing the numeraire, and the research costs concerning the evaluation of potential damages from individual pollutants.

To levy output and input taxes in the context of the chemical emission problem, the administrator would have to employ available price elasticities on products stemming from regulated industry and target chemical inputs. With high demand elasticity, agents would switch to alternative products after the implementation of an environmental tax intended to induce behavioral response. Among substitute commodities, policy makers should levy the tax according to pollution

potential. For example, product disposal threatens the environment or contributes to garbage-dump buildup. The levy on disposable cameras in Belgium encourages the consumption of reusable products (OECD, 1994a). The same policy could apply to products stemming from industries that emit chemical pollutants. While price elasticities are available for many products, administrative problems would entail the choice of products to tax in SIC 20-39 and the specific tax rates to somehow reflect pollution damage.

To levy an input tax to regulate chemical emissions, the regulator must consider the degree of substitutability between chemical inputs and the environmental objective. The own-price elasticity of demand for a particular input includes output and substitution effects. The former effect entails the reduction in output from a higher input price. The latter effect is the substitution to other inputs. Regulators want to encourage the switch to cleaner inputs in the production process. Input taxes would be designed discourage the use of the more harmful chemical inputs. With available damage information, input taxes could price according to relative damage values. The problem, however, is the relationship between the use of chemical inputs, emissions, and damage. An individual tax rate may not reflect environmental damage without information on a clear link between input quantity, level of emissions and damage, which would be difficult to gauge for individual production processes under regulation.

The point is that, with output and input taxes, government must estimate price elasticities of dirty resources or commodities. With the large-scale chemical emission problem, the governing authority wants to discourage waste output directly. The administrative benefit of levying an emissions-based tax is that the regulation would price the pollution externality directly. Emissions-based levies may adapt smoothly to changing conditions, including new information concerning the relationship between emissions and damage.

To estimate the full effects of the risk-based levy, the governing authority would observe the changes in business activity to see how firms adjust their pollution mix. Would target industry reduce the flow of the most harmful emissions as the risk-based system intends? The answer depends in part on the ability of firms to substitute cleaner emissions for their dirtier counterparts—i.e. change their production processes. These "emissions elasticities" would benefit regulators before the implementation of the risk-based tax, but are not available on a large scale (see, for example, Davis *et al.* (1994a) for a discussion of product and input substitution). Not knowing the substitutability between target chemical emissions does not help the policy maker in the implementation of risk-based taxation.

The third administrative issue concerns the collection mechanism. Minimizing collection points reduces administrative cost. Authorities must gather emissions data from firms, establish a per-unit tax rate and develop a mechanism for policy implementation. If an existing regulatory mechanism is used to administer tax policy instead of a new system, cost savings occur. With output and input levies, an excise charge is necessary to price the item. For these two instruments, an established taxing mechanism exists. Since firms in SIC 20-39 must report annually to the TRI, correspondence between these companies and government already exists. Tax payment from a per-unit emissions charge could accompany the reporting of TRI pollutants.

With the fourth concept, administration must allocate resources for enforcement. In response to any form of emissions taxation, firms have the incentive to underreport. As a result, the governing authority must establish a plan to ensure that reported emissions equal actual levels. Enforcement may entail a periodic check of firms under policy jurisdiction. If an emissions tax prices pollution according to damage, firms will have the incentive to underreport the chemical emissions that pose a high degree of risk to humans and the environment. As a result, the risk-based tax system may entail relatively high enforcement costs as firms attempt to underreport the

emissions with larger hazard values. In contrast to emissions taxation, enforcement costs are lower for output and input levies since it is easier to count the number of products sold or resources used.

In terms of the fifth administrative issue, imperfect observability in terms of the damage effects of pollution has an impact on regulation. In other words, the place where pollution impacts humans and the environment may be different than the source of the pollution. This possibility complicates policy jurisdiction in the case of emissions that cross national, state or local boundaries. With emissions taxation, a case may be made for the implementation of a policy on the national level in order to incorporate polluters and damage effects.[44] For emissions that cross national borders, some form of cooperation must occur between relevant countries to internalize the externality. As policy jurisdiction increases in scope, administrative costs rise. However, these same issues do not apply to output and input levies. Presumably, the same person or company that purchases the item and is taxed performs the activity that pollutes the environment.[45]

An advantage that emissions-based taxation has over its input and output counterparts concerns the ability to adjust rates to reflect the damage from waste output. Regulators may adjust tax rates upward if they determine that chemical pollution is inflicting an increasing degree of damage on humans and the environment. A direct correspondence between damage valuation and incentive policy would therefore be established. In the case of chemical emissions, it would be more difficult to link the tax payment directly to environmental degradation if the tax base is a resource input or commodity output.

[44] However, states may have an easier time applying emissions taxes that take into account the environmental and population factors that help determine damage effects.
[45] Examples include the disposal of batteries or the emission of carbon dioxide from energy use.

Compliance

Monitoring represents an important cost. As chapter one explains, the Emergency Planning and Community Right-To-Know Act requires manufacturers in SIC 20-39 to report the emissions of over six hundred toxic chemicals into the air, water and land. Monitoring cost depends on emission characteristics, which includes flow, concentration and stability. The cost also depends on available abatement technology. Cost decreases as technology improves. The availability of measurement technology represents an important factor for the success of an emissions tax policy. Monitoring systems form the basis for taxation and the evaluation of policy effectiveness. The systems must calculate the emissions of individual pollutants. In contrast to this regulation, individuals and firms do not monitor emissions to pay output and input taxes. Monitoring costs for the individual firms are therefore greater with an emissions-based tax than an output or input levy.

In addition to monitoring expenses, firms must allocate time and resources toward legally complying with the law and minimizing tax payment. Record keeping, learning about tax law, preparing for tax payment and sending the tax payment are activities that the firm must undertake when complying (Graetz *et al.*, 1989, 322). It is not clear if these expenses are greater for output, input or emissions taxation.

"Effective" environmental levies that induce behavioral response may lead to legal, tax avoidance activity. In the case of emissions-based taxation, the firm has the incentive to reduce the quantity of each pollutant under regulation in order to lower tax payment. The firm must allocate time and resources to the consideration of ways to alter the production process to reduce tax payment.

Research and abatement costs could impact a firm's production decisions. In this context, compliance costs for output and input levies entail the time and resources allocated toward

estimating a less-expensive payment alternative. For the firm, this activity may include research and production costs, and price calculations for regulated resources and their alternatives. For the individual, compliance entails the information cost necessary to find an alternative to the taxed product. Compared to emissions-based environmental taxation, output and input levies would entail lower enforcement costs for the administrator.

Environmental tax payment is lower for a firm if it can decrease its tax base, given a constant rate. As a result of this incentive to reduce the base, firms may choose to evade taxation. Evasion entails the underreporting of tax liability by illegal means (Alm *et al.*, 1990, 603). Since they monitor their own emissions, firms could establish a method of providing lower emission figures to government. In the absence of effective enforcement, this activity would lower tax payment. Risk-based taxation would provide a greater incentive for evasion of the most harmful chemical emissions, as these are priced at higher rates. For a particular firm, evasion would depend on the tax rate, the penalty (if any) of underreporting, and the probability of detection. In the case of emissions taxation, higher penalty rates and detection probabilities would discourage evasion. The same causal factors would not hold true with output and input levies, as tax evasion would be more difficult to undertake. As with avoidance activities, emissions-based taxation could lead to higher levels of evasion in the context of the large-scale chemical emission problem.

Revenue[46]

To analyze the portfolio problem and compare the different tax bases, five interwoven revenue issues are important. The first is yield, or the level of revenue that each tax bases raises. The second issue, adequacy, determines whether a tax can support government expenditure requirements. Third, short-term stability makes clear how a levy changes in relationship to the

[46] This discussion of revenue draws on the analysis in Murray (1997).

business cycle. The fourth concept, elasticity, concerns the change in revenue due to a change in the tax base. The fifth issue, long-term base behavior, is the potential for the base to decrease over time and enhance environmental quality. Since it is crucial to the analysis of environmental taxation, this last issue is addressed in the next subsection.

The revenue yield of emissions taxes is currently not as high as environmental output and input levies. As a result, only a small amount of revenue information exists. To put this first revenue issue in context, consider the following information for environmental emissions, output and input taxes. Table 5 lists tax revenue as a percentage of total revenue in selected countries. In the Netherlands, Norway, and Denmark, the revenue from taxes on air and water pollution and CO_2 levies has increased over time. Still, the revenue is an insignificant part of the public budget.

Table 5
Emissions tax revenue as a percentage of total revenue

	1980	1985	1990	1991	1992	1993
Netherlands						
Air pollution tax	0.04	0.05	0.29	0.37	0.56	0.58
Water pollution tax	0.53	0.59	0.62	0.59	0.63	0.65
Noise pollution tax	----	0.06	----	----	----	----
Norway						
CO2 tax	----	----	----	0.25	0.59	0.68
Denmark						
Tax on waste	----	----	0.10	0.12	0.11	0.12
CFC tax	----	----	0.01	0.00	0.00	0.00
CO2 tax	----	----	----	----	0.36	0.76

Source: Organization for Economic Cooperation and Development. 1995a. *Environmental Taxes in OECD Countries*. Paris: OECD.

Output levies with environmental implications raise a significant amount of revenue, at least when compared to emissions taxes. Some output levies may also be input taxes. The revenue from the petrol excise remains relatively constant over time. These energy taxes serve as important revenue instruments. Input taxes with environmental implications raise revenue in many

countries. Tables 6 and 7 list some revenue figures from three countries. The levies in Table 7 are increasing as a percentage of total revenue as the countries are relying more on these environmental taxes.

As Tables 5 through 7 help to make clear, taxes on emissions are not consistent revenue generators for the public budgets in the selected countries. Adequacy, the second revenue issue, is therefore a useful concept for the comparison of the four tax bases. With the chemical emission problem, we have to consider that the regulator would implement a *system* of fully informed emissions or risk-based taxes. These systems would create a larger revenue stream than a government would receive with taxes on individual pollutants. Chapter three shows, for example, that a significant amount of revenue could be generated at the state level with the inclusion of more than one hundred pollutants in a risk-based tax system.

But emissions taxes are not intended to support the revenue needs of public budgets. On the other hand, output and input levies contribute significantly to the budgets and expenditure needs of many countries (OECD, 1995a). However, emissions-based taxes that regulate over 100 chemical emissions (like in chapter three) could support short-term expenditure needs. Since the tax base is large, the regulator would not tax the revenue source away. A key difference exists, therefore, between the proposed policy of environmental taxation to regulate chemical emissions and existing regulations that target other pollution problems: the relevant chemical emission problem is large-scale and offers a wide base consisting of a number of individual pollutants. In sum, emissions-based taxes that regulate the chemical emissions under consideration would generate a revenue stream that is not insignificant, as would output and input levies.

Table 6
Output tax revenue as a percentage of total revenue

	1980	1985	1990	1991	1992	1993
Netherlands						
Excise on petrol	1.83	1.70	1.61	1.57	1.74	1.89
Petroleum product levies	----	----	0.07	0.05	0.04	0.05
Norway						
Excise on petrol	1.72	1.70	2.31	2.47	2.86	2.78
Electric energy	1.13	1.04	1.12	1.05	1.06	1.12
Denmark						
Excise on petrol	2.18	1.49	1.45	1.36	1.36	1.29
Excise on electricity	0.72	0.67	1.11	1.10	0.95	0.81
Finland						
Liquid fuels tax	4.24	3.30	2.45	2.82	3.14	3.83
Excise on electricity	0.55	0.74	----	----	----	0.30

Source: Organization for Economic Cooperation and Development. 1995a. *Environmental Taxes in OECD Countries*. Paris: OECD.

Table 7
Input tax revenue as a percentage of total revenue

	1980	1985	1990	1991	1992	1993
Netherlands						
Mineral oil excise	0.42	0.35	0.77	0.77	0.82	1.06
Norway						
Oil and gas excise	2.78	4.99	2.87	2.96	2.64	2.52
Mineral oil excise	0.08	0.04	0.36	0.64	0.61	0.52
Denmark						
Duty on coal	----	0.07	0.22	0.22	0.18	0.17

Source: Organization for Economic Cooperation and Development. 1995a. *Environmental Taxes in OECD Countries*. Paris: OECD.

As revenue instruments, governing authorities intend environmental taxes to contribute to public budgets or specific environmental projects (Appendix 1). With traditional revenue-generating devices such as sales and income taxes, policy makers intend to minimize the risk, i.e. revenue instability. The potential for instability exists with environmental taxation. This is the third revenue issue. Specifically, pollution tax revenue will be tied to the business cycle as firms adjust their production patterns during downturns. For example, a recession reduces output sold and therefore the use of resource inputs. This effect lowers tax revenue during the downturn of the business cycle. In addition, as emissions are a function of output, the revenue generated by the emissions base will also depend on production decisions in correspondence to the business cycle. If an item such as gasoline is a necessity, the revenue stream may remain independent of downturns in the economy. In general, it is not clear which tax base, output, input, emissions, or risk units has the advantage in terms of revenue stability in the context of the chemical emission problem. It is important to note, however, that if the environmental tax policy is intended to reduce pollution, the revenue stream will not be stable over time anyway as the tax base decreases.

With the fourth of the revenue criteria, a tradeoff exists between revenue stability and elasticity: "In general, the most stable tax base is likely to be inconsistent with an elastic tax base" (Murray, 1997). To enhance environmental quality, the regulator wants polluters to switch to cleaner production alternatives. As a result, the implementation of a pollution tax is intended to decrease the tax base over time. How will revenue receipts change as this happens? Pollution levies could influence behavior at the margin (as with a gasoline tax) or seek to tax away the base and significantly enhance environmental quality. With certain emissions or resources in the production process, however, an increase in tax rate will not reduce the use of the input. Examples include products or resources necessary for the production of certain outputs. While producing stable revenue streams, the taxation of these latter items would not significantly help to enhance

environmental quality, unless the tax rate is significantly high. The governing authority could raise the tax rate on inelastic bases in order to increase the revenue stream and continue pressure on firms to reduce pollution. If applied on a large level in SIC 20-39, taxes on output, inputs, or emissions to reduce chemical pollution will all regulate elastic and inelastic bases.

Environmental Impact

From the perspective of the regulator, the long-term possibility of environmental improvement represents the most important criterion. Given the chemical emission problem, the effectiveness of a particular form of taxation depends on the link between pollutant and tax source. The closer the link between source and pollutant, the greater the influence of the levy in discouraging the harmful activity. Emissions taxes, for instance, help to enhance environmental quality when they elicit behavioral response. If firms implement abatement equipment, end-of-pipe technology or change the production process or output composition, environmental quality improves. The tax may also help to impact environmental quality if it serves as an important part of production expenditures. At the present, only a few environmental taxes create incentive response (Appendix 1). In the context of the chemical emission problem, the emissions levy would tax the source of pollution directly by pricing individual pollutants.

The impact of output taxes in the context of the large-scale chemical emission problem would depend on individual behavior. If individuals consume fewer dirty products from the production processes and limit disposal, the long-term prospects for environmental quality would improve. Input levies, moreover, would help to enhance environmental quality if producers have the incentive to change their resource mix and employ cleaner inputs.

But with large-scale chemical emissions, an output levy would raise product price, while an input levy would tax the chemical inputs. Presumably, an output or input tax could regulate industries in SIC 20-39 that report to the TRI. A major problem with either of these policies would entail the administrative difficulty of setting the tax rate on individual products to value environmental damage from chemical emissions. The CCPCT index provides information that relates emissions to damage, but not the relationship between inputs, output, emissions and damage. While discouraging product consumption, output levies would not tax the pollution source as effectively as an emissions-based tax. An input tax entails certain advantages when compared to the output tax. Given that the regulator decides that firms are allowed to use certain chemicals, an input levy may serve to discourage the use of the most harmful. While potentially effective in application, the problem of linking the input tax to pollution damage still exists.

With risk-based taxation, a policy maker may set the rate on a risk unit so polluters have the incentive to implement abatement technology and alter the emissions mix. (This is no different than a system of fully informed emissions taxes.) Over the long term, the extent that risk-based taxation helps to enhance environmental quality depends on the level of the tax rate and the behavioral response among polluters. The risk-based tax will be effective to the extent that the tax represents an important part of the expense budget of regulated firms.

This last criterion addresses the concept of the temporal dimension or how a tax helps to enhance environmental quality over time. In a changing environment, a tax must be adaptable to different conditions. If a tax is rigid in its application and cannot adjust to new information concerning technology, individual behavior or environmental damage, the instrument would not serve as an appropriate policy. The risk-based tax is designed to be flexible as new information emerges, especially concerning the relationship between emissions and damage.

THE PREFERENCE FOR DIFFERENT TAX BASES

With the different evaluation criteria, certain of the four environmental taxes (fully informed emissions, output, input and risk-based) have preference over the others. In general, the levies apply to different pollution problems. Even though the evaluation criteria are the same for each base, implementation is different. *In the context of the chemical emission problem*, how would the choice of each environmental tax base fair in comparison to the others? To answer this question, Table 8 reproduces the criteria checklist of Table 4 and shows when one tax is superior (+) or inferior (-). The comparison of regulations is made between (1) the fully informed emissions tax (column two) and the output and input taxes, (2) the risk-based tax (column three) and the output and input levies, (3) the output tax (column four) and the risk-based and input taxes, and (4) the input tax (column five) and the risk-based and output levies. This procedure is chosen for the following reason: if damage information is known for target chemicals, the fully informed emissions tax would serve as the appropriate policy and the risk-based tax would be irrelevant. As a result, it is necessary to make an information assumption concerning damage functions. When the fully informed emissions tax is compared to the output and input levies in the context of the chemical emission problem, full damage information is assumed to exist. In contrast, when the risk-based levy is compared to the output and input taxes, we may assume that full damage function information does not exist. Finally, when comparing the output and input levies, we assume imperfect damage information. In many of the following situations, it is impossible to make a determination (blank space.) An explanation of the choices follows Table 8.

With the first criterion, an emissions-based tax would regulate the chemical emission source directly which would facilitate a greater efficiency impact when compared to output and input levies. Research by Bovenberg and de Mooij (1994) and Parry (1995) has shown that tax-interaction effects outweigh revenue-recycling effects with environmental taxation under certain

Table 8
Summary of evaluation criteria

	EMISSIONS-	BASED LEVIES		
Criteria	**(1) Fully informed emissions tax**	**(2) Risk-based tax**	**(3) Output tax**	**(4) Input tax**
Efficiency	+	+		
Tax-interaction effect				
Revenue-recycling effect				
Equity				
Administration	-	-	+	+
Adaptability	+	+	-	-
Collection				
Enforcement	-	-	+	+
Source/destination	-	-	+	+
Compliance	-	-		
Monitoring	-	-	+	+
Avoidance	-	-	+	+
Evasion	-	-	+	+
Revenue				
Yield				
Adequacy				
Short-run stability				
Elasticity				
Environmental impact	+	+		

assumptions. These analyses, however, include product taxes and not emissions-based levies. The latter regulations may entail the same efficiency losses in a policy of revenue substitution. As a result of a lack of research on the emissions-based taxes, it is impossible to conclude anything about these latter efficiency effects.

A distinction between the effects of taxing the different bases is clearer for administration than equity. The implementation of an emissions-based levy would incur higher enforcement costs than a tax on output or inputs. A difference exists, moreover, between the source of the emissions and the destination. This difference would lead to important costs in estimating environmental damage. The source and destination of outputs and inputs is not as ambiguous. Conceivably, the emissions-based levies could improve environmental quality for everyone. But chapter two makes clear that an environmental tax could burden a manufacturing region with blue collar workers more than another area. To address the chemical pollution problem, it is unclear which tax is more equitable.

Compliance costs would differ for emissions-based, output and input taxes. Specifically, monitoring costs and incentive for evasion and avoidance under emissions-based taxation exceed the costs that would exist for product and input levies. The revenue comparison between bases, however, is inconclusive since revenue simulations have not been made. Finally, the emissions-based levies have a greater potential for improving environmental quality since the tax base is the pollution source.

The portfolio problem means that tradeoffs occur when attempting to enhance environmental quality with a particular tax base. Ceteris paribus, if a regulator is concerned with efficiency/environmental gains and is willing to incur administrative and compliance costs, emissions-based taxation serves as an appropriate policy. On the other hand, if the policy maker is concerned with keeping administrative and compliance costs low, environmental output and input

levies could represent suitable alternatives. If the pollution problem warrants the reduction of the flow of a number of chemical emissions, a policy choice exists between fully informed emissions and risk-based taxes. In the absence of fully informed damage information, the policy maker must recognize with risk-based taxation that distinct administration and compliance costs will persist over the life of the policy. In addition, tax system efficiency losses will result if the governing authority decides to attach an additional policy of revenue substitution. If the benefits of improved environmental quality, revenue recycling and gains in equity outweigh the costs of any tax-interaction effects, equity losses, administration and compliance, then the tax would increase the welfare of society.

REVENUE, BASES, AND RATES

The four environmental taxes from this chapter would generate revenue. The following equations show that revenue equals base times rate.

(1) Output tax: ***Revenue = (base)(rate)***, where the base is a product and the rate per-unit.

(2) Input tax: ***Revenue = (base)(rate)***, where the base is a product from another production process and the rate is per-unit.

(3) Fully informed emissions tax: ***Revenue = (emissions)(rate)***, where the per-unit rate is levied on a quantity of emissions and is a function of marginal damage.

(4) Risk unit tax: ***Revenue = (risk units)(emissions)(rate)***, where the risk units equal the damage value[47] of a unit of the emission, and the rate is per one risk unit.

The latter tax offers policy makers a feasible opportunity to regulate a set of emissions with one base. To illustrate this concept, consider that each emission is unique, but monitors measure each according to some unit, for example, in pounds. By transforming one pound of each

[47] See "The Computational Algorithm" subsection in chapter three.

emission into a number of risk units, a policy maker may levy a single tax on one risk unit and encompass an entire set of chemical emissions:

$$Revenue = t_{ru} \Sigma (RU_i)(e_i),$$

where t_{ru} = tax on one risk unit, RU_i = number of risk units of one unit of emission i, and e_i = quantity of emission i. Different tax rates would raise alternative levels of revenue and provide varying degrees of incentive for pollution reduction.

THE CASE FOR RISK-BASED ENVIRONMENTAL TAXATION

As a result of the indirect link between an output or input levy and chemical pollution damage, the regulator may want to levy an emissions-based tax to address the externality. With information limitations prohibiting the implementation of a system of fully informed emissions taxes to regulate the TRI chemicals, risk-based taxation presents a feasible alternative. The risk-based policy links directly to the pollution problem, but entails important administration and compliance costs. As chapter three explains, the risk-based tax uses the CCPCT index of damage values for individual chemical emissions, raises the price of the pollutants with high risk values and levies low rates on emissions that pose little risk to society.

CHAPTER 3

RISK-BASED TAXATION CONTRASTED WITH FULLY INFORMED REGULATION

Risk-based taxation provides a method to account for marginal damage from specific chemical emissions. By using a ranking methodology, the framework establishes logic for environmental tax policy. Higher taxes apply, for example, to chemicals with larger risk values. The risk-based system targets a set of emissions and attempts to create a behavioral response among polluters. In order to explain these concepts and the system's framework, chapter three discusses marginal damage function research, the methodology underlying the chemical emission ranking system, the risk unit and implications of risk-based taxation. The marginal damage focus of chapter three is relevant only to a risk-based environmental tax. The comparison in this chapter between direct and indirect measures of damage shows that the latter techniques are applicable to an environmental tax *system* that encompasses a number of chemical emissions. The extent that the risk-based system implements accurate prices depends on its correspondence to fully informed regulation.

MEASURING THE MARGINAL DAMAGE FUNCTION

A causal relationship exists between emissions and certain damages suffered by society. The damages may take the form of reduced crop yields, diminished enjoyment of the outdoors, increased prevalence of mortality and disease, more frequent replacement and maintenance of equipment and many other less-identified losses. A decrease in pollution levels will therefore bring

about a corresponding reduction in these damages. The benefits of lower levels of damage equals the difference in damages before and after reduction.

Chemical emissions from industries affect ambient air quality, water quality and the earth above and below ground. These pollutants may or may not undergo physical changes as a result of meteorological factors. Chemicals may persist in the environment in their current state. Economic damages can be determined as a function of changes in environmental quality through various techniques such as declared willingness to pay, surveys of property values, litigation awards or legislative decisions. Policy decisions often occur before the measurement of damages. But effective policy takes into account the monetary valuation of changes in environmental quality. For a particular pollution problem, this adaptive process becomes more accurate over time as researchers and regulators make successive refinements of each step.

Nature of Damage Functions

A damage function shows the relationship between exposure to certain pollutants and damage to a target population or environment. Many variables that are difficult to model enter the estimation process. As a result, the formulation of individual damage functions entails a degree of uncertainty. The damage of individual pollutants is expressed in monetary terms by assigning a dollar value to the effects of the physical or biological damage. The monetary valuations are often taken from surveys. Expressed as "dosage" or "dose," scientists measure exposure in terms of ambient concentration levels and exposure time. The time and concentration that the individual or environment has been exposed constitutes the dosage, while dose means the portion of the dosage that creates the observed damage.

To report a damage function, a researcher must specify the particular pollutant under consideration, the dose rate, the effect and the population and/or environment at risk. The rate that ambient quality changes with time—dose rate—influences the severity and nature of the resulting effect. Exposure over a long time period to a low concentration level may produce chronic diseases that develop slowly and are difficult to detect or impact the environment in such a way that inhibits reproduction and growth. Short-term exposure to high concentration levels, however, may produce acute symptoms with quick response and ready detection in humans, and also chronic and delayed effects. With the environment, short-term exposure to different chemical emissions could impact flora and fauna in a number of ways, depending on persistence, bioaccumulation and a number of other factors (Hershaft, 1976, 993).

The characterization of the population and environment at risk allows researchers to calculate the total damages that result from a particular level of exposure. With humans, this calculation also enables investigators to alter the results to reflect the influence of intrinsic (age, sex, race) and extrinsic (education, income, and occupation) variables. With the environment, this characterization permits policy makers to allocate resources to specific areas of concern.

The Damage Function Approach

Damage functions establish an impact-pathway from a pollutant to a particular endpoint[48] in the environment. Table 9 shows that five stages characterize the damage function approach (DFA).[49] Stage one names production activities and estimates their emissions and other residuals.

[48] Endpoints are the different impacts to measure the damage from pollution. For example, the CCPCT algorithm uses oral, inhalation, and carcinogenic endpoints for the human health effect. The algorithm uses other endpoints for the environmental effect and exposure potential. Common endpoints in the ranking methodologies are carcinogenicity, mutagenicity, genotoxicity, non-carcinogenic, aquatic toxicity, mammalian toxicity, development/reproductive toxicity, physical hazard, plant toxicity, terrestrial non-mammalian toxicity, and general ecological effects (Davis *et al.*, 1994b, 39).

[49] This discussion of the damage function approach is adapted from ORNL/RFF (1994, 4-45).

Specifically, this stage identifies the production process of target industry, resource inputs, production technology, abatement technology and location.

Table 9
The damage function approach

Stage	Purpose
One	Name activities and estimate emissions
Two	Model dispersion and changes in concentrations of pollutants
Three	Quantify physical impacts of emissions and residuals
Four	Translate physical quantities to economic damages and benefits
Five	Distinguish externalities from internalized costs and benefits

The second stage recognizes that emissions may alter the ambient concentration levels of pollution in surrounding areas. In this stage, transport models of pollution dispersion calculate the changes in ambient concentrations by location. Different meteorological conditions such as wind patterns, sunlight, time of day, geographical location and synergy between pollutants impact the results. Strong winds, for instance, help to disperse a pollutant and decrease its concentration in the atmosphere.

Stage three quantifies the physical impact of emissions and residuals with the application of dose-response functions. These functions entail a number of techniques to measure the impact of pollution on different endpoints. Dose-response functions require data on the response of humans and the environment to pollution through toxicological, clinical, epidemiological, field or laboratory investigations. Toxicological experiments involve the deliberate exposure of animals to chemicals in controlled environments. Clinical studies derive information from observation of exposed humans in hospitals. Epidemiological investigations evaluate the effects of pollution on segments of the target population to assess the extent of the effect. Field observations are similar

to epidemiological studies, but consider the effects of pollution on vegetation, animals and materials. Laboratory studies take a controlled approach to determine the effects of pollutants on plants and materials (Hershaft, 1976, 993-994). With epidemiological and field studies, scientists must remove the influence of intrinsic and extrinsic variables responsible for observed effects to achieve worthwhile results. These investigations show an association between pollution exposure and effect, suggesting a causal relationship. After scientists establish this relationship, they may test the association with laboratory and toxicological studies.

Epidemiological investigations require an intense effort to evaluate the damage that corresponds to a certain level of exposure. Because of the difficulty gathering data, research may establish only one point toward the formulation of a damage function. But additional studies may contribute further points if the conditions of investigation are similar to the original experiment. As a result of this difficulty, damage functions that model health effects are often sketchy (Hershaft, 1976, 994).

In contrast, damage functions that model the effects of pollution on vegetation, animal health and equipment do not experience the same level of difficulty. Scientists have a better opportunity to conduct controlled experiments. Favorable conditions prevail as a result of the ability to identify the target environment.

The fourth stage of the damage function approach translates the physical damage caused by the emissions of individual pollutants into an economic valuation. Researchers use different valuation techniques to quantify pollution damage, including contingent valuation (CV) and conjoint analysis. A method of contingent valuation calculates value by determining peoples' willingness to pay for changes in environmental quality. A survey technique, CV asks individuals how much they would be willing to pay for an improvement in environmental quality and specifies the payment mechanism. Statistical analysis then determines a value for a specific change in

environmental quality based on the survey data (Kahn, 1998, 101-106). Conjoint analysis, in contrast, does not specify a tradeoff between environmental quality and money. This technique asks individuals whether they would prefer different bundles of environmental quality at different costs. Each bundle possesses different aspects of environmental quality, such as a degree of visibility, water quality, and air quality. With statistical analysis, the tradeoff between money and changes in environmental quality is determined, although the individual taking the survey does not perceive the ranking as a tradeoff (Kahn, 1998, 106-107).

Stage five determines whether the economic values of the changes in environmental quality are externalities or already internalized by business operations or policy. In the presence of pre-existing policy, firms may already take into consideration the effects of pollution on humans and the environment. In the absence of regulation, however, firms raise the social cost of production with pollution without internalizing the cost. Stage five is highly subjective.

DIRECT MEASUREMENT EXAMPLES AND RESULTS

Early damage function studies evaluated, among other things, the effects of sulfur dioxide on human mortality, oxidants on vegetation and sulfur dioxide on the corrosion of zinc. In the 1970s, researchers implicated sulfur dioxide as one of the first pollutants to aversely affect human health. Buechley (1973) and Glasser and Greenburg (1971) studied the relationship between daily SO_2 levels and mortality rates in New York City and found a positive correlation (and an almost linear relationship). They found that mortality rates increase as SO_2 levels go up. Although the researchers did not establish pollution thresholds, they identified situations where changes in the mortality rate appeared both above and below the mean death rate, depending on the level of SO_2.

These early studies provided a rudimentary estimate of the consequences of a particular pollutant and established groundwork for future research. But problems existed with the research.

First, the damage function attempted to correlate daily levels of SO_2 with mortality rates, but lag times existed for the former that affected the latter. Second, the research did not identify the causes of death or the general state of health of the population. Third, the damage function did not provide any insight into the many days of discomfort that individuals must have experienced preceding death.

In a different study in the late 1960s and early 1970s, the EPA identified high oxidant levels in rural areas in many states, confirming that the 0.08 parts per million standard was being exceeded during summer. At these ambient levels, researchers found high crop losses. Heck *et al.* (1968) identified a damage function for a sensitive variety of tobacco called Bel-W_3, which exists in the southeast. They found a positive correlation between exposure to oxidants and leaf injury. As the concentration and length of exposure increased, the injury to the tobacco leaf went up.

Since data on the rural levels of oxidants were scarce, the results are taken with caution. The relationship posited by the damage function means that a decrease in the level of oxidant would reduce leaf injury. While helpful, this result does not come without caveats. First, the study did not identify the long-term effects on leafs of low exposure levels. Second, the research did not identify how the age of the plant affected the results. Finally, the damage function did not consider other important factors such as temperature, sunlight, or relative humidity.

In other research, Haynie and Upham (1970) formulated the relationship between sulfur dioxide, relative humidity and the corrosion rate of zinc. They undertook this study since metal corrosion constitutes the most important form of materials damage that results from air pollution. Companies use steel more than other metals. In many situations when steel is exposed to the atmosphere, the metal is galvanized with zinc. When the zinc coating dissipates, the steel tends to corrode faster. Humidity, temperature and SO_2 influence the corrosion process. Haynie and Upham carried out controlled experiments and expressed the corrosion rate of zinc as a function of

an atmospheric factor that consisted of a level of SO_2 modified by relative humidity. With their positively sloping damage function, Haynie and Upham found that a reduction in SO_2 levels should reduce the zinc corrosion rate and increase the life expectancy of steel.

Many of the early damage function studies formulated helpful relationships between pollutants and effects on humans, vegetation and equipment. An underlying problem, however, consisted of a lack of experimental data. In addition, the studies were place-specific and difficult to generalize in most cases. Unfortunately, this problem persists today. While the process of formulating damage functions has become more sophisticated, the process is still evolving.

The ORNL/RFF Fuel Cycle Study

For over two decades, researchers have developed damage functions to show the effects of pollutants on human health, animals and plants in the environment and equipment. In addition to the early damage function examples of the previous subsection, researchers have shown that air pollution is correlated with the incidence, prevalence and severity of various human disorders. Excess deaths from emphysema, chronic bronchitis and lung cancer, for example, relate to elevated levels of asbestos fibers, radon emissions and particulate matter. In addition, damages to the ecosystem include the effects of soil degradation and contaminated irrigation water on crops. Impaired air, water and soil quality, moreover, have impacted nature's biodiversity in countless areas.

The fuel cycle study conducted by Oak Ridge National Laboratory (ORNL) and Resources for the Future (RFF) establishes a rigorous method to estimate the benefits and damages for each fuel cycle of electricity generation. The report takes an "investment strategy" approach by

analyzing the change in the base level environmental and health conditions that result from an addition to an electricity generation site's capacity.

The study also develops an economic framework that applies external cost estimates in a policy setting. The report estimates the marginal consequences of a fuel cycle that creates spillovers at certain locations. The study estimates the impact from the coal fuel cycle on crops and suburban landscape, livestock, timber, commercial fishing, recreational fishing, hunting, recreation and biodiversity. The following discussion addresses a selection of these impacts.

The ORNL/RFF (1992) study adopts the impact pathway, damage function approach. The DFA begins with the fuel cycle. The cycle consists of primary resource extraction, transport of materials and resources, conversion and processing, end-use services and disposal. The study concentrates on each stage except end-use services. The ORNL/RFF report uses two sites for comparability across fuel cycles: one in east Tennessee (southeast) and the other in New Mexico at the Four Corners Region (southwest.)

Each stage in the fuel cycle implies a limited number of technological and abatement options. With these constraints, each fuel cycle creates a level of emissions, identified by the first step in the DFA. By disaggregating the emissions and quantifying the rates, the next DFA step entails the use of transport modeling and the physical characteristics of the sites to quantify the changes in concentrations, dose, and resource responses. These changes represent a set of physical impacts. Monetary valuation techniques then assign dollar values to the physical impacts. The last step distinguishes between external costs and damages by considering how much of the damages are internalized. The study reports specific annual damages that result from production and emissions, but does not calculate damage functions over a range of activity. Table 10 summarizes some of the results of the ORNL/RFF study.

Other Damage Studies

Other damage studies consider a wide range of topics. Table 10 also summarizes the articles discussed in this subsection. The purpose of this presentation is to make clear other direct formulations of damage that entail monetary methods to quantify the damage. These direct damage estimations calculate the monetary value of the damage (or benefits) of changes in a specific form of pollution in some target environment.

Adams (1989) analyzes the effect of climate change on agriculture. In this paper, climate change forecasts are used to reflect a doubling of atmospheric CO_2. Crop simulation models project the effects of climatic changes on the yields of rain fed and irrigated cultivators of soybeans, wheat and corn in the United States. The model in the paper simulates the effects of changes in agricultural demand, resource availability and usage, food processing, quantities produced, imports and exports. One of the climate models projects annual crop production at 80% of the base case with a 30% increase in prices. This translates into a loss of (1982) $10.5 billion.

Kopp and Krupnick (1987) anlayze the effects of ozone on agriculture. The authors employ data on baseline rural ozone levels, a dose-response function showing the effect of a change in ozone on crop yields and a model that translates the effect on crop yields to social welfare. The authors find that, in the United States, a 20% reduction in ozone would increase wheat, corn and soybean yields from 1% to 4%. Kropp and Krupnick conclude that a 10% decrease in rural ambient ozone levels would lead to a welfare gain of $700 million (in 1986) and $500 million annually by 1990.

Table 10
Summary of damage studies

Study	Pollution	Damage	Valuation Method	Region
ORNL/RFF (1994)	SO_2	cough and chest discomfort days: $3900, $8200	willingness to pay (WTP) to avoid	East Tennessee and Four Corners region of New Mexico
ORNL/RFF (1994)	SO_2	household materials per region: $53,000, $255,000	market prices	East Tennessee and Four Corners region of New Mexico
ORNL/RFF (1994)	SO_2	visibility: $2,600,000 per mile change	WTP	East Tennessee and Four Corners region of New Mexico
ORNL/RFF (1994)	particulates	restricted activity and asthma attack days	WTP	East Tennessee and Four Corners region of New Mexico
ORNL/RFF (1994)	particulates	mortality per region: $930,000, $3900	WTP	East Tennessee and Four Corners region of New Mexico
ORNL/RFF (1994)	NO_x	phlegm days	WTP	East Tennessee and Four Corners region of New Mexico
Adams (1989)	CO_2	agriculture: $10.5 billion	market prices	United States
Kopp and Krupnick (1987)	ozone	$500 million/year (damage reduction)	market prices	United States
Kopp *et al.* (1985)	ozone	varies	market prices	United States
Adams *et al.* (1982)	oxidents	$10 million	market prices	Southern California
Adams *et al.* (1986)	ozone	varies	market prices	United States

Kopp *et al.* (1985) develop an agricultural assessment model to explore the effects of different ozone levels on agriculture in the United States. Valuing changes in consumer and producer surplus, Kopp *et al.* calculate the welfare gains/losses that would occur with soybeans, corn, wheat, cotton and peanuts as a result of changes in the concentration (parts per million) of ozone. Soybean welfare losses are found to be $708 million (1978 dollars) with a 0.15 ppm, but a $529 gain with a 0.09 ppm.

Adams *et al.* (1982) analyze the economic benefits of reducing 1976 ambient oxident exposures of fourteen crops in Southern California. Using a price-endogenous mathematical model and dose-response relationships between pollutant and crops, the authors calculate the level of consumer surplus with and without the 1976 levels of oxident air pollution. They find that the elimination of 1976 air pollution levels would increase consumer surplus by over $10 million per year (1976 dollars) by increasing crop levels.

Adams *et al.* (1986) assess the benefits to agriculture that result from reductions in ambient ozone pollution. Using a spatial equilibrium model of U.S. agriculture, the authors find benefits estimates of 10%, 25%, and 40% reductions in ambient ozone levels to be $0.7, $1.7, and $2.5 billion, respectively. The changes in agriculture from different ozone concentrations come from data on corn, soybeans, wheat, cotton, grain sorghum and barley. The authors conclude that benefits would result (3% of agricultural production) from moderate ozone reduction.

Problems with the DFA for Policy Application

These studies measure the monetary valuation of environmental damage for specific pollution quantities and calculate economic impacts. But DFA studies do not analyze and apply monetary valuations of damage to all of the chemical emissions *targeted*[50] *by the risk-based tax*

[50] These TRI chemicals are discussed in chapter four.

system using a specific pathway or common measurement of damage. As a result, an index of damage values that incorporates these pollutants is appropriate for risk-based taxation. Extensive information requirements exist to levy a fully informed tax policy to regulate chemical emissions with the DFA.

INDIRECT MEASURES OF DAMAGE: RANKING AND SCORING METHODOLOGIES AND THE CCPCT INDEX

Scientists employ ranking methodologies to evaluate the relative toxicity of chemicals; however, no consensus exists concerning the best methodologies or results (Davis *et al.*, 1994b). Nevertheless, the risk-based tax system employs the ranking method of the Center for Clean Products and Clean Technologies (CCPCT) at The University of Tennessee. This system creates a ranking and scoring methodology for chemical emissions on the basis of toxicity and potential exposure. To put the CCPCT index in perspective, the following subsections first analyze different ranking methodologies. The CCPCT index is then discussed as an appropriate framework for the risk-based tax system. The key in the following presentation is to consider the methodologies in terms of their ability or inability to rank chemical emissions.

Criteria Necessary for a Feasible Proxy

For risk-based taxation, an indirect chemical ranking index takes the place of individual damage function studies. First, the risk-based proxy must evaluate chemical emissions and have the potential to address more. Second, the system must rank the pollutants using the same measure of damage. Third, the damage value for individual emissions must be formulated on a relative basis. Fourth, the relative ranking must be quantifiable for the application of tax policy. These criteria are necessary for a damage index to suit risk-based taxation.

Ranking Approaches as Indirect Measures of Damage

The indirect methodologies discussed below differ from the direct measures of damage from the previous subsections in terms of scope and purpose. The previous methods focus directly on one pollutant and calculate a specific impact on human health or the environment. The direct methods provide a framework to evaluate the relationship between emissions and damage in a particular case. In contrast, the following ranking and scoring mechanisms use different methods and endpoints to assess the damage potential of a set of pollutants. By establishing methods to evaluate damage potential, the indirect frameworks rank emissions according to guidelines and/or establish a way to calculate damage potential.

Davis *et al.* (1994b) evaluate fifty-one indirect ranking methodologies according to the following criteria: the purpose and application of the ranking systems, human health and environmental endpoints included, the inclusion of measures of exposure, data selection, the handling of missing data and the weighting of different impacts. The ranking systems evaluate chemicals in terms of toxic effects and (often) some measure of exposure. The indices have been developed for the purposes of regulatory action, priority setting and impact evaluations (Davis *et al.*, 1994b, 9-10). The following discussion contrasts five methodologies from the list of fifty-one since the CCPCT index shares their purpose. The ranking approaches are contrasted in terms of the Davis *et al.* (1994b) criteria from above. The purpose of this presentation is to address features of relevant ranking and scoring mechanisms to contrast with the CCPCT index.

First, the Toxic Release Inventory (TRI) Environmental Indicators Methodology, developed by the EPA Office of Pollution Prevention and Toxics, addresses TRI chemicals in order to "devise a measure that reflects the impacts of chemical releases and transfers, which can be used

to assess progress in reducing these impacts over time" (Davis *et al.*, 1994b, A-1). This methodology includes a human health chronic indicator and ecological chronic indicator to calculate the impact of chemical releases. The former includes mammalian toxicity,[51] human exposure potential, and population. The latter includes aquatic toxicity[52] and ecological exposure. For a particular chemical emission, the human health chronic indicator includes toxicity weight, exposure weight and adjusted population. Exposure weight takes into account the potential for the chemical emission to impact the target population or environment. Adjusted population means the individuals that would be potentially affected by the chemical release. The ecological chronic indicator includes a measure for toxicity weight and exposure weight for the chemical, facility and medium of exposure. The final measure of impact for each chemical release takes into account each of these elements within the human health and ecological indicators. The toxicity weights are the highest weights in the methodology (Davis et al., 1994b, A-1-A-2).

Second, the New Pollutant Standards Index (PSI), developed for use by local and state air pollution control agencies, "provides a simple, uniform way to report daily air pollution concentrations, to tell the public about the general health effects associated with these concentrations..." (Davis *et al.*, 1994b, A-87). This approach applies to any air pollutant that has a calculable concentration and its National Ambient Air Quality Standard (NAAQS) is known. A subindex is used to assign a label for a particular air concentration of a pollutant. The subindex = observed concentration / NAAQS. The following scale provides the conditions:

PSI	Conditions	PSI	Conditions
0 - 5	"good"	200-299	"very unhealthy"
Above 50	"moderate"	300 +	"hazardous"
Above 100	"unhealthy"	500	"significant harm level"

[51] The endpoints are carcinogenicity and non-carcinogenic.
[52] The endpoints are include an aquatic toxicity category and bioaccumulation.

Third, the Hazard Ranking System (HRS) of the EPA "evaluates the relative potential human health and environmental threat posed by hazardous waste sites for possible inclusion on the National Priority List" (Davis *et al.*, 1994b, A-17). This system addresses a single substance that regulators expect to be the most hazardous at a site. The HRS scores include three factors in three categories: waste characteristics, likelihood of release and targets. The category scores are normalized to 100 points to obtain a pathway score. The final score includes the category scores from four pathways: groundwater migration (S_{gw}), surface water migration (S_{sw}), soil exposure (S_s) and air migration (S_a). A root-mean-square method combines the four pathways to formulate the final score (S):

$$S = (S_{gw}^2 + S_{sw}^2 + S_s^2 + S_a^2/4)^{1/2}$$

The following list summarizes the criteria used to formulate each pathway.

	Likelihood of Release	Waste Characteristics	Targets
S_{gw}	*observed release or potential to release	*human toxicity/mobility *hazardous waste quantity	*nearest well *population *resources
S_{sw}	*observed release or potential to release by land or flood	*ecosystem toxicity: persistence, bioaccumulation *hazardous waste quantity	*nearest intake *population *food chain *resources
S_s	*observed contamination *accessibility *area of contamination	*human toxicity *hazardous waste quantity	*resident individual *resident population *workers *resources
S_a	*observe release or potential to release	*human toxicity/mobility *hazardous waste quantity	*nearest individual *population *resources

Fourth, the Chemical Scoring System for Hazard and Exposure Identification of EPA's Office of Toxic Substances provides a method to screen chemicals. Specifically, the "purpose of scoring is to identify, from a large number of existing chemicals, those chemicals that may warrant further investigation" (Davis *et al.*, 1994b, A-51). This system evaluates compounds with

environmental and toxicological interest to the EPA. The system, for example, has analyzed over 100 petroleum substances and 100 chemicals with Chemical Hazard Information Profiles. The scoring method entails professional judgment and objective guidelines to evaluate chemicals on the basis of eleven parameters for effect and exposure. Data availability determines the parameters for the algorithm. For a chemical, the entire set of parameter scores is presented in a scoring profile which is then shown graphically. The following criteria are used (including different endpoints for each): developmental toxicity, acute lethality, nonlethal acute toxicity, subchronic/chronic toxicity, aquatic toxicity, bioconcentration, chemical production volume, occupational exposure, consumer exposure, environmental exposure, environmental fate and others. Experts score chemicals independently with available data (Davis *et al.*, 1994b, A-51-A-52).

Fifth, the system Hazardous Air Pollutants: Proposed Regulations Governing Constructed, Reconstructed and Modified Major Sources, developed for use by the EPA, provides "Guidance for identification of the relative hazard to human health from emissions of air pollutants...This system may be used to characterize pollutant emissions as more or less hazardous" (Davis *et al.*, 1994b, A-73). The system addresses hazardous air pollutants and assigns them to one of four categories: threshold, non-threshold, high-concern and unrankable pollutants. Non-threshold and high-concern pollutants are considered more hazardous than the others. In the system, relative ranking is performed within each group. With non-threshold pollutants, the ranking is based on cancer potency and weight of evidence. For threshold and high-concern pollutants, composite score differences form the relative ranking. Specifically, for non-threshold pollutants, the rank within category depends on weight-of-evidence from the EPA and dose-response endpoints. For threshold pollutants, ranking depends on reportable quantities, dose, and severity of effect endpoints. With high-concern pollutants, the rank within group depends on composite scores for

chronic toxicity endpoints. Unrankable pollutants do not meet the criteria for non-threshold or high-concern pollutants and lack data to qualify as a threshold pollutant.

The CCPCT Index: Method of Derivation

The CCPCT methodology provides feasible rankings as compared to the above scoring systems. The index includes a wide array of chemical emissions. It provides a relative hazard ranking according to human health impacts, environment impacts and exposure potential. The relative nature of the ranking and scoring methodology allows flexibility in the application the CCPCT index for policy purposes. The use of alternative endpoints, for example, facilitates the application of a tax system to encompass the pollutants. Other indirect measures discussed above do not readily apply to a risk-based tax policy for chemical emissions.

The ranking methodology developed by CCPCT requires extensive information for each chemical emission under consideration.[53] Davis *et al.* (1994c) use a tiered approach to simplify the procedure. A tiered approach reduces the amount of chemicals under consideration as the quality and depth of the information increases. The analysis consists of the first, or screening-level, tier.[54] The screening tier relies on accessible information and includes in the damage study (rather than eliminates) potentially harmful chemicals. (In other words, the first tier avoids false negatives. The damage ranking framework does not want to discard a chemical from consideration when it may, in fact, have a high hazard value.) The data developed for the screening tier are sufficient to rank the pollutants, as the following subsection on the scoring algorithm explains. The screening tier identifies harmless chemicals and removes them from consideration (Davis *et al.*, 1994c, 3).

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[53] These information/data requirements are discussed below.

[54] A yet-to-be designed second tier of confirmation will use the results to further the research.

A Priori Conditions

Several a priori conditions are necessary for the development of the CCPCT methodology. First, the ranking method must remain flexible so it may be modified as experience presents new ways to incorporate information. Over time, the process may develop additional methods to contribute to the framework. Second, expert judgment must play an important role in the procedure. The ranking process, in other words, should not become mechanical and isolated from experience. Third, to make the methodology transparent, the environmental and human health effects are separated according to different endpoints. This method makes it easier to categorize endpoints at the beginning and aggregate the information in the end (Davis *et al.*, 1994c, 2).

Sources of Toxicity and Exposure Data

At the present time, between 60,000 and 100,000 of the approximately 8,000,000 chemicals documented by the Chemical Abstracts Services Registry are produced commercially and are potential pollutants (Davis *et al.*, 1994c, 1). Scientists have already tested many for toxicity; however, the resources are not available to evaluate all of them. Out of the many possibilities, Davis *et al.* (1994c) test 158 chemicals with high emission levels. Chemical emission figures are from the Toxic Release Inventory (TRI) and annual pesticide usage data. Davis *et al.* (1994c) analyze 140 chemical emissions from the TRI and twenty-one high-volume pesticides from the annual usage estimates of the EPA. Three of the chemicals overlap the two data sets.

The CCPCT algorithm uses toxicological endpoint data to formulate a total hazard value for each chemical. Davis *et al.* (1994c) use peer-reviewed, experimental data from the Hazardous Substances Data Bank. Data are not available, however, for all of the endpoints. In the absence of

missing information, Davis *et al.* use structure analysis for the purpose of estimation. Structure analysis entails qualitative and quantitative formulation. Classified as structure-activity relationships (SARs), qualitative structure analysis evaluates a target chemical's molecular structure and correlates it to the known molecular structure of another chemical. The SARs formulate numerical values for the potential health effects of pollutants on the basis of similar health effects of other, familiar chemicals. Called quantitative structure-activity relationships (QSARs), quantitative structure analysis computerizes the molecular structure of a chemical under study and determines the potential health effects. Both SARs and QSARs use the hypothesis that chemicals with similar molecular structures have similar biological patterns. Structure analysis creates a worthwhile estimation of fragmentary and scattered data (Davis *et al.*, 1994c, 6).

The Computational Algorithm

The CCPCT framework includes an algorithm to assess toxicity and potential exposure. Specifically, the algorithm and subsequent scoring criteria include three categories: human health effects, environmental effects, and exposure parameters. Human health effects are the different toxic responses that occur after exposure to a chemical. These endpoints include acute and chronic impacts. The acute effects result from oral ingestion and inhalation. Chronic impacts include cancer, reproductive disorders, mutagenic and development effects, and others. The algorithm data incorporate laboratory rodents as surrogates for the acute toxicity in humans while the chronic impacts use similar data and qualitative estimates (Davis *et al.*, 1994, 9).

Environmental effects include the responses of different populations of organisms to a chemical. In particular, the model incorporates quantitative assessment of terrestrial and aquatic

effects. The former involves toxic effects on mammals, birds, and higher plants. The latter means toxic effects on aquatic populations (Davis *et al.*, 1994c, 12).

Davis, *et al.* define exposure as "the concentration of a chemical in space and time at the interface with a target population" (1994c, 13). After a chemical flows into the environment, a number of physical changes occur. Chemical transformations affect the quantity and distribution of emissions into compartments of the environment. The transformations also determine the potential levels of exposure. Both persistence and bioaccumulation affect exposure assessment. The former involves the transformations of a chemical as it persists in the environment. It may degrade over time or distribute throughout a compartment of the environment and persist. Bioaccumulation occurs when an organism absorbs a chemical and increases its concentration rate to exceed the level that exists in the surrounding environment. Bioaccumulation is a function of physicochemical properties and solubility (Davis *et al.*, 1994c, 13).

These three properties, human health effects, environmental effects and exposure potential form the components of the algorithm:

Total hazard value = (Human health effects + Environmental effects) x Exposure potential

where

$$\textbf{Human health effects} = HV_{oralLD50} + HV_{inhalationLC50} + HV_{carcin} + HV_{other}$$

$$\textbf{Environmental effects} = HV_{oralLD50} + HV_{fishLC50} + HV_{fishNOEL}$$

$$\textbf{Exposure factor} = HV_{BOD} + HV_{hydrolysis} + HV_{BCF}.$$

These equations calculate chemical hazard value, taking into account exposure effects on human and animal populations (Davis *et al.*, 1994c, 19). Appendix 3 explains the endpoints in the equations for human health and environmental effects and the exposure parameters. Davis *et al.* first calculate the hazard value for target chemical emissions with the human health and environmental endpoints weighted equally. They then test the sensitivity with different weighting

assumptions. The authors assign greater weight, for example, to the human health and/or environmental effect.

In the CCPCT algorithm human health and environmental endpoints contribute additively to total hazard value. Each endpoint in these two equations assumes an equal weight. The exposure parameters, in contrast, contribute multiplicatively to total hazard value: "Persistence and bioaccumulation are considered pivotal to the potential for exposure and are included as multiplicative factors rather than additive effects" (Davis *et al.*, 1994c, 21).

To assign a value to each chemical emission, Davis *et al.* use a numbering system. The system is designed so the experimenter may assign different weights to individual endpoints in order to alter the degree of importance. The flexibility is necessary since it is not clear ex ante which endpoint should assume a greater weight. After the initial results are presented below, three alterations in the weighting system are explored.

Davis *et al.* (1994c) arbitrarily assign values from one to twenty for human health effects: ten points for acute and ten for chronic effects, where a higher number means a more potent effect. For the sake of consistency, environmental effects take on values from one to fifteen: each endpoint in the equations for human health and environmental effects may assume up to five points. This system makes each endpoint within the two relevant equations impact the total hazard value equally. (The results of changing this numbering system are explored below.) Exposure parameters are also ranked arbitrarily from one to 7.5. Each parameter in the latter equation may assume from one to 2.5 points. With this system, the total possible numerical value is 262.5.[55] The actual maximum value for a single chemical emission is 187.5. In the final analysis Davis *et al.* normalize the hazard values on a scale from one to 100.

[55] The following equation gives this number: (20 + 15) * 7.5.

The algorithm is designed so users may alter the weights of individual endpoints. If a researcher determines that the cancer effect is more severe relative to other endpoints, a scalar could be applied to (HV_{carcin}) to increase its importance. Under what conditions are these weighting assumptions beneficial? By initially establishing the algorithm to additively compare human health and environmental impacts, the equations introduce a degree of relativity in the calculation of total hazard. (A researcher may conclude that neither endpoint should assume a greater weight than another.) In the absence of any reason to provide a greater weight to a singular endpoint, the additive weighting scheme adopted in the algorithm does not single out one measure of damage in the human health and environmental equations as more important than another. One exception exists. The "other specific effects" endpoint includes mutagenicity, developmental effects, reproductive effects, neurotoxicity and other chronic effects. The authors use qualitative data to evaluate this endpoint. Assigning hazard values on the same zero to five numbering scale as the quantitative endpoints ($HV_{oralLD50}$, $HV_{inhalationLC50}$, HV_{carcin}, $HV_{oralLD50}$, $HV_{fishLC50}$, and $HV_{fishNOEL}$) provides a greater weight to each quantitative endpoint rather than the group of qualitative effects grouped as the endpoint "other specific effects." Even though it provides equal weight between endpoints, the weighting scheme for the hazard equations are arbitrarily set.

The inclusion of the exposure parameters as multiplicative factors increases their importance in the formulation of the total hazard value. Davis *et al.* consider bioaccumulation and persistence as "pivotal" in the analysis. The provision of greater weight to these parameters creates a time element in the calculation of total hazard. A particular chemical emission, for instance, may continue causing a degree of risk to nearby populations if it persists in a consistent or degenerating form over a long period of time. The multiplicative influence of the exposure parameters provides a feasible method to weigh the importance of bioaccumulation and persistence (Davis *et al.*, 1994c, 21).

The use of the numbering system to assign values from zero to five for the endpoints in the human health and environmental equations provides a relative formulation of hazard for these impacts. Although arbitrarily set, this framework establishes a numbering system to distinguish between the toxicity levels of the different endpoints. The use of a scale between zero and twenty-five for each endpoint, for example, would not necessarily improve the results. Before normalization, a numbering system with a greater variance between the highest and lowest values would create a bigger variation in total hazard.

The numbering system for exposure parameters is also set arbitrarily. These parameters contribute a greater weight to total hazard value as a result of their multiplicative place in the total HV equation. As opposed to a scale that adopts the same numbering system for the human health and environmental endpoints, the 1-to-2.5 scale lowers the impact of potential exposure for the most toxic chemicals. With additional information or testing to show that a particular endpoint should assume a different influence, researchers could alter the algorithm's weighting mechanism.

The strength of the weighting mechanism of the algorithm, therefore, is its flexibility. In the absence of information that necessitates a different weighting or numbering system, the algorithm allows the formulation of relative hazard values. It also leaves the possibility open for the inclusion of a scalar to provide a different degree of influence for particular endpoints.

Under what conditions are the algorithm's weighting assumptions not beneficial? A priori, there is no reason to believe that the human health and environmental endpoints *should* contribute additively to the formulation of total hazard value, or that the exposure parameters *should* contribute multiplicatively. In the absence of reason for different equations, the authors propose equal impact, initiate flexibility and assign some degree of influence. In other words, the formulation of hazard by the CCPCT algorithm captures the effects of individual endpoints in a straightforward manner, but arbitrarily establishes the impact of the endpoints. If there is reason to

believe that a particular endpoint should assume a greater weight, or that exposure parameters should not assume multiplicative influence, the basic CCPCT algorithm could be altered.

Another area of concern with the algorithm is the controversy surrounding the method to calculate the impact of chemical substances on human health. The human health equation includes ($HV_{oralLD50}$), ($HV_{inhalationLC50}$) and (HV_{carcin}) endpoints. As Appendix 3 explains, these endpoints are comprised of data that come from the application of chemical substances to rodents. But the principal method in determining potential carcinogenicity of a chemical substance is based on research that applies "huge" daily doses to these creatures (Abelson, 1990). Applications such as these often adopt the procedure of calculating the risk of a chemical substance to the "maximally exposed individual." The MEI, however, is an inadequate target (Goldstein, 1989). The risk to the MEI is less important than the aggregate risk to nearby populations that are affected from the release of a substance. The use of data from high doses to rodents does not provide a realistic depiction of the impact of chemical pollution on target populations (Goldstein, 1989).

Consider, for example, that animal cancer tests are conducted at maximum tolerated doses (MTD). This exposure may cause mitogenesis, or induced cell division: "Chronic dosing at the MTD can be thought of as a chronic wounding, which is known to be both a promoter of carcinogenesis in animals and a risk factor for cancer in humans" (Ames and Gold, 1990, 970). As a result, a large percentage of all chemicals tested are found to be carcinogens at chronic doses. A problem occurs when extrapolating the results from rodents to humans. Ames and Gold (1990) note that half of all chemicals tested chronically at the MTD are found to be carcinogens.

Given the limitations with the initial weighting mechanism and the data, does the algorithm have redeeming qualities? Before normalizing on a scale from one to 100, a researcher may use alternative numbering systems to demonstrate the different impacts. Since it uses the same calculation procedure, the algorithm's strength is the formulation of relative hazard values.

Algorithm Results

Table 11[56] ranks the thirty most toxic chemicals as determined by the algorithm, including sensitivity analysis for missing data, and equal weighting for human health and environmental effects. Recently these results were published in *Environmental Toxicology and Chemistry* (Swanson *et al.*, 1997). The algorithm is designed to use a complete set of endpoint data. For the chemicals with missing data, QSARs or SARs derive a number. No reliable QSAR or SAR exists, however, that estimates particular endpoints for some chemicals. In the absence of information, Davis *et al.* (1994c) assign a default value of either zero or five (the minimum and maximum values) for acute inhalation, acute fish and chronic fish toxicological endpoints. The numbers in Table 11 incorporate the default value of zero.[57]

Table 11
Top 30 ranked chemicals from algorithm
(default hazard value to zero for missing data)

Rank	Not weighted by releases	Rank	Not weighted by releases
1	cadmium compounds (100)	16	diaminotoluene (57)
2	arsenic compounds (82)	17	hydrogen fluoride (55)
3	terbufos (81)	18	di(2-ethylhexyl)phthalate (55)
4	hexachloro-1, 3-butadiene (78)	19	chlorothalonil (53)
5	PCB (71)	20	2,4-D (53)
6	trifluralin (63)	21	1,3-dichloropropene (52)
7	hexachlorobenzene (62)	22	2,4-dinitrophenol (52)
8	1,2,4-trichlorobenzene (62)	23	epichlorohydrin (52)
9	chromium compounds (61)	24	decabromodiphenyl oxide (51)
10	2-nitropropane (60)	25	byphenyl (51)
11	formaldehyde (60)	26	copper compounds (51)
12	cobalt compounds (59)	27	hydrogen cyanide (51)
13	lead compounds (59)	28	styrene (50)
14	nickel compounds (59)	29	dibutylphthalate (50)
15	anthracene (58)	30	2,4-dinitrotoluene (49)

Source: Davis *et al.* (1994c)

[56] This is Table 8, on page 31, in Davis *et al.*, 1994c.

[57] The results with the default value of five are slightly different than the ranking of Table 11. Specifically, six new chemical emissions are in the top thirty when the default value is five.

Different weighting assumptions change the relative rankings of the chemicals. Davis *et al.* (1994c) also consider three different weighting schemes. The alterations apply lesser or greater importance to specific components of the algorithm. First, Davis *et al.* alter the basic algorithm by doubling the weight of the human carcinogenic endpoint. (To give the human carcinogenic endpoint a greater degree of importance, the endpoint could be tripled, increased by fifty percent or another amount.) The authors choose this alteration to give emphasis to the cancer effects on humans of chemicals. In particular instances some researchers or communities may believe that this endpoint has a greater importance in the formulation of hazard, relative to other endpoints.

Table 12 presents the results from the first alteration. The chemicals that are not ranked in the top thirty are decabromodiphenyl oxide, biphenyl, copper compounds, hydrogen cyanide, and dibutyl phthalate. The new chemical emissions in the top thirty are 1,3-butadiene, vinyl chloride, tetrachloroethylene, n-nitrosodiphenylamine, and trichloroethylene. Of the three variations of the basic algorithm in Davis *et al.* (1994c), the doubling of the carcinogenic endpoint in this alteration leads to the greatest effect in terms of chemicals added and deleted from the original top thirty.

Second, the algorithm was altered with respect to the acute human health endpoints, which include oral and inhalation. The weights were halved relative to the other endpoints. Davis *et al.* (1994c) initiated this change to provide greater importance to the human carcinogenic endpoint and the environmental effects. Table 13 presents the top thirty chemical emissions from the algorithm with this new weighting scheme. This alteration in the algorithm led to a smaller change in the original rankings as compared to the results in Table 12. Two chemicals in Table 13, hydrogen fluoride and hydrogen cyanide are no longer in the original top thirty. The two new chemicals are 1,3-butadiene and tetrachloroethylene.

Table 12
Top 30 ranked chemicals from algorithm: double weighting for human carcinogenic endpoint

Rank	Not weighted by releases	Rank	Not weighted by releases
1	cadmium compounds (100)	16	1,2,4-trichlorobenzene (53)
2	arsenic compounds (87)	17	anthracene (53)
3	polychlorinated biphenyls (74)	18	2,4-dinitrotoluene (53)
4	hexachloro-1,3-butadiene (71)	19	1,3-butadiene (53)
5	terbufos (70)	20	epichlorohydrin (52)
6	chromium compounds (68)	21	1,3-dichloropropene (52)
7	nickel compounds	22	styrene (52)
8	hexachlorobenzene (65)	23	cobalt compounds (51)
9	formaldehyde (63)	24	vinyl chloride (50)
10	2-nitropropane (62)	25	tetrachloroethylene (50)
11	lead compounds (62)	26	hydrogen fluoride (48)
12	diaminotoluene (59)	27	n-nitrosodiphenylamine (47)
13	di(2-ethylhexyl)phthalate (57)	28	trichloroethylene (46)
14	2,4-D (56)	29	chlorothalonil (46)
15	trifluralin (54)	30	acrylamide (45)

Source: Davis *et al.* (1994c)

Table 13
Top 30 ranked chemicals from algorithm: half weighting for acute human health endpoints

Rank	Not weighted by releases	Rank	Not weighted by releases
1	cadmium compounds (100)	16	anthracene (58)
2	arsenic compounds (83)	17	decabromodiphenyl oxide (57)
3	polychlorinated biphenyls (78)	18	1,3-butadiene (55)
4	hexachloro-1,3-butadiene (74)	19	2,4-D (55)
5	terbufos (73)	20	copper compounds (54)
6	lead compounds (65)	21	diaminotoluene (54)
7	hexachlorobenzene (64)	22	styrene (53)
8	nickel compounds (63)	23	1,3-dichloropropene (53)
9	chromium compounds (63)	24	2,4-dinitrotoluene (53)
10	di(2-ethylhexyl)phthalate (62)	25	2,4-dinitrophenol (52)
11	1,2,4-trichlorobenzene (61)	26	dibutyl phthalate (51)
12	2-nitropropane (61)	27	chlorothalonil (51)
13	cobalt compounds (61)	28	tetrachloroethylene (51)
14	trifluralin (60)	29	epichlorohydrin (50)
15	formaldehyde (59)	30	biphenyl (49)

Source: Davis *et al.* (1994c)

Finally, Davis *et al.* (1994c) changed the weighting for the environmental effects endpoints. The weights assigned to acute oral LD_{50}, acute fish LC_{50} and fish NOEL endpoints were halved. By altering the algorithm in this manner, Davis *et al.* assigned greater importance to the endpoints for human effects. This alteration makes the environmental effects less important in the formulation of hazard value as compared to human effects. In Table 14, three chemicals were removed from the original top thirty: dibutyl phthalate, copper compounds and decabromodiphenyl oxide. The three new chemicals are vinyl chloride, 1,3-butadiene, and acrylamide.

What the Results Measure

By assessing toxicity and potential exposure, the different weighting methods in the CCPCT framework provide a relative ranking of chemical emissions in terms of potential risk. The numbers in parentheses next to each chemical in Table 11 represent total hazard value normalized on a scale between zero and 100. Since the chemicals are not weighted by releases, the numbers represent relative risk values. Davis uses the rankings to identify priority chemicals for the evaluation of safe substitutes.

Correspondence with Direct Measures

The (direct) damage function method discussed earlier in chapter three uses five stages. It identifies residual discharges, explains the transfer function for discharges and ambient conditions, elucidates the dose-response functions for pollution concentrations and damage effects, places a monetary valuation on the identified damages, and determines if the damages are internalized. To form a damage function, this direct DFA identifies an impact pathway between a specific pollutant and environmental endpoint.

Table 14
Top 30 ranked chemicals from algorithm: half weighting for environmental effects endpoints

Rank	Not weighted by releases	Rank	Not weighted by releases
1	cadmium compounds (100)	16	epichlorohydrin (57)
2	arsenic compounds (85)	17	1,2,4-trichlorobenzene (56)
3	hexachloro-1,3-butadiene (77)	18	2,4-D (56)
4	terbufos (75)	19	vinyl chloride (54)
5	hexachlorobenzene (68)	20	di(2-ethylhexyl)phthalate (52)
6	lead compounds (65)	21	anthracene (52)
7	polychlorinated biphenyls (68)	22	2,4-dinitrotoluene (51)
8	diaminotoluene (63)	23	chlorothalonil (50)
9	2-nitropropane (62)	24	styrene (50)
10	nickel compounds (62)	25	1,3-butadiene (50)
11	chromium compounds (61)	26	acrylamide (49)
12	hydrogen fluoride (61)	27	1,3-dichloropropene (49)
13	trifluralin (61)	28	cobalt compounds (49)
14	lead compounds (58)	29	2,4-dinitrophenol (40)
15	hydrogen cyanide (58)	30	biphenyl (49)

Source: Davis *et al.* (1994c)

In contrast, the chemical rankings from the CCPCT index provide hazard values and relative damage effects of chemical emissions. The CCPCT index does not give a technology characterization of target industries, relate discharges to ambient concentrations, provide a monetary valuation or determine the extent that the externality of a specific pollutant is internalized. The CCPCT framework therefore represents a variation of the third step of the DFA. Despite the relative nature of the hazard values, they do not provide a quantitative estimate of changes that occur in the environment from emissions. The endpoint values for individual pollutants are derived from laboratory data. For a DFA, estimates from exposure-response functions could include more sick days, reduced fish yields and increased mortality and morbidity. In sum, the CCPCT index provides relative damage values, but does not explain how a given level of emissions of individual pollutants will impact humans or the environment. Nevertheless, the

CCPCT index provides a scientific basis for a risk-based environmental tax system by formulating a relative index of damage values.

IMPLEMENTATION OF A RISK-BASED ENVIRONMENTAL TAX SYSTEM

To implement a feasible risk-based tax, a policy maker could use the relative hazard values as a weighting mechanism. A levy system could tax individual chemical emissions according to relative hazard values from the CCPCT index. The following risk-based policy system incorporates the chemical rankings from Table 11.

The Risk Unit

In order to tax individual pollutants, a policy maker would need a common unit of measurement to compare the different emissions on the same scale. In the analysis that follows, this measurement will be called a risk unit (RU). Because of the relative nature of Table 11, each hazard value will represent a certain number of RUs. As a result, the charge on each chemical can be formulated when a tax rate is assigned to one RU.[58]

Consider the following hypothetical scenario. Suppose that chemicals A, B, and C are found to be toxic. Scientists perform tests to calculate relative toxicity levels for these chemical emissions. Table 15 presents the results. Relative to pollutant C, A has a toxicity value of four while B has a value of two. To formulate the relative toxicity values, scientists compare the different chemicals using a common unit of measurement of one emission pound. Suppose a policy

[58] Different weighting schemes for the algorithm would provide policy makers with the possibility of giving greater importance to a specific endpoint or a few endpoints. If, for instance, a regulator wanted to assign greater importance to the carcinogenic endpoint, the relative rankings in Table 13 would be appropriate.

maker defines one RU to equal one pound of chemical C in Table 15. With this numeraire pollutant established, the policy maker then calculates RUs for chemicals A and B.

Table 15
Hypothetical toxicity rankings

Chemical	Toxicity/pound	RU	Pounds	Tax Rate	Revenue
A	4/pound	4	20	$5/RU	$400
B	2/pound	2	20	$5/RU	$200
C	1/pound	1	20	$5/RU	$100
Total					$700

To calculate the revenue from a tax on one chemical, the following formula multiplies the number of risk units by the tax rate. In this case the base, i.e. number of emission pounds, must be multiplied by the RU and then multiplied by the tax rate per RU:

(Base) x [(RU/lb.) x (Tax Rate/RU)] = Revenue.

Suppose a policy maker implements a tax rate of $5 per RU. Since a pound of chemical A equals four RUs, the tax rate would equal $20 per pound. If a company emits twenty pounds of A, it would have to pay $400. Adding the tax revenue for each chemical emission in Table 15 gives the total revenue for the environmental tax system.

How would this risk-based tax system apply to the chemical emissions in Table 11? Suppose that one RU is defined to equal one pound of chlorobenzene[59] emissions (hazard value of thirty-one in the CCPCT index):

1 pound of chlorobenzene = 1 RU.

[59] In the original estimation of the policy system, 123 chemical emissions were used to calculate annual revenue. Chlorobenzene is number sixty-two in terms of the list of hazard values; therefore, sixty-one chemical emissions are below it and sixty-one are above it.

If a firm emitted one pound of trichlorobenzene, this quantity would equal two RUs, since the hazard value is sixty-two in Table 11:

1 pound of trichlorobenzene = 2 RU

No matter which numeraire pollutant defines a RU, all chemicals must represent relative quantities. This continuity allows a policy maker to levy one tax rate, which minimizes the administrative complexity of calculating a different rate for each pollutant. Table 16 lists the RUs for each chemical in Table 11: one RU equals one pound of chlorobenzene emissions.[60]

Table 16
Risk units

Rank	Chemical	RU	Rank	Chemical	RU
1	cadmium compounds	3.22	16	diaminotoluene	1.83
2	arsenic compounds	2.64	17	hydrogen fluoride	1.77
3	terbufos	2.61	18	di(2-ethylhexyl)phthalate	1.77
4	hexachloro-1, 3-butadiene	2.51	19	chlorothalonil	1.70
5	PCB	2.29	20	2,4-D	1.70
6	trifluralin	2.03	21	1,3-dichloropropene	1.67
7	hexachlorobenzene	2.00	22	2,4-dinitrophenol	1.67
8	1,2,4-trichlorobenzene	2.00	23	epichlorohydrin	1.67
9	chromium compounds	1.96	24	decabromodiphenyloxide	1.64
10	2-nitropropane	1.93	25	byphenyl	1.64
11	formaldehyde	1.93	26	copper compounds	1.64
12	cobalt compounds	1.90	27	hydrogen cyanide	1.64
13	lead compounds	1.90	28	styrene	1.61
14	nickel compounds	1.90	29	dibutylphthalate	1.61
15	anthracene	1.87	30	2,4-dinitrotoluene	1.58

Risk Units and Tax Payment

Figure 4 graphically demonstrates the relationship between RUs and marginal damage. Figure 4 assumes that one emission pound of chlorobenzene (c) equals one RU, while one emission

[60] To formulate risk units, Table 16 assumes the measurement quantity of each chemical is equal in emission pounds.

pound of trichlorobenzene (t) equals two RUs. The marginal damage functions are linear in Figure 4, corresponding to the results from the CCPCT index.[61] The risk-based tax system converts the CCPCT hazard values (not weighted by releases) into RUs. As a result, the RU represents damage for one pound of each chemical emission.

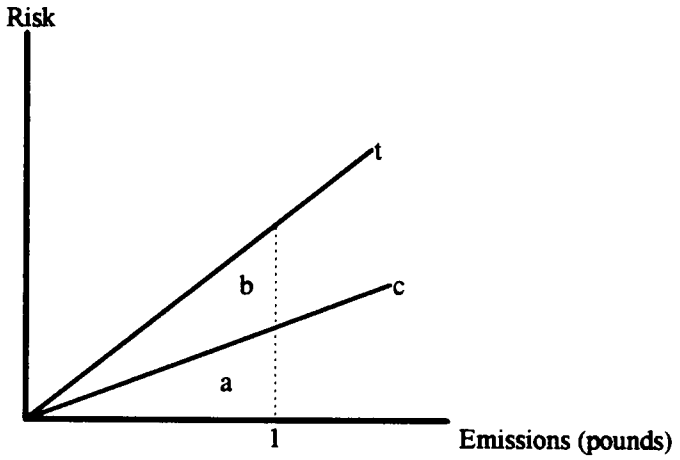

Figure 4
Risk units and marginal damage

At one emission pound, the area under the chlorobenzene function (a) represents one RU. In contrast, the area under the trichlorobenzene function (a + b) represents two RUs at one emission pound. The larger area for trichlorobenzene equals a damage value twice that of chlorobenzene. Suppose the tax rate is set to equal $1 per RU. At one emission pound for each chemical, the polluter would pay $2 for trichlorobenzene and $1 for chlorobenzene. This difference reveals that the tax payment goes up when the CCPCT hazard value increases. The risk-based system attempts to account for the pollution externality by pricing chemical emissions using an approximation of the damage function. When tax payment for specific chemical

[61] This linearity assumption limits the understanding of the relationship between emissions and damage for individual pollutants.

emissions equals the economic value of pollution damage, the risk-based policy internalizes the externality. This internalization occurs when the risk-based tax approximates fully informed regulation.

WHEN THE RISK TAX APPROXIMATES A FULLY INFORMED LEVY

The CCPCT index provides the third step of the DFA by formulating relative damage values. In the absence of information to apply an economic valuation to the hazard values in the CCPCT algorithm, risk-based taxation provides an approximation of fully informed regulation. Specifically, risk-based taxation assigns a tax rate to one RU which translates to a different price on individual chemical pollutants. Higher rates apply to the emissions that pose a greater degree of risk to humans and the environment.

But risk-based tax policy necessitates the formulation of a tax rate. Setting the tax rate becomes an imprecise procedure without incorporating the economic value of one RU—found through some estimation technique. In the absence of a logical method to set the rate, risk-based taxation establishes relative prices for chemical emissions, but the prices may not (and probably do not) equal some "true" value of damage. As a result, the risk-based system applies tax rates that over- or understate the economic value of marginal damage. The risk tax therefore approximates a fully informed regulation when the marginal damage functions implied by the former regulation convey an accurate measure of *pollution* damage and the tax rate equals a "true" value of marginal *economic* damage. To achieve a fully informed policy, the governing authority of the risk-based system would have to recognize the problem of tax-rate formulation and levy the rate with the best available information.[62]

[62] The "Setting the Tax Rate" section in chapter four explores this issue and different ways to calculate the tax rate.

If research could provide the economic value of damage from individual endpoints in the CCPCT index, the problem of setting the tax rate would be resolved. Using some weighted average for the contribution of each endpoint, a regulator could levy taxes according to the economic value of pollution damage from each chemical. Not only would the policy system assign prices to the pollutants in a relative manner for target chemical emissions, but the rates would equal the economic value of damage—the information from the fourth step of the DFA. This method of using economic damage values to levy taxes represents a necessary condition for the implementation of a fully informed policy. The risk-based levy therefore approximates a fully informed environmental tax for an individual pollutant when (1) the damage function implied by the former system represents a "good" estimation of "true" marginal damages, and (2) the risk-based price equals the economic value of marginal damage at the current level of emissions.

Information Requirements

What information requirements are necessary to implement a fully informed environmental tax for individual chemical emissions? First, after targeting chemical emissions for regulation, appropriate dispersion modeling would have to compute the changes in concentrations of pollutants or other physical conditions that result from emissions. This analysis would provide estimates of different pollution concentrations in areas of jurisdiction. Necessary for policy application, the dispersion modeling aids the estimation of the impact of target pollutants. Since a number of chemical emissions are involved, the estimation would have to take into account the effects of pollution synergy. The interaction of individual effluent complicates the observation of the change in concentration of a single chemical emission.

Second, researchers must calculate the level of damage at different emission quantities for target pollutants. For the sake of consistency, damage is defined here as the total hazard value

from the CCPCT index. This value includes human and environmental impacts. Given that the endpoints for the human health and environmental effects entail an analysis of the impacts of chemicals on laboratory animals in the CCPCT data bank, persistence and bioaccumulation affect human health. In other words, chemical emissions into the air, water and earth impact morbidity and mortality. With appropriate dose-response and exposure-response modeling, researchers could construct a damage function for each pollutant over a range of emissions.

Damage functions for each pollutant would assume different shapes. The damage functions for highly toxic chemicals, for example, may increase at a decreasing rate beyond some point as the pollutants begin to saturate the environment. Up to the inflection point, however, damage may increase at an increasing rate as the pollutants begin to affect human health and the environment for nearby populations. The formulation of a marginal damage function that shows how damage relates to emissions would allow an administer to identify total damages at annual emission quantities and marginal damage at some standard.

The CCPCT framework calculates relative hazard values for each chemical emission, either weighted or not weighted by releases. (Tables 11 through 14 summarize the thirty most toxic pollutants when the chemical emissions are not weighted by releases.) "Not weighted by releases" means that the hazard values for individual chemicals do not take into account annual emission levels. In other words, the hazard value for each pollutant does not use the fact that some pollutants are more prevalent than others. When the chemical emissions are "weighted by releases," high emissions quantities influence the hazard values. A chemical may receive a higher hazard value if it is released in larger quantities in the environment when it is weighted by releases. The idea is that a larger quantity has a higher propensity to affect humans and the environment.

The problem, however, with the use of the "weighting by releases" method in the application of policy is that some regions may not emit large quantities of the chemicals that have

high hazard values on the basis of high national emission quantities. In other words, if a tax system applies to different regions or states, the "weighted-by-releases" results could assign higher-than-necessary damage values to chemical pollutants that may not represent a significant percentage of emissions in a particular area. This method would result in hazard values not represented accurately in magnitude. The pollutants would be weighted heavily by releases but not enough by persistence or toxicity. The hazard values not weighted by releases, therefore, represent better figures to use for policy formulation if regulators want a flexible tax policy for different levels of government.

Given its methods to formulate hazard values, is the CCPCT framework conducive for the estimation of marginal damage functions? The purpose of the CCPCT algorithm is to calculate the hazard values of a *number* of chemicals in a *relative* manner. The framework is not intended to formulate the relationship between the changes in damage and emissions for one pollutant. The CCPCT method of determining hazard values for individual pollutants encompasses a number of damage effects from different pathways in which chemical pollutants affect humans and the environment. Of course, no method to estimate damage could encompass all impact pathways. But the CCPCT algorithm includes representative endpoints from three important aspects: effects on human health (represented through surrogates), the environment and exposure potential. The CCPCT framework of calculating a total hazard value offers a starting point to formulate marginal damage functions in a relative manner for chemical emissions.

Even though it is not intended for this purpose, could the CCPCT index provide a framework for the formulation of damage functions? Researchers could use known impacts to quantify the damage effects of individual endpoints at different emission levels or extrapolate in the absence of information. The first endpoint in the human health effects equation in the CCPCT algorithm ($HV_{oralLD50}$), for example, measures the lethal chemical concentration that will kill half a

rodent group in fourteen days when applied orally in a single dose. To formulate relative hazard values for this endpoint, Davis *et al.* gathered data from the Hazardous Substances Databank which conveyed the lethal concentration in kilograms for each target chemical. The authors then normalized the results on a scale from zero to five as they did with the other human health and environmental endpoints according to the pollutant's potential for harm. The second human health effects endpoint ($HV_{inhalationLC50}$) measures the lethal concentration of a chemical that will kill half a rodent group when inhaled continuously for a period of time. The data that Davis *et al.* (1994c) use in the algorithm for this endpoint are measured in parts per million for each pollutant. The third and fourth endpoints in the human health equation, (HV_{carcin}) and (HV_{other_i}) are based on weight-of-evidence classification and evidence of harmful effects, respectively.

The first endpoint in the environmental effects equation, ($HV_{oralLD50_i}$) is also measured in kilograms as a concentration in a substance that, when administered orally as a single dose, will kill half a rodent group in fourteen days. The second and third endpoints in the environmental effects equation, ($HV_{fishLC50}$) and ($HV_{fishNOEL}$), are measured in milligrams. The former endpoint represents the concentration of a substance in water that will kill fifty percent of a group of fish in ninety-six hours. The latter endpoint represents the highest dosage of individual chemicals that does not produce toxic effects. The reader may want to return to Appendix 3 for a concise description of the endpoints.

The point is that researchers could alter concentrations for the first two endpoints in the human health equation and the environmental effect pathways to reflect different emission quantities. By varying chemical concentrations to pattern different emission levels, researchers could obtain an array of specific damage effects that occur for each endpoint as emissions increase. Multiple estimations would eventually form damage functions for individual chemicals.

Over a range of emissions, the difference in damage effect for individual endpoints of *incremental* concentration changes may be inconsequential. But the effects may be substantial for changes that reflect large differences in emissions. The particular relationship between damage and emissions would depend on the endpoints, weighting mechanism between endpoints, target chemical and estimation technique. In sum, researchers could show how changes in the emission of different chemicals—reflected by chemical concentration—affect the first two endpoints in the human health equation and the environmental endpoints from the CCPCT algorithm.

For the sake of consistency, researchers would apply the same changes in concentrations of individual chemicals to the target endpoints to observe damage effects. With this procedure, the individual damage effects could be quantified, for example, in terms of the number of days it takes to affect an animal group in a certain manner, as with the ($HV_{oralLD50}$) endpoint. The resulting damage figures could be normalized on the scale used in the CCPCT algorithm for each endpoint or another manner. (The formulation of the exposure parameters from the CCPCT algorithm and the last two endpoints in the human health effects equation would stay the same since the former parameters are based on persistence and bioaccumulation and the latter two are based on chemical classification and scientific evidence.) The procedure for relating concentrations (emissions) and damage discussed here would have to be the same for each pollutant. As a result, the estimated marginal damage functions would entail a consistent measure of damage: the endpoints in the CCPCT algorithm.

The identification of marginal damage functions for target chemical pollutants would allow the application of an economic value to marginal damage. Accordingly, the third information requirement in the formulation of a fully informed tax policy entails the economic valuation of damage from individual chemical emissions. With relative damage information for a particular pollutant, an economic valuation would inform the regulator how to formulate an environmental

tax. In a perfect world, tax payment would equal total damage multiplied by its economic value at the optimal level of emissions. For theoretical policy implementation, the tax rate would be set such that marginal damage equals marginal control cost. For practical regulation, the tax rate would have to reflect marginal damage of additional emissions.

This proposal to apply economic valuation information necessitates the implementation of one of two potential methods. The first entails the valuation of individual endpoints in the CCPCT algorithm and the application of a weighted-average of each as they contribute to total HV. The second possibility entails the application of a single valuation estimate to represent different portions of the HV equation, as information permits. In application, neither choice is without its limitations or uncertainty.

Consider first the weighted-average valuation technique. The authors of the CCPCT index created the algorithm with flexibility in mind (Davis *et al.*, 1994c, 22). In the algorithm's basic form, each endpoint for human health and environmental effects contributes additively to the formulation of total hazard. Each endpoint may assume up to five points. In contrast, the exposure parameters contribute multiplicatively to the formulation of total hazard. Each of the three parameters may assume up to 2.5 points. With access to the data set used to formulate hazard values with the CCPCT algorithm, a researcher may alter the weights of individual endpoints or parameters and observe changes in the total hazard values of individual pollutants.

If one endpoint constitutes the total hazard value, economic valuation would be straightforward. Suppose the ($HV_{fishLC50}$) endpoint in the environmental effects equation represents the pathway from pollutant to damage to formulate total hazard value, but no other endpoints are included. In this case, information requirements for economic valuation would entail (a) the relationship between an increase in emissions of individual pollutants and an increase in the damage effects of sample fish groups, (b) an extrapolation between the results in (a) and some a

real-world scenario, and (c) peoples' willingness to pay to improve environmental quality, as represented by a healthier fish population. This latter economic valuation information is available in the travel cost literature.

What research exists in this area? A number of researchers, including Bockstael, McConnell and Strand (1988), Bockstael, Hannemann and Strand (1986), Bell, Sorenson and Leeworthy (1982), and others have applied travel cost models to calculate the consumer surplus associated with the number of trips to a recreational fishing site. The idea behind the research technique is that travel cost may be used as a proxy for the price of access to a site. If the travel cost curve is measured using surveys with quality as an explanatory variable, a proposed increase in the demand (willingness to pay) for a site that results from an improvement in environmental quality would lead to a change in consumer surplus. This change reflects the dollar value of the quality improvement. The improvement could mean a higher fish population and therefore a greater potential to catch fish as a result of a reduction in pollution. If one endpoint measures total hazard value, a researcher could quantify the damage of individual chemical emissions by incorporating the economic value of pollution reduction for the endpoint.

Suppose now the ($HV_{oralLD50}$) endpoint in the human health effects equation constitutes the only measure of total hazard value. The measurement of this endpoint incorporates the use of rodents as surrogates for humans. The problem with this particular endpoint is deciding how to extrapolate to a real-world scenario. To undertake valuation research, it is necessary to calculate how much people are willing to accept to let a situation occur, or willingness to pay to avoid it. No plausible scenario exists for an increase or decrease in rodent populations. But with information on the relationship between emissions and damage over a range of emissions, researchers could extrapolate the damage effects on rodents to humans to use some economic valuation technique to

quantify damages. Controversy exists, however, with this extrapolation technique as a previous subsection explains.

Assume, nevertheless, that the extrapolation takes place and the damage effects of individual pollutants are graphed in relationship to changes in emission quantities. The next step is to assign dollar values to physical impacts. If accurate extrapolation occurs, economic valuation techniques could measure potential changes in morbidity or mortality for target populations. For many criteria air pollutants, measurements for economic valuation for these physical changes are available in the literature. But the problem of extrapolating from the damage effects on rodents to humans proves to be a difficult obstacle.

Another problem exists. The use of all of the endpoints in the CCPCT algorithm presents a formulation of damage that is more encompassing than choosing one pathway from pollutant to a single measure of damage. The point is that it would not be accurate to base tax policy on an index of damage values that uses a narrow definition of damage. If a policy maker adopts a measure of damage that entails the CCPCT total hazard value, and each endpoint within, then a fully informed policy would have to include an economic valuation of each endpoint weighted in a certain manner.

Suppose the first two endpoints in the environmental effects equation are used to define damage at one emission pound of a target chemical. In addition, assume that economic valuation is possible for the two endpoints. If each endpoint contributes fifty percent to total damage, how would a policy maker assign the economic valuation of each endpoint to the measure for total damage? An economic value for one emission pound from the impact pathway of the first endpoint would provide a dollar figure. An economic valuation from the second endpoint would provide another dollar value. One method to incorporate the economic value of total damage in this case—as defined by the two target endpoints—would be to take fifty percent of the value of economic

damage at one emission pound for each endpoint. This way, the economic value of one emission pound of each endpoint would contribute to the total economic value of damage according to the weighting technique. Other weighting methods could also be established.

The economic valuation of damage becomes more complicated with the inclusion of all of the endpoints from the CCPCT algorithm or in any context when a policy maker wants a formulation of damage that includes more than one or two endpoints. Suppose researchers calculate the damage value of each endpoint from the CCPCT algorithm that may vary with emissions (environmental effects endpoints and the first two endpoints in the human health equation) at one emission pound, then two emission pounds, etc. The resulting damage effects would be measured in a relative manner for each chemical pollutant. If researchers could identify the economic value of pollution damage from each endpoint at each emission quantity, they may calculate the total damage value. If each endpoint contributes additively to the total damage, the economic value of individual endpoints could be formulated in the same manner established in the previous paragraph. The economic damage value from each endpoint would contribute to the total value according to its weighted contribution in the measure for HV in the CCPCT index.

Consider second the possibility of applying a more straightforward valuation technique. This method does not present restrictive information requirements, but also does not attempt to encompass each endpoint. Instead of acquiring a separate economic valuation estimate for emission reduction for each endpoint, a policy maker could adopt a general valuation. Suppose target chemical pollutants have the potential for increasing cough or chest discomfort days for a nearby population. A regulator could apply appropriate economic valuations as in ORNL/RFF (1994). Another economic valuation that encompasses a relevant impact on environmental quality could also apply, such as the effect of chemical pollution on recreational activity. Instead of being endpoint specific, economic valuation could weigh two (or more) impacts to form a workable

figure. While the formulation of marginal damage would entail a number of impact pathways, economic valuation would include specific endpoints. The strength of this variation would be the ability to apply valuation estimates from the literature and continue in the absence of information to quantify specific endpoints. In this case, efficient regulation could be established by creating region-specific policy. The key for the application of this second valuation technique involves recognizing the characteristics of the target area. In a particular region, the estimates from economic valuations of certain aspects of pollution activity would be more relevant.

In sum, the formulation of a fully informed environmental tax to regulate chemical emissions is an imprecise task. First, the policy maker would want an encompassing measure of damage that includes many endpoints. Second, the endpoints have to be aggregated (and weighted) in some manner. Third, researchers must calculate marginal damage. Fourth, damage has to be valued in dollars. Fifth, the policy maker must set a tax rate for each emission that reflects marginal damage over the course of the reporting year. As a result of the information limitations to carry out each of these steps, the formulation of a fully informed environmental tax presents a formidable task for researchers and policy makers using the endpoints in the CCPCT framework.

Economic Valuation and Fully Informed Policy

But the policy maker could adopt an average endpoint technique in the formulation of damage functions for individual pollutants according to the perceived importance of each endpoint to total damage. The regulator could then apply economic values for as many endpoints as possible to translate marginal damage into dollar terms. As with the formulation of hazard values for the CCPCT index, the weighting technique could vary according to circumstance, i.e. the perceived importance of a particular pollution impact, and the availability of information.

Suppose researchers find that marginal damage from chemical A is twice that of B under normal circumstances, so that $2D_B = D_A$, where D stands for marginal damage. Assume, moreover, that extrapolation between laboratory experiments and human impact reveals that airborne emissions of A and B will impact human health. In this hypothetical case, the relationship between D_B and D_A means that an individual is twice as likely to experience the effects of morbidity from exposure to additional emissions of A.

An increase in the prevalence of health disorders may take many different forms, including emphysema, chronic bronchitis, lung cancer and others. Each pollutant by itself and in conjunction with other may lead to different maladies. An identification of a specific relationship between pollutants assumes that a meaning for damage (pathways or endpoints) is adopted. Herein lies the problem of economic valuation. It would be subjective for a regulator to choose a particular disorder or number of disorders that result from chemical pollution, value the consequences in dollar terms and use the results to form policy. One or many other impacts may go unnoticed. However, it may not be possible in many cases to link pollution to particular health problems. Economic valuation of unknown problems is impossible. As a result, many externalities cannot be reliably quantified as a result of a lack of scientific knowledge (ORNL/RFF, 1994).

The point is that a regulator has to choose the particular impacts to quantify in dollar terms. While a thorough measure of marginal damage is needed, economic valuation may prove impossible for each endpoint. The availability of information may therefore dictate the application of valuation techniques to marginal damage. Returning to the example with chemicals A and B, suppose researchers find that an applicable dose-response function links a twenty-four hour average concentration of chemicals A and B to the probability that a child experiences a cough-day and an adult chest discomfort. These are the only identifiable impacts. Assume research shows

that the probability of sickness varies directly with airborne concentration of chemicals A and B. Regional impacts would result from population concentrations and meteorological trends.

Assume also that regulated companies increase production from a baseline case which raises the releases of A and B. For a target population not close to point sources of the pollutants, research finds that a marginal increase in concentration of A would lead to a higher midpoint number of child cough days per year of 100 and adult chest discomfort days of 50 relative to the base case. ORNL/RFF (1994) value cough days at $4.77 and chest discomfort days at $6. A marginal increase in emissions, then, would be valued yearly at $477 + $300 = $777 for chemical A and $388.50 for B. Total tax payment for the polluting firm would have to equal these amounts if additional emissions occur. If two firms have equal emissions, each would pay half the total. If policy makers target pollutants with relative damage values, the method could derive tax payment.

Suppose now that researchers determine that, along with an increase in chest discomfort and cough days, the pollution would reduce fish catches at one nearby lake. In terms of ($HV_{fishLC50}$), an increase in the concentration of chemicals A and B would reduce fish populations. Assume travel cost research estimates a consumer surplus value equal to $300 for anglers at the lake. By adopting environmental quality as one explanatory variable, a researcher finds that a decrease in recreational demand, based on a smaller catch relative to a baseline case, would lead to a decrease in consumer surplus, equal to $100 per angler per year. If ten anglers use the lake, marginal damage is valued at $1000 per year. Adding this number to the previous amount gives $1777, which represents the total increase in damage. Tax payment must account for this figure.

This marginal approach analyzes the change in the baseline level environmental and health conditions following an increase in chemical emissions. Identification of a target population and environment are important in the analysis. The potential for sickness and degradation is a function of the capacity of effluent to impact individuals or aspects of environmental quality that people

value. Furthermore, the economic values focus on willingness to pay. As a result, human preference forms the basis of the economic valuation of marginal damage from changes in pollution concentrations. Finally, other techniques could weigh the monetary valuations differently. The impact on human health, for example, could receive twice the weight for changes in recreational activity. The contribution of each depends on the regulator's view of relative importance.

The crucial aspect in the implementation of fully informed policy—and the basis for the difference between this policy and risk-based taxation—is the choice of pollution impacts to value in economic terms and the valuation methods themselves. Not every damage pathway is possible to include in a formulation of marginal damage of a particular pollutant. Consider some potential impact pathways in Figure 5, noting that morbidity could include a number of ailments.

Researchers cannot make every correlation between pollutant and impact for human health and environmental effects outside a laboratory. Too many variables are involved. In addition, correlation does not mean causation. An increase in pollution may be correlated with some change in health, but one change may not cause the other. When choosing impact pathways, the researcher must consider if *a priori* evidence exists of market failure, potential damage to third parties is large, potential damage is due to additional emissions, observed damage is likely an externality and if the baseline is identifiable (ORNL/RFF, 1994, 4-35 - 4-37). The point is that endpoints from the CCPCT index—which are adopted to constitute damage—are chosen subjectively. Their adoption in a DFA means that others are excluded correctly or incorrectly.

The magnitude of externalities depends on the location of emission sources, nearby populations and potential impacts. A large population center located downwind from a number of sources would experience a higher level of damage than a town located upwind. Marginal damages from a single pollutant therefore vary across regions. The results from one set of data (such as the CCPCT index) are not universally applicable. Multiple calculations would improve estimates.

Discharge or residual	Potential impact
Airborne chemical emissions	*Health effects*:
	Mortality
	Morbidity
	Environmental effects:
	Effects on materials
	Visibility
	Recreational fisheries
	Other recreational activities
	Impacts on crops
	Forests
	Ecological effects
	Aesthetic effects
Emissions in water	*Health effects*:
	Mortality
	Morbidity
	Environmental effects:
	Fish loss
	Recreation loss
	Ecological effects
	Aesthetic effects
Land use	*Health effects*:
	Mortality
	Morbidity
	Environmental effects:
	Ecological effects
	Aesthetic effects

Sources: ORNL/RFF (1994) and Hershaft (1976)

Figure 5
Potential impact pathways from chemical emissions

Research has concentrated more on certain aspects of the economic valuation of pollution damage. Most of the effort has gone to the theory and estimation of recreational benefits as well as the value of reduced mortality. A lot of research has also considered the valuation of pollution effects on economic production, visibility and health. Few studies, however, have calculated the value of reducing mortality risks from improvements in environmental quality (ORNL/RFF, 1994, 4-69). The results from the valuation of marginal damage from chemicals A and B, therefore, depend crucially on the body of valuation literature. While a particular impact may be relevant, no studies may be available to provide a dollar value of pollution damage.

Payment from Risk-Based and Fully Informed Taxation

While a fully informed regulation applies a monetary valuation to the damage impacts of target chemical releases, the choice of pathways, the weighting of the importance of each, the choice of pathways to value in monetary terms and the weighting of the economic valuations makes the procedure subject to policy choice. The policy maker may incorporate a measure of damage that encompasses a number of pollution effects (endpoints), including those on humans and the environment. When this choice occurs, the monetary valuation of the damage of individual chemical pollutants depends on available research, valuation techniques and the relative importance of the results. If the emissions of chemical A, for instance, impact a large population and a number of recreational areas, which impact pathways are more important than others? Are all equally important? Valuation of marginal damage depends crucially on a regulator's decisions.

The calculation of tax payment would differ for the fully informed and risk-based levies. With proper dose-response modeling and economic valuation (steps three and four of the DFA), a fully informed policy is possible. This regulation implements a rate that equals the calculated economic value of marginal damages. In an optimal framework (which this paper does not

assume), the rate would be set to equate marginal damage with marginal abatement cost. In the absence of information on sufficient impact pathways chosen by a regulator and economic valuation, risk-based regulation would set prices on a relative basis according to each pollutant's RUs. The key is the tax rate on one RU, a subject addressed in chapter four.

Returning to Figure 2 in chapter one, total damage equals the area under the curve at the level of emissions E_o. With fully informed taxation, policy would establish a rate so that payment would equal the economic value of pollution damage from a marginal change in emissions, relative to a baseline case. If incremental emission units are described in small quantities, for example in pounds, and the policy maker wants to regulate flow over time, the tax rate would be set to equal the economic value of marginal damage from additional units. The key with this regulation is that the economic valuation provides an accurate measure of marginal damage in dollar terms.

The difference between an optimal, fully informed policy and one that is not optimal involves the difference between formulating the tax rate to equate marginal damages and marginal abatement costs and not adopting this approach. The previous examples calculate the economic value of a marginal increase in pollution and argue for tax payment to equal this amount. This approach corresponds to the methods adopted by ORNL/RFF (1994). On a marginal basis, however, the tax rate would not equate marginal damage with marginal abatement costs.[63] As a result, the tax policy attempts to internalize the value of a marginal increase in pollution, but not reduce emissions to their optimal level. This issue of practical tax implementation is considered again in the mathematics-of-policy-choice section in chapter four.

In contrast, risk-based taxation assumes that a monetary valuation of the marginal damage of the target chemicals from the TRI is unknown. As a result, the policy develops a proxy for a "true" marginal damage function for individual pollutants on a relative basis, as shown by Figure 4

[63] At any rate, the regulator may not know marginal abatement costs with certainty.

in chapter three. The damage values are higher for chemical emissions that pose a greater degree of risk to humans and the environment. Focusing on relative risk, the risk-based framework does not reveal how the tax rate should be set. Presumably, the rate should come as close as possible to equaling the economic value of marginal damage from individual pollutants. But problems of pollution synergy, a lack of information on marginal abatement costs and other factors—described in the "Weaknesses" subsection below—make rate determination an imprecise procedure.

The risk-based tax approximates a fully informed levy when the error term, or difference between true and approximate marginal damage, equals zero.[64] When does this occur? If marginal damage from a target chemical increases at a relatively stable rate, as in Figure 4, the risk-based proxy will reflect the slope and marginal damage. When pollution synergy from target chemical emissions does not alter the relative nature of the hazard values from the CCPCT index, the values provide workable proxies. At its best, risk-based taxation sets a rate that comes close to approximating true marginal damage on a relative basis for target chemicals. At its worst, the rate does not come close to equaling marginal damage.

STRENGTHS OF THE CCPCT INDEX AND RISK-BASED TAXATION

The CCPCT framework generates a linear relationship between emissions and damage. This relationship aids policy formulation for the risk tax, but does not share the characteristics of "traditional" damage functions that show varying levels of damage over a range of emissions. The damage relationships posited by risk-based taxation serve as a proxy for those of a fully informed policy. The damage functions do not accurately show how a measure of damage goes up as individual chemical emissions increase.

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[64] This topic is discussed further in chapter four, along with the issue of setting the rate for the risk-based levy.

A strength of the policy is the relative formulation of damage values. This formulation aids the implementation of tax policy as regulators may levy an environmental tax according to an approximate value of risk. The tax administrator assigns risk units to a pound of each chemical emission according to the CCPCT index and levies a single tax rate on one RU.

The risk-based system regulates a number of chemical emissions by applying RUs. This encompassing framework allows firms to react to consistent policy. Traditional environmental tax literature discusses the issue of optimal taxation. The optimal rate occurs at the point where marginal damage equals marginal abatement cost. At this optimum point, many authors disregard the marginal benefits of pollution reduction since it equals the marginal cost of abatement. As a result, the literature focuses largely on the cost side of the problem: polluters will reduce emissions toward the optimal level as long as the tax rate exceeds marginal abatement cost.

But the risk-based system recognizes the difficulty of obtaining cost information. In addition to being unknown to regulators, abatement cost information may be unknown to firms (Adar and Griffin, 1976, 179). As Fishelson explains, "The uncertainty of the optimal policy level...(results) from the uncertainty of the costs associated with lowering pollution" (1976, 189). With the coverage of multiple emissions under the jurisdiction of the risk-based policy, the optimal level of emission reduction is unknown to regulators.

The emphasis on the damage side of the problem therefore stems from reasons of practicality. With a lack of cost information, one feasible way to implement incentive policy is to compare emissions with the risk unit. (The risk-based policy suffers the same fate as a fully informed environmental tax in the absence of abatement cost information. It does not find the optimal level of emission reduction.) The system eliminates the administrative problem of setting different tax rates for chemical emissions according to unknown levels of marginal damage. In a comprehensive manner, the risk-based tax system attempts to elicit an intended response among

polluters and reduce emissions. The system does not operate in a first- or even second-best world; therefore, it adopts an nth-best framework. It is intended for practical use.

WEAKNESSES OF THE CCPCT INDEX AND RISK-BASED TAXATION

One drawback of the use the CCPCT index is that the hazard values do not change with emissions. The discussion in the beginning of this chapter addresses the fact that damage functions may take non-linear forms, depending on wind patterns, the concentration of pollution in the environment, and proximity of pollution sources to humans and animals. The linearity assumption from risk-based taxation therefore limits the understanding of marginal damage.

The damage values generated by the CCPCT algorithm are not site-specific. As a result of the variability of the pathway endpoints at different sites, the relative risk values of the CCPCT index do not rank a set of emissions accurately for different pollution situations. The ranking methodology of the CCPCT index provides a starting point in the formulation of a system of risk-based taxation. As the subsection "Improving the Algorithm" in the chapter five explains, additional research that applies the algorithm to a set of chemical emissions on a site-by-site basis would improve the relative nature of the damage values for a specific area of jurisdiction.

The formulation of hazard values for individual pollutants in the CCPCT index does not take into account the effects of synergy between different chemical emissions. When pollutants interact in water, for instance, the combined effect on a particular endpoint may differ than if either pollutant impacts the endpoint separately. In particular, two or more pollutants could interact in any pollution situation, depending on the propensity of chemicals to interact, ingestion patterns of humans and animals, the degree to which pollution already exists in an area, and many other factors. Interaction between pollutants could increase or potentially decrease the damage effect on

a particular endpoint as compared to the hazard value from the CCPCT index. The hazard values are therefore limiting. A number of chemical emissions could interact in any pollution situation, clouding the impact of a particular pollutant.

In sum, problems of linearity, pollution impact and synergy make the use of the CCPCT index and risk-based taxation a less-than-perfect policy system. Further research could address each of these issues. Even though it levies one tax rate and is practical from a policy standpoint, the risk-based system possesses certain limitations.

WHEN RISK-BASED TAXATION IS APPROPRIATE

To address policy appropriateness, it is first necessary to explain how uncertainty enters into the analysis. A discussion of uncertainty reveals more caveats concerning policy application. At the firm level, uncertainty enters into the formulation of marginal abatement costs. This uncertainty stems from unproven control measures, unpredictable cost savings from future economies of scale and unknown future input prices. The governing authority also faces uncertainty with individual firms' cost functions. This translates into an unknown aggregate control cost function for society. In addition, the pollution control agency does not have information on social marginal damage functions (Adar and Griffin, 1976, 179). The hypothetical cases below explain under what circumstance the risk-based system constitutes an appropriate policy (Case 1), and when it would not provide accurate results (Case 2).

Case 1: Using a common measure of damage, a policy maker wants to tax a number of chemical emissions in a relative manner. In the absence of information that provides *direct, marginal damage* values in a relative manner for target pollutants, the regulator implements risk-based taxation. Suppose marginal damage from target pollutants increases in a stable manner. In this situation, risk-based taxation serves as an appropriate proxy for fully informed policy.

Case 2: When target chemical emissions do not lead to linear damage functions, risk-based taxation may regulate the pollutants with consistent policy, and perhaps serve as an appropriate policy—depending on the number of pollutants under regulation—but would not price individual chemicals accurately.

An issue fundamental to the risk-based system is the availability of information. The damage values of individual pollutants must be compared on a relative basis. With risk-based taxation, the assumption from the policy perspective is that the regulator wants to achieve a certain degree of continuity. In Case 1, the CCPCT index provides damage information that approximates the true measure of damage for target pollutants. In Case 2, risk-based taxation may be chosen to regulate a set of chemical emissions, but the pricing mechanism does not provide an accurate measure of marginal damage for target chemicals.

WHEN FULLY INFORMED REGULATION IS APPROPRIATE

With the target set of chemical emissions in Davis *et al.* (1994c), *marginal damage function research with common toxicological endpoints* is not available for individual pollutants. As a result, individual chemical emissions cannot be measured with some common damage value without the use of an indirect framework (like the CCPCT index.) The difficulties of developing relative marginal damage functions for a number of chemical emissions and applying an economic valuation to each emphasizes this point. If the regulator wants to target the set of pollutants in a relative manner in the CCPCT index, or some subset, marginal damage function information (with different pathway endpoints) on individual pollutants may be used, but the application of a different tax rate to each pollutant is necessary. Cases 3 and 4 explain scenarios when a fully informed tax serves as an appropriate policy.

Case 3: Suppose a regulator targets one chemical pollutant with an environmental tax. If the relationship between emissions and damage is known for the region under jurisdiction, and an economic valuation of the measure of damage is available, a fully informed levy serves as an appropriate environmental policy.

Case 4: If a policy maker wants to regulate a number of chemical emissions, damage function relationships for the individual pollutants must define a measure of damage that allows economic valuation on a marginal basis. For instance, the measure of damage must entail impacts on humans and the environment for a comprehensive measure. A technique must establish a method to value individual endpoints and aggregate in order to implement accurate policy.

CONCLUSION

The risk-based tax policy uses relative damage values to levy environmental taxes on chemical emissions. The benefits of the system include the emphasis on relative damage for target chemicals, the ease of administration, and the possibility of including a number of pollutants in the policy. The weaknesses include the linear nature of damage estimation, a lack of site-specific estimates, and problems of synergy.

CHAPTER 4

IS A RISK-BASED TAX BENEFICIAL?

The portfolio decision means that, when implementing tax policy, the regulator must balance the benefits of certain criteria (environmental gains and revenue recycling) against the costs (administration, compliance and tax interaction). Chapter four focuses on one portfolio issue: efficiency. In the current environmental tax (double dividend) literature, this concept motivates theoretical models. As a result, the efficiency criterion is used to evaluate the risk-based system. The analysis of chapter four, however, does not intend to mitigate the importance of the other portfolio criteria from chapter one. The implementation of policy without consideration of all costs and benefits would leave the regulator with an incomplete assessment of the potential outcome. (Chapter five analyzes the portfolio problem as it applies to risk-based environmental taxation in the context of all evaluation criteria.)

Another important issue concerns the degree of difference between the risk-based system and fully informed policy. It is important to offer some concrete comparisons between the two systems. Beyond the discussion in chapter three, under what conditions is the risk tax a good proxy? How does it relate to true damages? What are the assumptions that make that link? For example, damage is a per-unit measure. Is damage truly constant per unit? What other types of assumptions are embedded in this measure? Even though they are the topics of chapter four, the development and explanation of these points is an extension of chapter three.

In a mathematical framework, chapter four first contrasts environmental taxes in terms of how they adjust marginal private cost to account for marginal social cost. Next, chapter four identifies target pollutants, discusses ways to set the tax rate and simulates revenue for Tennessee

using different rates. Chapter four then details the efficiency effects of Parry (1995) that result from the implementation of environmental taxation. Revenue substitution is assumed to be a political necessity for the implementation of the new environmental tax system (Pearce, 1991). The development of the Parry framework includes the tax bases previously discussed (output, inputs and emissions), and a specific analysis of the risk-based system. Chapter four shows that risk-based taxation provides incentive to reduce the most harmful chemical emissions.

REPRISE: WHEN ENVIRONMENTAL TAXES ARE APPROPRIATE

Environmental taxes reduce pollution when they achieve environmental goals and increase economic efficiency. In particular, when the tax base is linked to the pollution source, an increase in the relative price of the dirty input, output or emission encourages the use of cleaner alternatives. A Pigouvian tax prices pollution damage. Fully informed emissions and risk-based taxes establish an association between policy and potential for environmental improvement. A tax rate that exceeds marginal abatement cost will alter production behavior.

But the incentive policy may or may not impact target firms. Since an environmental tax increases costs to the firm if it does not replace existing regulation, the levy may distort future production decisions or exacerbate pre-existing tax distortions. The risk-based system may therefore alter tax-system efficiency. The relevant question is whether efficiency losses occur. The answer depends on the extent of improvements in environmental quality and assumptions concerning a number of key variables, which are analyzed below. To evaluate a policy's impact, a regulator must weigh the benefits of environmental improvement against any efficiency costs. Many authors adopt this framework in the environmental tax literature. But much of the research focuses on efficiency losses and excludes environmental gains.

MATHEMATICS OF POLICY CHOICE

The potential to tax different bases to address the chemical emission externality provides an opportunity to regulate pollution directly or indirectly. In the environment, the chemical emission externality will persist when pollutants create a divergence between private and social costs. The choice of emissions as base creates a regulation that prices the source of pollution. In the absence of environmental regulation, the price of output created in a dirty production process reflects input costs and market competition, but does not include the value of environmental damage from the chemical emissions. Social costs, however, include private costs to the firm and any pollution costs suffered by society. The choice of regulatory instrument will impact how the social costs are internalized in the production process. This dissertation argues for an incentive policy—environmental taxation on emissions—to address the chemical emission externality.

From the perspective of the regulator, the idea is to adjust marginal private cost to reflect the value of environmental damage imposed by the chemical effluent. Taxes on output, chemical inputs or emissions are linked differently to the source of damage. If emissions are directly related to the production of output, and output is a function of a number of inputs, a reduction in either the use of dirty inputs or the creation of output will lower emissions. But a tax on output or inputs does not directly discourage the emissions of harmful pollutants. Only a tax on the chemical emission itself provides the necessary incentive for emission abatement. In the context of the externality problem, how would the taxation of different bases affect the potential for embedding the social costs of pollution in private cost, or, in other words, increase welfare? In addition, how do alternative shapes for marginal damage impact the analysis? Which taxes would best account for marginal damages? The following presentation addresses these questions in a mathematical framework. The framework shows how each tax accounts for marginal damage. The regulation of a single pollutant is discussed first and multiple emissions second. Possible tax bases include

output, inputs or emissions. Emissions-based levies are analyzed in terms of a risk tax, a simple emissions tax and fully informed regulation.

The Case of Damage from One Pollutant

Assume that output (X) is a function of capital (K), labor (L) and chemical inputs ($Z_1, \ldots, Z_n$): $X = X(K; L; Z_1, \ldots, Z_n)$. Emissions from one chemical input serve as the only source of environmental costs. In the absence of regulation, the original level of emissions of chemical one (E_{Z1O}), for example, are proportional to the production of output: $E_{Z1O} = \phi X$. The implementation of environmental regulation may reduce emissions to E_{Z1R}, where $E_{Z1R} \leq E_{Z1O}$. Total emissions, E, equal the sum of individual emissions of chemical inputs Z: $E = \Sigma E_i$ where $i = 1, \ldots, n$. The representative firm's internal production costs C_1 are a function of output, $C_1(X)$, while control costs C_2 are a function of emission abatement and the production of final output, $C_2 = C_2(E_O - E_R, X) = (\phi X - E_R, X)$. The implementation of an environmental tax means that polluting firms make annual tax payments to the governing authority.

Damage (D) is assumed to be an increasing function of chemical emissions: $D(E)$. Damage may increase at an increasing, decreasing or constant rate for individual effluent. An efficient level of emissions and output requires that marginal social cost equal marginal private cost. In addition, marginal damage cost must equal marginal control cost: $C_{21} = \delta D/\delta E$. No reason exists for the latter equation to hold in the absence of regulation.

The choice of policy instrument influences total private and total social costs. The taxation of particular base may or may not lead to the abatement of chemical emissions. From the perspective of the policy maker, the choice of a particular base offers the potential to link directly or indirectly to the source of pollution damage. From society's perspective, the optimal policy will

internalize the externality fully by embedding the value of social costs into private costs. In the absence of perfect information, however, it is impossible to determine the economic value of the optimal level of pollution damage as a result of information limitations. So policy effectiveness may be judged on the basis of the ability of an individual tax to account for the best estimate of marginal damage at a target quantity of emissions. Since the optimal level of pollution abatement is unknown, the model takes a second-best approach.

The following equations[65] show how total social cost, total private cost, marginal social cost and marginal private cost are influenced by the choice of policy instrument. First assume a regulator decides to implement a per-unit output tax, t_X, that does not vary with emissions. For the sake of simplicity in this initial presentation of examples, suppose environmental damage results from the emissions of one harmful chemical input. The model also includes the possibility of multiple emissions in the next subsection. The output tax could serve as an appropriate policy in the presence of high monitoring costs, no options for pollution reduction except a decrease in output, and a high substitutability of the final product (Schmutzler and Goulder, 1997). With the tax on output, the firm does not encounter a price for the chemical emission and therefore does not think of it as an input in the production process. Any reduction in emissions results from a lower level of output. In this case, the firm's total private cost, TPC_X, equals

$$TPC_X = C_1(X) + t_X X.$$

In the absence of emission abatement, total social cost, TSC_X, is

$$TSC_X = C_1(X) + D(E_{Z1O}) = C_1(X) + D(\phi X).$$

The difference between the two costs equals

$$TSC_X - TPC_X = D(\phi X) - t_x X.$$

[65] The following presentation is adapted from Dodds and Lesser (1994). This paper provides emissions and output equations for total and marginal costs.

The extent that this difference is greater than, equal to, or less than zero depends on the level of tax revenue from the environmental policy. If the cost difference exceeds zero, tax payment does not account for total damage. Using available information to determine the best estimate of the value of pollution damage, no reason exists why tax revenue may necessarily equal damage in the case of the output tax, unless damage is constant as emissions increase. But this functional form could occur for an individual firm that emits a small amount of one chemical emission relative to the total emitted by industry. The charge, in other words, could be set to reflect damage in this case. With the above equations, tax payment will exceed total damage for any tax rate that is set above $t'_x \leq \phi D'$. In the case where marginal damage is an increasing function of emissions, a per-unit output tax $t'_x < \phi D'$.

Changes in prices (tax rates) should be based on the difference between marginal private and marginal social costs. Marginal private cost, MPC_X, and marginal social cost, MSC_X, equal the following equations, respectively:

$$MPC_X = C_1' + t_x,$$

$$MSC_X = C_1' + \phi D'.$$

The difference between the two costs equals

$$MSC_X - MPC_X = \phi D' - t_x.$$

This value will equal zero if the tax is set at the optimal rate to equal the economic value of pollution damage. Otherwise, the difference will be positive or negative as the tax rate is less than or greater than the optimal tax rate, respectively.

Instead of an output levy, suppose the regulator decides to tax one chemical input, Z_1, as a result of harmful emissions. In their paper on the derivation of environmental tax rates in the presence of market distortions, Bovenberg and Goulder (1996) show in a mathematical framework

that the optimal rate on a "clean" intermediate input should equal zero. In this case total private cost is

$$TPC_Z = C_1(X) + t_{Z1}(X) = C_1(K; L; Z_1, \ldots, Z_n) + t_{Z1}X(K; L; Z_1, \ldots, Z_n).$$

Control costs are not included in TPC_Z since the taxation of an intermediate input does not lead to emission abatement. The change in relative prices encourages the switch to cleaner inputs, if this change is possible. Total social costs in this case equal

$$TSC_Z = C_1(K; L; Z_1, \ldots, Z_n) + D(E_{Z1}).$$

When damage is a function of the emissions of a number of chemicals, the right-hand-side equation is altered. With one target chemical pollutant, the difference between the two costs is

$$TSC_Z - TPC_Z = D(E_{Z1}) - t_{Z1}X(K; L; Z_1, \ldots, Z_n).$$

When the chemical input and its emissions display a direct correspondence, a tax on dirty inputs may account for the pollution externality. The extent that the difference between TSC_Z and TPC_Z is greater than, equal to, or less than zero depends on the regulator's ability to determine the economic value of pollution damage and set the input tax accordingly.

In the presence of the input tax, marginal private cost and marginal social cost equal the following two equations, respectively:

$$MPC_Z = \delta C_1/\delta X + t_{Z1},$$

$$MSC_Z = \delta C_1/\delta X + D'dE_{Z1}/dX.$$

The difference between the two costs equals

$$MSC_Z - MPC_Z = D'dE_{Z1}/dX - t_{Z1},$$

where a positive, negative or zero value will result when t_{Z1} is less than, greater than or equal to some $t'_{Z1} \leq t^*_{Z1}$, where the latter tax is optimal in that it internalizes the pollution externality. Again, it is unlikely that a regulator would set the marginal tax rate on an input at an appropriate level to fully internalize the pollution externality given the constraint of information limitations.

As opposed to output and input levies, a tax on emissions provides the incentive for emission abatement. By providing a permanent price for the source of pollution, the tax rate may account for marginal damage. The question is how the regulator chooses to set the rate. If it is set on a per-unit basis, a problem arises concerning the equality of the rate with changes in marginal damage. As emissions increase, the damage value from incremental units of pollution may increase at varying rates. As a result, a constant tax rate will not account for all levels of marginal damage at emissions increase. A rate that varies to account for changes in damage value, however, would propose problems from the point of view of business. Tax payment would be difficult to calculate. The policy maker may also levy a tax rate equal to the value of marginal damage from the last unit of emissions, and apply the rate to all units. In other words, the regulator would have to know the shape of the damage function, as in Figure 6.

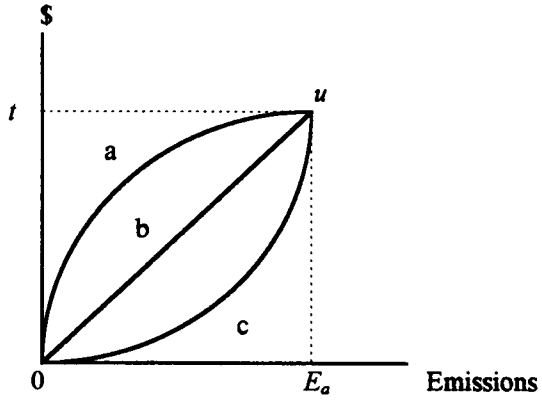

Figure 6
Alternative marginal damage functions

Figure 6 shows three possible shapes for marginal damage functions. Function a increases at a decreasing rate; b at a constant rate; and c at an increasing rate. Assume E_a represents the

annual emission level. At E_a, the three marginal damage functions come together at point u. If a policy maker sets the per-unit emission tax rate at t, total tax payment will equal tE_a, or the area of the rectangle $0tuE_a$. Whereas, at the margin, the tax rate equals marginal damage, total damage for functions a, b and c—the area under the curve from the origin to the emissions level—is less than tax payment. As they are drawn, total tax payment comes the closest to equaling total damages for function a, and the farthest for function c. Damage function b, which represents the estimate of marginal damage from the risk-based tax framework, lies in the middle. If E_a equals the optimal emission level, then t would reduce effluent to this point. But in the absence of this knowledge concerning optimality, the issue concerns the ability of regulators to make tax payment equal the best estimate of total damage for the polluter.

If a policy maker decides to tax emissions, waste disposal in the environment becomes a priced input like other resources. Suppose monitoring costs are low, effective end-of-pipe technologies exist and the sources of emissions are identifiable (Schmutzler and Goulder, 1997). In this case, the emissions tax will increase the firm's cost of production by the amount of control costs and tax payment. The risk-based tax (t_{RU}) is multiplied by both a proxy (RU) for marginal damage and by emissions to generate tax payment. The total private cost that results from the risk-based levy is

$$TPC_E = C_1(X) + C_2[\phi X - E_{ZIR}(RU\, t_{RU}), X] + t_{RU}(RU)E_{ZIR}.$$

Total social cost from the implementation of the regulation equals

$$TSC_E = C_1(X) + C_2[\phi X - E_{ZIR}(RU\, t_{RU}), X] + D(E_{ZIR}).$$

The difference between the two costs is

$$TSC_E - TPC_E = D(E_{ZIR}) - t_{RU}(RU)E_{ZIR}.$$

The extent that this value equals zero depends on how well the proxy RU and the tax rate account for the economic value of true damage at the annual emission level. As chapter three explains, the

risk-based system may accurately form approximate damage functions when marginal damage increases at a stable rate.

Changes in tax rates should reflect differences between marginal private and social costs. In the context of the risk-based tax, marginal private cost equals

$$MPC_E = \delta C_1/\delta X + C_{21} + C_{22} + t_{RU}RU(dE_{Z1R}/dX),$$

and marginal social cost is

$$MSC_E = \delta C_1/\delta X + C_{21} + C_{22} + D'(dE_{Z1R}/dX).$$

The difference between MPC_E and MSC_E becomes

$$MSC_E - MPC_E = D'(dE_{Z1}/dX) - t_{RU}RU(dE_{Z1}/dX) = (D' - t_{RU}RU)dE_{Z1}/dX.$$

The difference will be positive, equal zero or negative as $t_{RU}(RU)$ is less than, equal to or greater than a tax $t_{RU}` < t_{RU}^*$. The latter rate represents an optimal environmental tax that internalizes the externality. In the presence of uncertainty, the only way for the risk-based levy to approximate a fully informed regulation is if damage is relatively stable. If marginal damage is increasing, tax payment will exceed total damages from the polluting firm.

In limited cases, full information could provide a policy maker with the ability to internalize the pollution externality. The formulation of a fully informed tax policy would require information about the economic value of pollution damage, which would dictate the level of tax payment. Perhaps if the impact from a single chemical emission is not significant and the pollutant does not interact with many other chemicals, the measurement of damage could be accurate over time. In any case, the estimation of damage is crucial for the implementation of a fully informed regulation. Total private cost for a fully informed emissions tax (t_e) is

$$TPC_E = C_1(X) + C_2(\phi X - E_{Z1R}(t_e), X) + t_eE_{Z1R}.$$

Total social cost equals

$$TSC_E = C_1(X) + C_2(\phi X - E_{Z1R}(t_e), X) + D(E_{Z1R}).$$

External cost means the divergence between TPC_E and TSC_E. This difference equals

$$TSC_E - TPC_E = D(E_{Z1R}) - t_e E_{Z1R}.$$

A fully informed regulation results in a situation where $t_e = t_e^*$ and tax payment equals the economic value of pollution damage. Considering problems of data collection in the formulation of a damage pathway, the existence of pollution synergy, unpredictable meteorological conditions, problems of site-specific estimation and many other factors, a fully informed tax is unrealistic from the point of view of practical policy implementation for target chemicals from the TRI.

Marginal conditions for this fully informed levy show that the tax rate is set to equal marginal damage. Marginal private and social costs are the following two equations, respectively:

$$MPC_E = \delta C_1/\delta X + C_{21} + C_{22} + t_e dE_{Z1R}/dX,$$

$$MSC_E = \delta C_1/\delta X + C_{21} + C_{22} + D' dE_{Z1R}/dX.$$

The difference between the two costs is

$$MSC_E - MPC_E = D' dE_{Z1R}/dX - t_e dE_{Z1R}/dX = (D' - t_e) dE_{Z1R}/dX.$$

This result equals zero as the fully informed regulation internalizes the pollution externality.

Finally, a simple emissions tax would be set at a rate to raise a target level of revenue but may or may not reflect marginal pollution damage. The levy would not incorporate the different impacts of pollution on humans and the environment. The simple emissions tax system is not Pigouvian in nature, but would allow firms to estimate their tax payment ahead of time with emission projections. Total private cost from the implementation of this tax (t_S) is

$$TPC_E = C_1(X) + C_2(\phi X - E_{Z1R}(t_S), X) + t_S E_{Z1R}.$$

Total social cost, on the other hand, is

$$TSC_E = C_1(X) + C_2(\phi X - E_{Z1R}(t_S), X) + D(E_{Z1R}).$$

The difference between the two provides external cost:

$$TSC_E - TPC_E = D(E_{Z1R}) - t_S E_{Z1R}.$$

No reason exists why the simple emissions tax would internalize the externality from chemical Z_1. The difference will be positive, equal zero or negative as t_S is less than, equal to or greater than a tax $t_S` < t_S^*$, where the latter levy is optimal in that it internalizes the externality.

As before, appropriate regulatory decisions depend on the difference between marginal private and social costs. With the simple emissions tax, MPC_E is

$$MPC_E = \delta C_1/\delta X + C_{21} + C_{22} + t_S dE_{Z1R}/dX.$$

Marginal social cost equals

$$MSC_E = \delta C_1/\delta X + C_{21} + C_{22} + D' dE_{Z1R}/dX.$$

The difference between the two costs is

$$MSC_E - MPC_E = D' dE_{Z1R}/dX - t_S dE_{Z1R}/dX = (D' - t_S) dE_{Z1R}/dX.$$

This difference will most likely be positive as the simple emissions tax does not account fully for the pollution externality.

Table 17 shows the adjustments that a regulator may make to *MPC* to account for the existence of *MSC*. The extent that marginal social cost differs from marginal private cost depends on the shape of the relevant marginal damage function. At a given level of emissions and private production cost, the difference between *MSC* and *MPC* may be greater for a pollutant's damage path that increases at an increasing rate. The cost difference depends on the level of emissions and marginal damage at the target level. Welfare increases as marginal private cost is adjusted to include marginal social cost that is not accounted for previously.

How do the five taxes compare in their ability to internalize the pollution externality? In the presence of information limitations concerning the shape of a damage function, it would be more difficult for a regulator to set tax rates on output and inputs. Schmutzler and Goulder (1997) show mathematically and graphically, however, why an environmental tax on output or inputs may still be appropriate. But from the point of view of the policy maker, it may be easier to set

marginal tax rates on emissions to reflect marginal damages with an information constraint. The three emissions-based taxes, however, have different strengths and weaknesses in terms of environmental effects. The fully informed tax operates in the presence of accurate environmental valuation techniques and measures of marginal damage. But this information does not exist for many pollution situations. The risk-based tax may apply to any chemical pollutant with a corresponding number of risk units, but may not accurately price the externality with damage functions that are not stable or impacted by pollution synergy. Finally, the simple emissions tax lowers administrative costs, but does not attempt to account for pollution damage.

Table 17
Adjustments to MPC to account for pollution damage

Environmental Tax	Adjustment to MPC
Output	$\phi D' - t_x$ greater than, equal to, or less than zero as t_x is less than, equal to, or greater than t_x*
Input	$D'dE_{ZI}/dX - t_{ZI}$ greater than, equal to, or less than zero as t_{zI} is less than, equal to, or greater than t_{zI}*
Risk-based emissions	$(D' - t_{RU}RU)dE_{ZIR}/dX$ greater than, equal to, or less than zero as t_{RU} is less than, equal to, or greater than t_{RU}*
Fully informed emissions	$(D' - t_e)dE_{ZIR}/dX$ greater than, equal to, or less than zero as t_e is less than, equal to, or greater than t_e*
Simple emissions	$(D' - t_s)dE_{ZIR}/dX$ greater than, equal to, or less than zero as t_s is less than, equal to, or greater than t_s*

The Case of Damage from Multiple Emissions

A review of the Toxics Release Inventory reveals many businesses that emit a number of chemicals simultaneously. The synergy between chemical emissions may exacerbate or alleviate environmental damage, depending on the molecular composition of the pollutants, meteorological conditions and other factors. The formulation of private and social costs differs when

environmental policy targets multiple emissions. To limit the extent of the presentation, this subsection focuses exclusively on the emissions-based levies.

In contrast to the previous examples, damage is now assumed to stem from the emissions of chemicals Z_1 through Z_{10}: $D(E_{Z1}, \ldots, E_{Z10})$. Each pollutant impacts target endpoints in particular ways, depending on a number of factors discussed in chapter three. But chemical interaction creates the potential for pollution synergy, which is difficult to account for when measuring the economic value of environmental damage.

The three emissions-based levies would address the externality problem from different angles. The risk-based tax provides a proxy for marginal damage in the form of a risk value for chemical effluent. A strength of this system (chapter three) is the implementation of one tax rate that translates different pollution prices to individual emissions. Weaknesses result when the system cannot account for the effects of pollution synergy or non-linear damage functions. The extent that the risk-based levy approximates a fully informed regulation determines policy effectiveness. In the presence of multiple emissions and risk-based taxation, total private cost is

$$TPC_E = C_1(X) + C_2[\phi X - \Sigma E_{iR}(RU_i\, t_{RU}), X] + t_{RU}\Sigma(RU_i)E_{iR},$$

where $i = 1, \ldots, 10$. Total social cost becomes

$$TSC_E = C_1(X) + C_2[\phi X - \Sigma E_{iR}(RU_i\, t_{RU}), X] + D(\Sigma E_{iR}).$$

External cost, or the difference between TPC_E and TSC_E, is

$$TSC_E - TPC_E = D(\Sigma E_{iR}) - t_{RU}\Sigma(RU_i)E_{iR},$$

where i = 1,..., 10. For policy formulation, the risk-based tax system applies a number of RU to each chemical emission to approximate damage. The extent that tax payment accounts for total environmental damage of the ten target chemicals depends on the accuracy of the system. This accuracy depends, in turn, on the degree of equivalence between the CCPCT proxies and "true"

marginal damage at annual emissions levels. External cost will be greater than, equal to, or less than zero as t_{RU} is less than, equal to, or greater than some $t_{RU}` \leq t_{RU}^*$.

With the risk-based system, marginal private cost equals

$$MPC_E = \delta C_1/\delta X + C_{21} + C_{22} + t_{RU}\Sigma RU_i(dE_{ZiR}/dX),$$

and marginal social cost is

$$MSC_E = \delta C_1/\delta X + C_{21} + C_{22} + \Sigma D'(dE_{ZiR}/dX).$$

The difference between MPC_E and MSC_E becomes

$$MSC_E - MPC_E = \Sigma D'(dE_{Z1R}/dX) - t_{RU}\Sigma R_i(dE_{Z1R}/dX) = (\Sigma D' - t_{RU}\Sigma RU_i)dE_{ZiR}/dX.$$

The variable D' represents marginal damage of multiple emissions. If the tax rate (t_{RU}) multiplied by the proxy (RU) for individual effluent approximates the true economic value of pollution damage, then this regulation results in an effective policy. If the tax rate is set too high, or if pollution synergy increase marginal damage from individual chemicals, then the result above will be negative or positive, respectively.

With fully informed regulation, a single tax rate would not be able to account for varying levels of damage from a number of chemical emissions. If a policy maker wanted to regulate a set of pollutants, an individual tax would have to apply to each. Large information requirements would exist in terms of site-specific estimation, the damage variation that result from synergy, uncertainty with respect to damage effects on humans and the environment and many other factors. In contrast to the risk-based levy, the fully informed policy would increase the costs of administration and compliance, but would internalize the externality. Total private cost after the implementation of a fully informed levy on each emission (t_{ezi}) is

$$TPC_E = C_1(X) + C_2(\phi X - \Sigma E_{ZiR}(t_{ei}), X) + t_{e1}E_{Z1R} + \ldots + t_{e10}E_{Z10R}.$$

Total social cost equals

$$TSC_E = C_1(X) + C_2(\phi X - \Sigma E_{ZiR}(t_{ei}), X) + D(\Sigma E_{ZiR}),$$

where, as before, $i = 1,\ldots, 10$. The difference between the two costs is

$$TSC_E - TPC_E = D(\Sigma E_{ZiR}) - t_{e1}E_{Z1R} - \ldots - t_{e10}E_{Z10}.$$

The series of fully informed taxes on each chemical emission accounts for total pollution damage, the external cost equals zero and the externality is internalized.

The difference between marginal total and marginal social cost determines how the policy maker adjusts the tax rate for individual emissions. With fully informed regulation, marginal private cost equals

$$MPC_E = \delta C_1/\delta X + C_{21} + C_{22} + \Sigma t_{ei}(dE_{ZiR}/dX),$$

and marginal social cost is

$$MSC_E = \delta C_1/\delta X + C_{21} + C_{22} + \Sigma D'(dE_{ZiR}/dX).$$

The difference between MPC_E and MSC_E becomes

$$MSC_E - MPC_E = \Sigma D'(dE_{ZiR}/dX) - \Sigma t_{ei}(dE_{ZiR}/dX) = (\Sigma D' - \Sigma t_{ei})dE_{ZiR}/dX,$$

where $i = 1,\ldots, 10$. The fully informed regulation means that the policy maker sets the rate on each chemical emission to internalize the externality. Rates will differ according to the level of marginal damage of each pollutant.

A simple emissions tax does not distinguish between different levels of marginal damage. A single rate to regulate the target pollutants, this tax would not be implemented unless in limited cased if chemical emissions display similar damage effects or for reasons of administrative ease. With multiple emissions, total private cost with a simple emissions tax (t_S) is

$$TPC_E = C_1(X) + C_2(\phi X - \Sigma E_{ZiR}(t_S), X) + t_S\Sigma E_{ZiR}.$$

Total social cost, on the other hand, is

$$TSC_E = C_1(X) + C_2(\phi X - \Sigma E_{ZiR}(t_S), X) + D(\Sigma E_{ZiR}).$$

The difference between the two provides external cost:

$$TSC_E - TPC_E = D(\Sigma E_{ZiR}) - t_S \Sigma E_{ZiR}.$$

External cost will exist if the simple emissions tax is set too low and tax payment is less than total damage or the difference is positive.

Price changes depend on the difference between marginal private and social costs. With the simple emissions tax, MPC_E is

$$MPC_E = \delta C_1/\delta X + C_{21} + C_{22} + t_S \Sigma dE_{ZiR}/dX.$$

Marginal social cost equals

$$MSC_E = \delta C_1/\delta X + C_{21} + C_{22} + \Sigma D' dE_{ZiR}/dX.$$

The difference between the two costs is

$$MSC_E - MPC_E = \Sigma D' dE_{ZiR}/dX - t_S \Sigma dE_{ZiR}/dX = (\Sigma D' - t_S) dE_{ZiR}/dX.$$

This difference will most likely be positive as the simple emissions tax does not account fully for the pollution externality.

In terms of practical policy implementation, the fully informed regulation would apply in limited cases when detailed information is available to calculate the economic value of pollution damage. The risk-based tax could regulate target emissions in the presence of stable damage functions and weak effects of emission synergy. The simple emissions tax has limited applications and is not realistic from a policy perspective. Table 18 summarizes the adjustments to price for each emissions-based environmental tax in the case of multiple emissions.

Taxation in the Presence of Pre-existing Regulation

In the presence of pre-existing environmental regulation, a portion or all of the chemical emission externality may already be internalized. It is also possible that environmental policy

could over-regulate firms by increasing cost more than the value of pollution damage. In the absence of information for a fully informed policy on all pollutants and pre-existing control, additional regulation becomes an nth-best problem. *Further policy implementation would be deemed necessary only if the regulator could improve on existing regulation.* The job of the policy maker could be to provide a permanent incentive for emission abatement in the form of taxation. But if the marginal costs of additional policy exceed marginal benefits, welfare losses occur. If appropriate, the implementation of an environmental tax in the presence of pre-existing regulation should account for the economic value of any remaining portion of the externality not already internalized.

Table 18
Adjustments to MPC with multiple emissions

Environmental Tax	Adjustment to MPC
Risk-based emissions	$(\Sigma D' - t_{RU}\Sigma RU_i)dE_{ZiR}/dX$ greater than, equal to, or less than zero as t_{RU} is less than, equal to, or greater than t_{RU}*
Fully informed emissions	$(\Sigma D' - \Sigma t_{ei})dE_{ZiR}/dX$ greater than, equal to, or less than zero as t_e is less than, equal to, or greater than t_e*
Simple emissions	$(\Sigma D' - t_s)dE_{ZiR}/dX$ greater than, equal to, or less than zero as t_s is less than, equal to, or greater than t_s*

In this case, the problem for the policy maker is to work with any existing regulation. Heavy policy burden would threaten industry competitiveness and discourage business location. So the governing authority must be careful when considering additional policy. An environmental tax could complement direct control by providing the incentive for continuous abatement with a pollution price. But in an economy with many pre-existing distortions (such as with the tax system or inefficient environmental policy), addressing them one at a time may not increase welfare. Nevertheless, if a policy maker decides to adjust marginal private cost to account for any

remaining social costs, each value becomes constrained. Any subsequent price adjustments would account for the constraint. The environmental taxes in the context of the previous subsection would adjust the "constrained marginal private cost" for "constrained marginal social cost." In this situation, *MTC* and *MPC* in Table 18 would become constrained variables.

The Impact of Policy Choice on Profit Maximization

The taxation of alternative bases impacts profit maximization. A tax on output will lower quantity demanded. Depending on price elasticity, this price increase could increase or decrease total revenue. An input tax will raise production costs, as will a tax on emissions. Profit (Π) equals total revenue (TR) minus total cost (TC). Since $X = X(K; L; Z_1, \ldots, Z_n)$, each resource requires an input price for the calculation of total cost. With multiple chemical inputs, the profit function is

$$\Pi = P_x X - P_k K - P_l L - P_{Z1} Z_1 - \ldots - P_{Zn} Z_n,$$

where P_x is the price of output, and input prices are represented by the subscript on the coefficient for each resource. A tax on emissions—if it exists—increases the firm's production cost. The scenarios below do not assume pre-existing regulation or that the environmental tax replaces any policy. Suppose a competitive firm in which the implementation of an environmental tax does not alter the use of the inputs K or L or their prices.

The examples entail a comparison of output, input, risk-based, fully informed and simple environmental taxes. Suppose $n = 3$ while the use of Z_1 and Z_2 leads to harmful emissions. The price of output and all inputs equals \$1. Initially, output produced and sold is one hundred units per production cycle. Ten units of each of the five inputs is required to produce one hundred units of output. Assume zero elasticity of substitution between inputs. The firm emits one unit of

emissions from each chemical unit during the production cycle: thirty chemical inputs lead to thirty units (pounds) of emissions. In the absence of an environmental tax, profit equals $50.

Consider first a tax on output of $0.10 per unit. When the tax is imposed, the marginal cost of selling the product increases by ten percent per unit. In addition to covering all other variable costs, the producer must cover the cost of the tax to avoid losses. This effect reduces supply. If the firm is operating in the elastic portion of the demand schedule, the price increase will also reduce quantity demanded. The decrease in supply results in a new post-tax equilibrium, say, at ninety units of output. The consumer pays a higher gross price (P_G) of $1.05 for the product. This is the price received by sellers. But the sellers must pay $0.10 per unit of the gross price received as tax. The net price (P_N) is therefore $0.95 per unit. The difference between P_G and P_N is the tax. Total revenue from the tax is $9.00 = ($0.10)(90). Since ninety units are produced and sold, and the relationship between inputs to output is one to ten, the post-tax profit equals

$$\Pi = TR - TC = (\$0.95)(90) - (\$1)(45) = \$40.50.^{66}$$

Now suppose a tax on the chemical inputs that lead to harmful emissions. The levy raises the input prices to the producer by a value $(1 + t_z)$. If the tax is $0.10 per unit to reflect approximate pollution damage of chemical emissions, the prices of Z_1 and Z_2, the dirty chemical inputs, become $1.10 = $1(1 + 0.10). Since these specific inputs are needed in fixed proportion in the production cycle, the increase in the prices of Z_1 and Z_2 does not cause the firm to substitute less-expensive alternatives. Total revenue from the input tax equals $2.00 = ($0.10)(20). With the production of 100 units of output, profit equals

$$\Pi = (\$1.00)(100) - [(\$1)(10) + (\$1)(10) + (\$1.10)(10) + (\$1.10)(10) + (\$1)(10)]^{67}$$

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[66] Excess burden from the tax, assuming linear demand and supply functions, equals $0.50. Fifty percent of the tax is borne by the seller and fifty percent by the buyers.

[67] As it is originally organized, the numbers in the total cost bracket represent K, L, Z_1, Z_2 and Z_3, respectively.

$$= \$100 - \$52 = \$48.$$

An emissions-based tax would add to total costs. Suppose it is found that marginal damage from Z_1 is twice as harmful as marginal damage from Z_2. The actual economic value of marginal damage at the annual level of emissions of Z_1 is \$0.24 and of Z_2 is \$0.12. In a risk-based framework, the emissions from Z_1 are assigned a risk unit (RU) value of two while the emissions of Z_2 receive a value of one. In an attempt to internalize the externality, the policy maker decides to apply a tax rate on one *RU* of \$0.10.[68] With risk-based taxation, the profit function becomes

$$\Pi = P_x X - P_k K - P_l L - P_{Z1} Z_1 - P_{Z2} Z_2 - P_{Z3} Z_3 - t_{RU}(RU_i) \Sigma E_{Zi}.$$

Chemical Z_3 receives a *RU* value of zero since it does not lead to harmful emissions. After the implementation of the risk-based levy, profit equals

$$\Pi = \$100 - \$50 - t_{RU}(RU_i) \Sigma E_{zi} = \$50 - (\$0.10)(2)(10) - (\$0.10)(1)(10) = \$47.$$

Tax revenue to government equals \$3.

A fully informed regulation would set the tax rate equal to the actual level of marginal damage of chemical emissions. The profit function becomes

$$\Pi = P_x X - P_k K - P_l L - P_{Z1} Z_1 - P_{Z2} Z_2 - P_{Z3} Z_3 - t_{ez1} E_{Z1} - t_{ez2} Z_2 =$$

$$\$100 - \$50 - (\$0.24)(10) - (\$0.12)(10) = \$46.40.$$

Government tax revenue is \$3.60.

A simple emissions tax would entail the same rate on all emissions. In this last case, a rate of \$0.10 per unit of emissions to raise a certain level of revenue will lead to profit of

$$\Pi = P_x X - P_k K - P_l L - P_{Z1} Z_1 - P_{Z2} Z_2 - P_{Z3} Z_3 - t_s E_{Z1} - t_s Z_2 - t_s E_{Z3} =$$

$$\$100 - \$50 - (\$0.10)(10) - (\$0.10)(10) - (\$0.10)(10) = \$47.$$

In this last case, the government collects \$3 in tax revenue.

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[68] See the subsection below, "Setting the Tax Rate," for a discussion on how a policy maker may decide to levy different rates in alternative situations.

Policy Observations

What conclusions result from this section on the mathematics of policy choice? First, the impact of individual environmental taxes depends on production assumptions. In the above example, the environmental output tax lowers profit the most as consumers reduce quantity demanded. This result is sensitive to demand elasticity. A more inelastic product would lead to an after-tax equilibrium above ninety, which would not reduce profit to $40.50. The other four levies reduce profit by relatively the same amount. This result leads to the second point: different regulatory schemes may allow the firm to make "appropriate" tax payment, even if the base does not link directly to the source of pollution. If the appropriate payment for the representative firm is around $3.60, the emissions-based levies (not fully informed) and the input tax raise close to this amount. As production and output increase, however, tax payment would begin to deviate from the appropriate amount at greater levels. Third, a tax on emissions applies a price to the source of damage directly. This regulation creates a value for waste disposal. The use of a chemical input is not harmful to the environment per se if the residual is properly contained. Fourth, if the cost of environmental regulation is small relative to total cost, firms may not adjust production, as the policy is not substantial enough. A high-enough price for a particular input may alter the substitution elasticity. Fifth, market structure would influence the impact of an environmental tax on profit. Ceteris paribus, the extent that the seller can pass the cost on to the consumer would raise profit. Sixth, the difficult task for the regulator is the application of the appropriate rate. The problem is setting the tax rate so that total tax payment equals total damages. But the implementation of the rate to internalize pollution damage cannot occur without knowing the definitive shapes of damage functions from target chemical emissions. In the absence of this information, a policy maker may set the rate equal to marginal damages at a chosen level of

emissions. The level is optimal with full information on abatement costs and marginal damages or sub-optimal at the annual level of emissions. In the absence of information to implement optimal tax policy, the risk-based system offers a proxy for true marginal damages.

SHIFTING TO A RISK-BASED SYSTEM

To implement a risk-based tax system, policy makers first have to target chemical pollutants and set the rate. They then have to identify polluters, develop an administrative system, and evaluate the costs and benefits of the proposed policy. The following subsections address the first step and then contrast the environmental and efficiency effects of the taxes (output, input, fully informed emissions and risk units) of chapter two. An empirical extension then calculates the welfare impact of the risk-based taxation in a simulation experiment using different values for key parameters. Discussion of step two is reserved for chapter five.

Target Pollutants

The risk-based system could initially target the 158 chemical emissions from the CCPCT index, or a subset of this list, as discussed in chapter one. As more relative damage information becomes available over time, the policy system could incorporate additional pollutants. Consider that the 1990 Amendments to the Clean Air Act contain provisions to reduce toxic air emissions by 75% within ten years. The regulatory method entails the application of a maximum achievable control standard (MACT). The law currently lists 189 toxic pollutants,[69] but allows EPA the flexibility of adding or deleting emissions from the policy. In addition, major sources (that release over ten tons every year of one pollutant or twenty-five tons of a combination of pollutants), gas

[69] For a list of these chemicals, point to www.law.cornell.edu/uscode/42/7412.html on the internet.

stations, and dry cleaners are regulated. By 1995, EPA had to list enough area sources to account for 90% of the emissions of the thirty most toxic pollutants from the list. The law allows EPA to phase in the program over ten years.

The risk-based system could eventually include the toxic chemicals from the above MACT list, others in addition to the 189 toxic pollutants, and toxic water and ground releases. Incidentally, the risk-based levy system could supplement these provisions of the Clean Air Act or eventually replace the direct regulation altogether. Taxation provides the incentive for continuous innovation over time, a concept absent from a policy of direct control. After the implementation of the technology standard, for instance, polluters could become complacent. They may not innovate further. Risk-based (or another emissions-based) regulation would continuously price emissions.

Consider that Copeland (1992) includes 192 chemicals (three conventional pollutants and the rest toxic) in her environmental tax system to regulate wastewater discharges. Copeland devises a tax framework that uses available toxicity information from the EPA. The toxicity values are formulated according to the impact of the pollutants on human health and aquatic life. Copeland targets the 192 chemicals as a result of available information. Incidentally, the risk-based policy in this dissertation covers a wider range of chemical emissions than Copeland's system. The risk-based framework regulates more than wastewater discharges.

Setting the Tax Rate

How should a policy maker formulate the tax rate in the absence of perfect information on the economic value of damage for the target chemical pollutants? The regulator wants to price pollution damage, internalize the externality, and move society towards the optimal level of pollution without knowing where the optimal level is. With perfect information, the tax would account for the economic value of marginal damage of individual pollutants at the optimal level,

where marginal abatement costs equal marginal damages. In the absence of marginal abatement cost and fully informed marginal damage functions, the regulator would want an approximation of the economic value of marginal damage. At the very least, the risk-based tax system levies a charge on chemical pollutants that accounts for damage at annual emission levels using a proxy for the damage function. The rate must reflect an economic value of marginal damage for annual quantities of chemical emissions.

An indirect method exists to examine whether the charge established by the risk-based system would impact production behavior at the margin. By comparing the charge per unit of emissions to marginal abatement costs (when available), the regulator would have an idea if the tax would reduce emissions when applied to a particular chemical pollutant. If abatement cost exceeds the tax rate and no behavioral adjustment occurs, the charge could be adjusted.

In a recent paper, Bovenberg and Goulder (1996) analyze the impact of different carbon tax rates on efficiency. For the purpose of a tax simulation, the authors vary the damage value for one ton of carbon emissions between twenty-five and 100 dollars per ton of emissions. The authors find that revenue-raising policies lead to lower efficiency losses than regulations that achieve the same environmental benefits but do not provide revenue for the government. The point is that Bovenberg and Goulder assign tax rates to pattern the assumed damage values at *one ton of carbon emissions* in their simulation experiment. In fact, the authors call the tax that sets a charge equal to marginal damage of an additional ton of carbon emissions an "optimal Pigovian tax" (Bovenberg and Goulder, 1996, 992).

In the context of the risk-based system, at least five possibilities exist for setting the tax rate. First, the policy maker could set the tax to achieve a target level of revenue. The authority could then implement a policy of revenue substitution, generate additional money for the general budget, or fund a particular project. In her toxicity-based tax system to regulate wastewater, for

example, Copeland (1992) taxes 192 chemical emissions to achieve a target level of revenue. Second, the policy maker could price one risk unit in terms of its perceived environmental damage. The tax rate that translates to individual emissions would then reflect different damage values. A problem with this possibility, however, is the variability of damage across regions. Assume, for example, that one risk unit equals a pound of chlorobenzene emissions, as in the examples from chapter three. In a closed laboratory environment, a pound of chlorobenzene emissions would impact human health and the environment in a particular way. But with unpredictable atmospheric conditions, alternative environmental characteristics and different population centers, *actual* damage varies across regions. Another problem with this application entails the economic valuation of emissions. Since different endpoints formulate damage, economic valuation becomes an imprecise procedure. Third, policy makers may believe that a certain tax rate would impact production behavior at the margin. The governing authority could have information from businesses, for instance, that reveals abatement costs for different emission levels. A certain tax rate could make it profitable for firms to reduce pollution. Fourth, one could adjust tax rates to achieve a pollution standard. This system would pattern the "charges and standards approach" developed in Baumol and Oates (1988), Chapter 11. The problem with this approach, however, is knowing what the standard should be. Fifth, the tax rate could pattern or exceed the rates suggested for carbon taxes or the price of SO_2 permits. Recent research suggests a carbon tax rate of $25-$50 per ton (Goulder, 1995), while the price for annual SO_2 permits fluctuates around $100 per ton (see the EPA web page.)

The subsections that follow analyze the efficiency impacts of setting different tax rates on one RU. The lowest charge initially adopted in the risk-based framework equals the proposed upper limit for carbon taxes ($50/ton) while the highest rate exceeds the price of an SO_2 permit ($100/ton.) This approach is adopted for the following reason. Compared to carbon and SO_2

emissions, the chemical substances regulated by the risk-based system entail high damages per unit. This impact results from chemical composition, toxicity, persistence and bioaccumulation. As a result, the charge levied on the risk unit reflects the potential for higher levels of economic damage than would result from equivalent carbon or SO_2 emissions.

A comment from Baumol and Oates (1988) may contribute to this analysis. In part two of their book that explains practical methods for environmental regulation, the authors conclude that, as a result of restrictive information requirements, the calculation of the economic value of pollution damage at the optimal level is an impossibility. "Instead of trying to go directly to the optimal tax policy, as a first approximation, one could base a set of taxes...on the current net damage (benefit) levels. In turn, as outputs and damage levels were modified in response to the present level of taxes, the taxes themselves would be readjusted to correspond to the new damage levels" (Baumol and Oates, 1988, 161). These authors are describing an iterative process where a regulator uses previous information to calculate future tax adjustments. The tax rates described above, then, may be thought of as an initial set of proposed rates. As a policy authority learns how firms react to the risk-based taxes, the rates could be adjusted over time.[70]

After the implementation of the tax rate, a policy maker could evaluate the charge by comparing it to marginal abatement costs (MAC) *if they are known*. Suppose, for example, that industry in the state of Tennessee emits 500 pounds of trichlorobenzene in a year. At a tax rate of $1 per risk unit, total tax payment would equal $1000, since one emission pound equals two RU. As a ratio of $/emissions, $1000/500 = $2. If MAC for this chemical are greater than $2 per pound of emissions, the tax rate would be too low to influence behavior. If MAC are less than this ratio, the policy would provide the incentive for pollution reduction. If MAC are not constant over

[70] Since firms incorporate taxes in their cost estimations, it is not politically feasible to propose a new tax system that would rely on rates that would change consistently. The regulator would have to take this planning factor into account.

the range of emissions, the policy maker would have to compare the tax rate to MAC at the actual level of emissions.

Revenue Calculations

The risk-based policy would generate annual tax revenue by assigning the risk units from Table 16 to target emissions. To calculate annual revenue for Tennessee, data have been gathered for the 158 chemical emissions from 1994 that Davis *et al.* (1994c) include in their study. As chapter one explains, it may be beneficial from society's point of view to ban the use of the most harmful toxic chemicals. For this reason, chemicals one through ten from Table 11 were deleted from the data set since they possess the highest damage values in the CCPCT index. This deletion includes PCBs. In addition, chemicals in the data set that are severely restricted (Table 1) were also eliminated from the data set. (In Appendix 4, this deletion is shown by having no emission pounds for these chemicals.) After eliminating these pollutants, 146 chemical emissions remain. Even though the risk-based system could regulate other chemicals, this list encompasses many emission compounds.

Using the information from Davis *et al.* (1994c), a total hazard value is assigned to each pollutant on a scale from one to 100. If one pound of chlorobenzene emissions equals one risk unit, every other chemical represents a number of risk units relative to this numeraire. Appendix 4 presents this information.

This subsection employs different tax rates to calculate annual revenue for the state of Tennessee. To implement a price of \$100 per ton to pattern the cost of one SO_2 permit, an equivalent tax rate of \$0.05 per RU is applied since one pound of chlorobenzene emissions equals one RU. A tax rate of \$0.025 per RU, moreover, equals \$50 per ton. The following equation constitutes the revenue function:

$$G_R = (t_{ru}) \sum_{i=1}^{n} (RU_i)(e_i),$$

where G_R = government revenue; t_{ru} = tax on one risk unit; RU_i = number of risk units of the ith chemical emission; e_i = the ith chemical emission; and n = 146. By applying different tax rates to one risk unit, the amount of state tax revenue is calculated and summarized in Table 19. Adding the revenue from each chemical emission at a particular rate gives the total revenue for the risk-based system.

Table 19
Revenue calculations with risk-based taxation

Tax Rate Per Ton of Emissions	Tax Rate Per Risk Unit	Total Revenue from Risk Tax	% Revenue from 20 Most Harmful Emissions (For Each Tax Rate)
$50	$0.025	$2,238,947	7%
$75	$0.0375	$3,358,420	7%
$100	$0.05	$4,477,894	7%
$125	$0.0625	$5,597,367	7%
$150	$0.075	$6,716,840	7%
$200	$0.10	$8,955,787	7%
$500	$0.50	$44,778,935	7%

Table 19 reveals that the state government of Tennessee could use revenues from the risk-based tax contribute to its general budget. (Appendix 4 shows the total revenue for a tax rate of $0.025 per RU.) To put Table 19 in perspective, Tennessee collected almost three billion dollars in revenue from the sales tax in 1994. The gasoline levy raised 500 million dollars. Appendix 5 presents Tennessee's main revenue sources for 1994 when total revenue was $5,314.1 million. A $0.50 rate per risk unit would raise approximately 0.8% of the total 1994 tax revenue.

The risk-based system implies a different rate per emission pound, depending on the pollutant. With a risk unit value equal to one, chlorobenzene has a tax-rate-per-unit-of-emissions

value of $0.025/1 when the rate is $0.025 per risk unit. (In the numerator, the tax rate is multiplied by the risk units. The denominator is one emission pound of the pollutant.) Table 20 reveals that the implied tax rate per unit of emission goes up as the damage value of the pollutant increases.

Table 20
Implied rate per emission for selected chemicals

Chemical	RU	Emission (lbs.)	Tax rate/risk unit	Tax rate/emission
Zinc	0.52	1	$0.025	$0.013/1 = $0.013
Chlorobenzene	1.00	1	$0.025	$0.025/1 = $0.025
Trichlorobenzene	2.00	1	$0.025	$0.05/1 = $0.05
Trifluralin	2.03	1	$0.025	$0.0507/1 = $0.0507

Consider a simple emissions tax levied on the 146 chemical pollutants for the sake of comparison. Not realistic from a policy implementation perspective, this tax does not distinguish between the damage value of individual emissions. Given the same tax rate on each pollutant (per pound, not risk unit), the simple emissions tax raises three percent of its revenue from the twenty most toxic emissions when applied to the same 146 pollutants (Appendix 4). In contrast, the twenty most toxic emissions account for seven percent of the total revenue with the risk tax (Table 19 and Appendix 4). The risk-based tax provides incentive to reduce the emissions of the most hazardous substances.

Problems with the Double Dividend Literature

The application of the risk-based tax system at the state level would raise revenue for the governing authority, but would also lead to tax-system distortions (in the context of Bovenberg and de Mooij, 1994). In addition, the tax policy would presumably lead to benefits from enhanced

environmental quality. Would the revenue gains and environmental benefits outweigh tax-system efficiency losses? The answer depends on the tax rate and important assumptions concerning the substitutability between labor, leisure and consumption. The double dividend literature offers structure to address this issue.

The double dividend literature has evaluated environmental taxes in terms of efficiency consequences and revenue potential.[71] Early papers on the topic concluded that welfare gains would result from environmental improvements and reductions in excess burden [Terkla (1984), Pearce (1991), Oates (1991), Repetto *et al.* (1992) and Von Weizsacker and Jesinghaus (1992)]. It was assumed that revenues from environmental taxes would finance lower rates on levies that distort economic activity. But as chapter two explains, recent research has not affirmed the double dividend hypothesis [Bovenberg and de Mooij (1994), Parry (1995) and Bovenberg, Parry and Burtraw (1996)]. This subsection argues that the omission of environmental benefits from these latter papers constitutes an important drawback to the welfare conclusions.

When calculating welfare changes from environmental taxation, the latter three papers introduce benefits and costs but go on to analyze tax system effects (costs) only. Why does this occur? Preoccupation with this second dividend of environmental taxation results from uncertainties in measuring the magnitude of the environmental benefits of the first dividend (Goulder, 1995, 158). Thus from the outset, the theoretical papers address a question concerning one of the impacts of environmental taxation, even though they purport to be "double" dividend analyses.

The original purpose of environmental taxes is to help reduce pollution. The revenue/tax system issues are subtopics within the realm of environmental management. The first (environmental) dividend should not be excluded from analyses of environmental taxation. The

[71] See Oates (1995) and Goulder (1995a) for an overview of the topic.

exclusion of the economic value of environmental improvement makes the welfare conclusions from the double dividend literature incomplete.

As the double dividend literature explains, a policy of environmental taxation coupled with revenue substitution leads to three effects: (1) environmental, (2) revenue-recycling (RR) and (3) tax-interaction (TI) [Oates (1995) and Goulder (1995)]. The first explains the impact of the levy on environmental quality. The second effect represents a decrease in tax-system distortions from a lower income tax. The third effect captures the welfare costs of the introduction of the environmental tax, which may include the substitution of more leisure for labor. With the implementation of an environmental tax, the first two effects are found to have a positive impact in the research, while the latter impact is negative. Bovenberg and de Mooij (1994), Goulder, Parry, and Burtraw (1996), and Parry (1995) conclude that the tax-interaction effect outweighs its revenue-recycling counterpart under plausible assumptions, while disregarding the benefits of enhanced environmental quality.

By ignoring these benefits, the authors assume that models may separate the costs and benefits of environmental management and analyze each effect individually. This separation implies the calculation of costs in one experiment, the formulation of benefits in another, and finally a comparison of the two figures (Gillis *et al.*, 1996, 2). The problem with this approach is that benefits and costs are intertwined. While many of the aforementioned authors calculate the costs of environmental regulation, the benefits are impressive. A cleaner environment, for instance, raises labor productivity, which in turn increases investment. This impact could compound benefits over time (Gillis *et al.*, 1996, 5).

SIMULATION EXPERIMENT

The following theoretical conception and empirical application extend an existing framework to address the benefits and costs of the risk-based tax policy. A policy maker would benefit from having the calculations of the efficiency effects of environmental tax systems. As a result, the model isolates the three efficiency effects (environmental, tax interaction, and revenue recycling) for the different tax bases discussed in chapter two. Using the notation of Parry (1995)—with minor changes to fit the format of the following subsections—the theoretical discussion extends his analysis of output and input taxes to include fully informed emissions and risk-based levies.

In his 1964 paper on "The Measurement of Waste," Harberger explores a number of ways to calculate the deadweight loss of an economy in sub-optimal positions. Harberger derives equations for the welfare impacts of different tax combinations, including levies on output, capital, and labor. The presentation of Parry (1995) is a straight-forward application of Harberger's system. The former author's welfare analysis analyzes the impacts of environmental taxation in a model with pre-existing tax distortions.

Environmental Output Tax

Suppose, as Parry does, that a competitive market economy produces final commodities, $X_1, \ldots, X_k$, and the production of X_j, a consumption commodity ($1 \leq j \leq k$), creates a waste flow in a polluting industry. Assume the pollution comes from the use of chemical inputs. All other goods are produced in environmentally clean production processes. To explain labor market interactions, Parry assumes a tax on full labor income and a permanent tax on the dirty commodity which does not spillover into the capital market; therefore, capital is excluded from the analysis. The model

does not include the impact of non-labor tax distortions and assumes constant marginal costs. The Parry framework makes the same assumptions of Harberger (1964) concerning the linearity of demand and supply functions. The economy does not internalize the negative externality in the absence of public intervention.

The following household budget constraint includes a proportional tax, t_L, on labor income:

$$\sum_{i=1}^{k} X_i + (1 - t_L)N = (1 - t_L)T + G,$$

where leisure time (N), and the household time endowment (T) mean that $T - N$ equals labor supply. The variable G represents an exogenous, lump-sum transfer. To simplify the presentation, Parry assumes the government returns all income tax revenues in such transfers: $G = m(T - N)$. With this framework, the government levies the labor tax on full income and subsidizes leisure.

Figure 7 displays equilibrium levels for the two distorted markets in the economy. All curves are compensated. In the dirty commodity market, S_j and D_j are the supply and demand of X_j, where X_j^o represents the original level of output. Since polluting emissions result from the production of X_j, marginal social costs exceed marginal private costs. To pattern Parry's framework, the $S_j + c$ schedule represents marginal social costs in Figure 7. If social costs equal marginal damage from chemical emissions, then the cost function will pattern the shape of the damage function: marginal social costs increase as the economic value of damage goes up. If damage increases at an increasing rate, so will marginal social costs.

For a given level of technology, the production of a certain quantity of output will lead to a specific level of chemical emissions. In equation form, emissions (e) are a function of output (X), and positively correlated: $e = f(X)$, $\delta e/\delta X > 0$. Marginal damages, moreover, result from the emissions in the production process. The fact that marginal social costs exceed private product costs reflects the existence of marginal social damage from different levels of production. The following analysis does not specify a technological relationship between emissions and output.

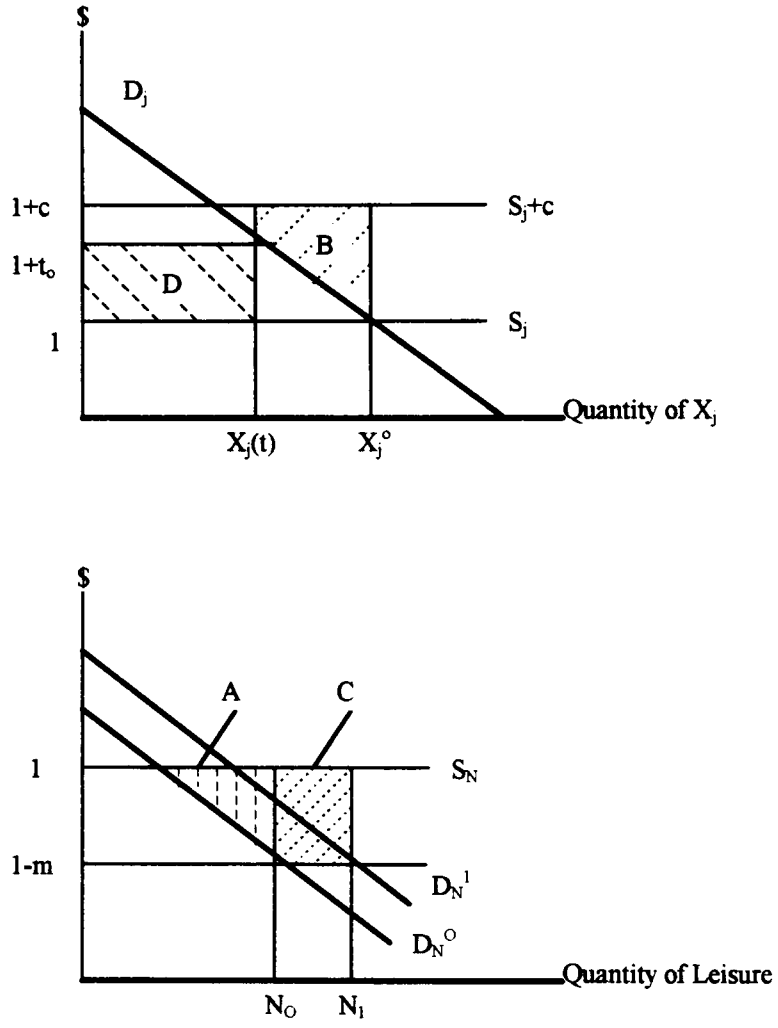

Figure 7
Welfare effects of environmental taxation:
output market (top) and leisure market (bottom)

This relationship would differ between firms. The presentation below assumes that emissions increase as output goes up. The more fundamental issue to evaluate is the relationship between marginal damage and emissions. As the production of output and emissions increases for alternative firms, the marginal damage from chemical emissions may rise at different rates.

Assume initially, as Parry does, that marginal social damage ($S_j + c$) is a function of emissions and constant (unrealistic) as production increases (Figure 7). (Alternative assumptions concerning the relationship between damage and emissions are explored below.) In the leisure market, D_N^O and S_N serve as leisure demand and supply. The subsidy to leisure creates a wedge, m, between the demand and supply price. The original level of leisure is N_O. As a result of the wedge, the welfare loss in the leisure market equals area A.

To make the Parry analysis more applicable, it must be adjusted. First of all, emissions are positively correlated with production for the polluting industry. Second, marginal damages from emissions increase as production goes up; however, Figure 7 displays constant marginal damages. This graphical presentation is unrealistic. To extend the analysis, Figures 8 and 9 assume that marginal damage from production (emissions) increases as output goes up. Figures 8 and 9 examine the possibility of marginal damage increasing at different rates.

Consider first the case when marginal damage—reflected through the variable c—increases linearly as emissions increase. In this case, marginal damage equals $ce(X)$. If c equals the scalar one, marginal damage rises at a constant rate as emissions increase. Table 21 presents damage calculations using the new assumption of increasing marginal damages. With alternative assumptions for the value of the scalar, c, different marginal damage functions result. A lower c results in a smaller quantity of damage at every level of emissions compared to the situation when c equals one. Figure 8 shows the marginal social cost function that corresponds to this first alteration of the Parry analysis when $c = 1$.

Table 21
Emissions and marginal damage: increasing at a constant rate

Quantity of emissions	Marginal damage (MD) = $ce(X)$, $c = 1$	MD = $ce(X)$, $c = 0.5$	MD = $ce(X)$, $c = 2$
1	1	0.5	2
2	2	1	4
3	3	1.5	6
4	4	2	8

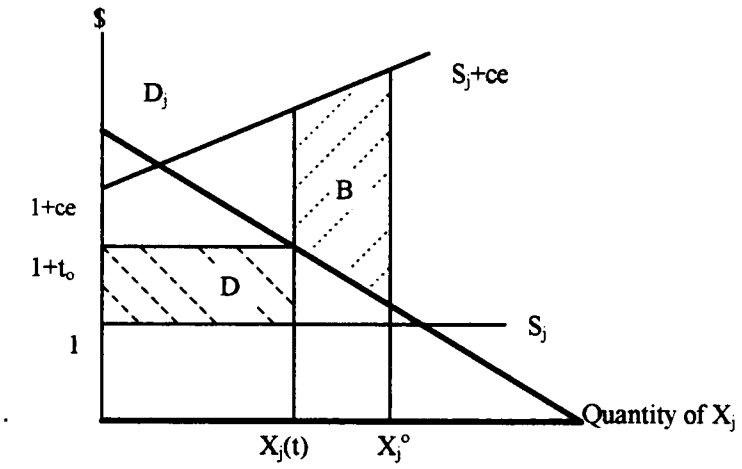

Figure 8
Marginal social costs (damage): increasing at a constant rate

The assumption of linearity in Figure 8 presents the same problem of narrowness encountered in Figure 4, which compared graphically the risk units associated with chlorobenzene and trichlorobenzene. Figure 4 demonstrates that the constant damage values from the CCPCT index imply linear damage functions for the chemical emissions included in the risk-based environmental tax system. Similarly, Figure 8 does not capture the range of relationships of the emissions of a particular chemical and environmental damage. The constant damage function is simplifying and narrow in scope and assumes away any non-linearities that may occur.

Marginal damage may also rise as a non-linear function of emissions. Most likely, marginal damage would not increase in a linear, stable manner as emissions flow from the production process. Table 22 addresses the case when marginal damage increases at an increasing rate. Table 22 lists different marginal damage values using alternative assumptions concerning the specific relationship between c and e. For damage to increase at an increasing rate, the marginal damage from the second unit of emissions must exceed the marginal damage from the first, and so forth. As with Tables 20 and 21, emissions are a function of the production of output in Table 22. Figure 9 demonstrates marginal damage increasing at an increasing rate.

Table 22
Emissions and marginal damage: increasing at an increasing rate

Quantity of emissions	MD = $c(e)$, $c = e^3 + 2e$	MD = $c(e)$, $c = e^{3.5} + e$	MD = $c(e)$, $c = e^4 + 2e + 1$
1	3	2	4
2	12	13.31	21
3	33	49.76	88
4	72	132	265

Another possibility exists. Marginal damage resulting from the flow of chemical emissions may increase at a decreasing rate in certain regions. A lake that is so saturated with pollution, for instance, could reach a point where additional chemical emissions would reduce environmental quality very little. Marginal pollution damage could increase with additional exposure, but the additional pollution could be incremental. In this instance the $S_j + c(e)$ curve could reflect this relationship between emissions and damage.[72]

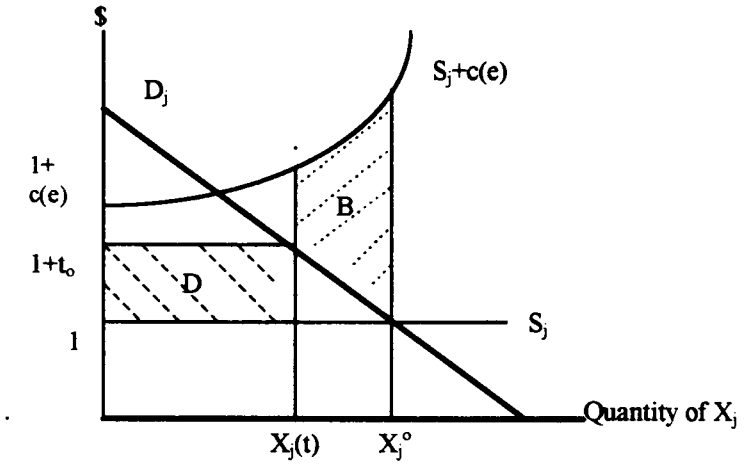

Figure 9
Marginal social costs (damage): increasing at an increasing rate

A governing authority that wants to fully internalize the chemical emission externality must set a tax on individual pollutants that equals the economic value of total damage at the annual pollution level. As Figures 7, 8 and 9 demonstrate, the regulator must take into consideration different levels of marginal social cost, depending on marginal damage from target pollutants.

Chapter two explains that the policy maker may choose to tax output, inputs, or emissions. Suppose, as Parry does, that the government first levies an output tax, t_o, on X_j that is produced in

[72] The table and figure showing this relationship are omitted.

the dirty production process, where $t_o \leq c(e)$. The tax is shown in Figures 7, 8 and 9. Assume also that the government implements a policy of revenue substitution out of political necessity. The revenue from the output levy finances a lower labor tax in a revenue-neutral policy. Consider now the three efficiency effects that result from the output tax. For the sake of continuity, Figure 8 is adopted as the relevant scenario.[73]

Choosing a different relationship between marginal damage and emissions than is assumed in the Parry framework impacts the value of the environmental effect that results from tax implementation. An improvement in environmental quality or reduction in marginal social cost would have to take into account non-linear damages with Figure 9. As Figures 8 and 9 are drawn, the difference between the environmental effects with the two diagrams is small. The actual difference, or error term, depends on the slopes of the curves and the degree of non-linearity.

(1) Environmental impact. The assumption is that emissions and damage are positively correlated. A tax on output will increase the price and reduce quantity demanded. With the implementation of the tax, output declines to $X_j(t_o)$ in Figure 8. Since damage increases as emissions (output) go up, the area B represents environmental improvement. Since emissions decrease when output goes down, an improvement in environmental quality occurs with an after-tax reduction in X_j. This levy does not tax the polluting source (emissions) directly, but raises the commodity price to discourage consumers from purchasing the product. Section B equals the rectangle $(ce - t_o)(X_j^o - X_j(t_o))$ plus the square $t_o(X_j^o - X_j(t_o))$[74]. The distance $(X_j^o - X_j(t_o)$ equals $-(dX_j/dp_j)t_o$, where p_j is the price of X_j. Adding these two areas gives

[73] The adoption of Figure 6 patterns the risk-based system from the CCPCT index. The use of a non-linear measure for marginal damage would improve design, but would not correspond to the results derived from chapter three. In other words, by analyzing the efficiency effects of a policy that assumes a non-linear damage function, the analysis would shift to a fully informed regulation as opposed to consideration of the proxy.

[74] Assume that the slope of the demand is equal to the slope of supply in absolute value. If this holds, the triangles above and below the rectangle in Figure 6 are equal.

$$t_o X_j^o \eta_{jj} ce, \qquad (1)$$

where $\eta_{jj} = -(dX_j/dp_j)/X_j^o$, the compensated own-price elasticity of demand of X_j at point X_j^o. If t_o equals ce and damages are constant (as in Figure 7) equation (1) is the familiar Harberger triangle.

(2) Tax-interaction effect. Parry assumes that leisure and X_j are substitutes. This important assumption impacts the results of the model. The TI effect occurs when the implementation of the tax on X_j increases marginal cost and raises the demand price for the product. Since X_j and leisure are substitutes, the higher price for X_j increase leisure time. This key result assures the that the taxation of dirty commodities will reduce welfare since people choose more leisure and real income decreases. In terms of Figure 7, the demand for leisure shifts up. At a given nominal wage, the amount that a consumer can purchase goes down. The new leisure demand schedule is D_N^1. As a result, leisure increases by $N_1 - N_O = (dN/dp_j)t$. Assume that the cross-price term, dN/dp_j, remains constant in the specific range. Since a wedge, m, exists between the supply and demand prices of leisure, marginal social cost exceeds marginal benefits by this same amount for each unit. A welfare loss, rectangle C, results from the increase in leisure:

$$m \frac{dN}{dp_j} t_o. \qquad (2)$$

The area C also represents the loss in labor tax revenues that occurs as a result of the implementation of the environmental tax.

Parry gives an equation for the marginal welfare cost of labor tax revenues. Consider first a result from the Slutsky symmetry property:

$$\frac{dN}{dp_j} = \frac{dX_j}{dw} \qquad (3)$$

where $w = 1-m$ is the net of tax wage or the price of leisure. Now, the marginal welfare cost of the labor tax revenue, V, represents the additional welfare costs that result from this revenue. The value equals the increase in welfare costs that result from a higher labor tax divided by the increase

in revenue. This efficiency effect comes into play when the environmental tax finances a lower rate on the labor tax. In equation form,

$$V = \frac{(m/1 - m)\varepsilon}{1 - (m/1 - m)e}, \qquad (4)$$

where $\varepsilon = -(dN/dw)(1 - m)/(T - N)$, the compensated labor supply elasticity. This elasticity is evaluated at w. Parry assumes that environmental tax revenues are small relative to the labor tax revenues. As a result, the change in leisure and m are small. Thus, V is relatively constant.

Using equations (2) and (3), and multiplying by and dividing by $(1 - m)X_j^o$, the TI welfare loss (area C multiplied by $(1 + V)$) is:

$$(1 + V)\frac{m}{1 - m} t_o X_j^o \eta_{lN}, \qquad (5)$$

where $\eta_{lN} = (dX_j/dw)(1 - m)/X_j^o$ represents the compensated elasticity of demand for X_j with respect to the price of leisure, at point X_j^o.

It is important to note at this point that other products such as skis serve as complements to leisure. What is the impact of incorporating this alternative assumption with respect to leisure within the framework of the Parry model? In the framework, the higher price for the dirty commodity that results from the tax leads to an increase in leisure and a decrease in the amount that a consumer may purchase at a given nominal wage. If X_j and leisure are complements, the increase in the price of the dirty commodity would decrease the demand for leisure. In Figure 7, the demand for leisure schedule would shift left. This decrease in leisure time would increase real income and the amount that a consumer could purchase at a given wage. As a result, the tax-interaction effect would have the opposite impact on welfare than occurs in the Parry model. In other words, the TI effect would contribute positively to welfare.

(3) Revenue-recycling effect. Welfare gains result from the revenues of the environmental tax. This amount equals V multiplied by area D from Figure 7:

$$Vt_o X_j(t). \qquad (6)$$

Given the equations for the three effects, Parry goes on to show that, under plausible circumstances, the value of the TI effect exceeds the RR effect. He excludes from the analysis, however, the impact of a cleaner environment that results from a smaller quantity of X_j and lower damages. But to determine the overall welfare impact of the output tax, it is necessary to consider all three effects. The welfare effect is positive if

$$(1) + (6) > (5).$$

With equations (6) and (5), the revenue-recycling and tax-interaction effects, respectively, Parry (1995, S-70) explains that, under plausible circumstances, the latter effect exceeds the former. (The necessary condition is that the dirty commodity, X_j, is an average substitute for leisure at the pretax price.) As a result of this conclusion, the environmental impact must exceed the difference between the RR and TI effects for an overall positive welfare impact.

Environmental Input Tax

Parry extends the presentation of the welfare impact of output levies to evaluate environmental taxes on inputs. Assume now that the use of a particular input (Z) in the production of X_j pollutes the environment. We could think of Z as a chemical input. The input has constant marginal production costs equal to one. In this second case, the government levies a tax, t_i, on Z. Damage remains a function of emissions, but the governing authority decides to tax inputs in the production process.

(1) Environmental impact. The environmental improvement in the Z market occurs as a result of the increase in the factor price of the dirty chemical input, as producers use less. The environmental gain equals

$$t_i Z_o \eta_{zz} ce, \qquad (7)$$

where $\eta_{zz} = -(dZ/dp_z)/Z_o$ is the own-price elasticity of demand for Z, at point Z_o. Output and substitution effects comprise the elasticity. The former effect entails the reduction in X_j, while the latter means the use of different inputs in the production process.

(2) Tax-interaction effect. Since it is similar to the TI effect with the output tax, this effect is given (without derivation):

$$(1 + V) \frac{m}{1 - m \Delta t_i} \Delta p_i X_j^o \eta_{jN}. \qquad (8)$$

(3) Revenue-recycling effect. In the case of the input tax, the RR effect is

$$V t_i Z(t_i). \qquad (9)$$

Thus, a welfare gain requires that

$$(7) + (9) > (8).$$

As with the output tax, the literature concludes that the TI effect will exceed the RR effect given the current assumptions.

Fully Informed Emissions Tax

To extend the analysis of Parry, chemical emissions may be thought of as another input in the production process along with capital, labor, energy, and materials (Goulder, 1995a, 167). Firms under consideration emit a certain level of chemical emissions in the production of output. With new environmental tax policy, assume that the emissions levy raises production costs and the price of output. But with this input, a problem occurs formulating the emissions price, p_e. Firms purchase steel and energy, for instance, according to market prices. But this procedure does not occur with emissions. In the context of this problem, the emissions price may be formulated as a

function of input switching costs. A higher price for a particular chemical emission, for example, will encourage firms to substitute other, less-expensive emissions if it is possible. The substitution cost is the "price" of emissions, or the marginal cost of abatement.

(1) Environmental impact. An emissions tax links to the source of pollution directly. Using quantity of chemical emissions as the base, the fully informed levy (t_e) is a per-unit tax on individual effluent. The tax rate reflects the economic value of marginal damage at the annual emissions quantity. Firms use this chemical input e in the production of X_j. Assume that marginal abatement costs are linear over the relevant range. In the context of the Parry framework, an improvement in environmental quality results from a lower level of emissions:

$$t_e e_o \eta_{ee} ce, \tag{10}$$

where $\eta_{ee} = -(de/dp_e)/e_o$ is the change in emissions from a change in price, evaluated at point e_o. The elasticity variable shows a decrease in e. It includes an output effect with the reduction of X_j and substitution effect as firms use other chemical inputs to replace or lower the level of e.

(2) Tax-interaction effect. Welfare goes down with the TI effect when leisure and X_j are substitutes. The value for TI (which includes the lost revenues from the labor tax) equals

$$(1 + V)\, m \frac{\Delta N}{\Delta t_e}, \tag{11}$$

where the last term is the positive change in leisure that results from the higher cost for the substitute good X_j. As before, we may write the last term

$$\frac{\Delta N}{\Delta t_e} = \frac{dN}{dp_j} \frac{\Delta p_j}{\Delta t_e}, \tag{12}$$

where $\Delta p_j/\Delta t_e$ represents the increase in the price of X_j (marginal cost) that results from the implementation of t_e. Using the same procedure and multiplying by $(1 - m)X_j^o/(1 - m)X_j^o$, leaves

$$(1 + V) \frac{m}{1-m} \frac{\Delta p_j}{\Delta t_e} X_j^o \eta_{jN}. \tag{13}$$

(3) Revenue-recycling effect. The RR effect leads to a welfare gain of

$$V\, t_e\, e(t_e). \qquad (14)$$

For an overall positive impact,

$$(10) + (14) > (13).$$

Risk-Based Tax

The risk-based tax serves as a proxy for a fully informed emissions levy. The latter policy sets a tax rate based on known marginal damages. The former regulation implements a rate that reflects approximate marginal damage. Recall the revenue function for a risk-based levy:

$$G_R = t_{ru}\,(RU)(e),$$

where G_R = government revenue, t_{ru} = tax on a risk unit, and RU = number of risk units. The number of RU is multiplied by the emissions tax base.

(1) Environmental impact. Since a policy maker will calculate the number of risk units of an emission quantity, the firm will take this into account. Risk units and t_{ru} are variables the firm will consider when making production decisions. The environmental gain reflects the reduction in chemical emissions:

$$t_{ru}\,(RU)(e_o)\;\eta_{ruee}\,ce, \qquad (15)$$

where $\eta_{ruee} = -(d(RU)(e)/dp_e)/e_o$ represents the change in emissions as a result of a change in price, evaluated at point e_o.

(2) Tax-interaction effect. The derivation of this equation patterns the previous subsection. The only difference between equation (13) and the following equation is the appearance of a different rate t_{ru} in (16), instead of the tax rate t_e. The effect is

$$(1 + V)\,\frac{m}{1-m}\,\frac{\Delta p_j}{\Delta t_{ru}}\,X_j^o\,\eta_{jN}. \qquad (16)$$

(3) Revenue-recycling effect. The derivation of the revenue-recycling effect for the environmental risk unit tax patterns the derivations above. The final RR equation is

$$V t_{ru} [RU e(t_{ru})]. \qquad (17)$$

For an overall welfare gain to occur with risk-based taxation,

$$(15) + (17) > (16).$$

Problems with the Parry Framework and Justifications of Its Use

The Parry model incorporates a key assumption from another influential double dividend paper: Bovenberg and de Mooij (1994). The assumption is that leisure and the dirty commodity are substitutes. In the extension of the Parry framework in the above subsection, this assumption is carried over to the emissions-based taxes. In effect, the assumption guarantees that the demand for leisure will increase as a result of the implementation of an environmental tax. This result in turn means a decrease in real income and a welfare loss. As a result of narrowness in approach, Parry does not include all possible associations between the tax base and leisure.

How does Parry respond to the use of the substitute assumption with respect to leisure and the commodity? He says that "It is possible that X_j and leisure are complements...However this is only applicable to a narrow range of goods...and will be ignored" (Parry, 1995, S-68). Despite the choice by Parry to eliminate this possibility from the analysis, certain products could serve as complements to leisure, including speed boats, video cassette recorders, skis, and chlorine for swimming pools. Household consumption processes take time, so leisure is a complement in many cases. In sum, the assumption incorporated into the Parry framework impacts the final result by making the TI effect reduce welfare. The analysis in the following subsection considers the effect of using an alternative assumption of complementary.

In addition, the Parry model in its original form assumes constant marginal damage. The above adaptation shows, however, that it is possible to include increasing and non-linear damages within the calculation of marginal social cost. The Parry framework, therefore, is not so rigid that it cannot be altered in a more realistic manner. The different measures of marginal damage provide an accurate depiction of the relationship between the change in emissions and the change in marginal damage.

Since the externality problem includes a number of chemical emissions, moreover, a concern in the application of the above efficiency analysis entails the potential for nonconvexities. In chapter eight of their book, Baumol and Oates (1988) show that if a detrimental externality is sufficiently large, the social production set must be non-convex. This result is likely to happen in the presence of multiple emissions. The non-convex social production set causes a situation where a unique solution—in terms of the optimal level of emissions—does not exist. A policy maker may not rely on prices to provide the right signal or to show whether a solution is a maximum or minimum, local or global. The nonconvexity problem is a valid concern. However, the issue of nonconvexities addresses the problem of knowing if an obtained solution is a maximum and makes unclear the welfare improvement of moving to an optimal solution. But the analysis above and below does not find the optimal solution for the risk-based tax or the other levies. The risk-based system, in particular, is operating at a non-optimal solution, assigning rates to pollutants based on approximate marginal damages. Not in an optimal framework, the efficiency analysis attempts to find plausible numerical values for the TI, RR and environmental impacts, which are the relevant effects identified in the literature. It must be the case that the implementation of each levy leads to some value for each of these effects. The Parry framework, while flawed and subsequently adjusted to consider the potential for increasing and non-linear marginal damage, provides equations to calculate the three effects. Given the fact that the presence of nonconvexities clouds

the results of whether each tax leads to welfare improvements with the proposed values for the relevant variables, any framework would encounter the same concern as long as multiple emissions are involved. Baumol and Oates (1988), incidentally, conclude that non-optimal environmental taxes can still be appropriate in the presence of nonconvexities (chapter eleven).

Given its flaws, how may the Parry model be justified as a method to contribute to the analysis? The model—with its adaptations above—establishes a framework in which environmental taxes have been compared. Specifically, the identification of the tax-interaction and revenue-recycling effects in the literature have cast doubt on the potential for welfare contributions from environmental taxation. The Parry framework allows the calculation of the relevant impacts (including environmental) and shows the importance of incorporating particular assumptions in the analysis. The point is that a new environmental tax policy will lead to efficiency costs. The extent that these costs may offset environmental benefits in different circumstances is the subject of the next subsection.

Efficiency Effects

To summarize, the following equations show the conditions necessary for an overall welfare gain to occur. On the left-hand side of each equation, the environmental effect is added to the revenue-recycling effect. The right-hand side shows the tax-interaction effect.

Output tax: $t_o X_j^o \eta_{jj}\, ce + V t_o X_j(t) > (1 + V) \frac{m}{1 - m} t_o X_j^o \eta_{jN}$

Input tax: $t_i Z_o \eta_{zz}\, ce + V t_i Z(t_i) > (1 + V) \frac{m}{1 - m} \frac{\Delta p_j}{\Delta t_i} X_j^o \eta_{jN}$

Emissions tax: $t_e e_o \eta_{ee}\, ce + V t_e e(t_e) > (1 + V) \frac{m}{1-m} \frac{\Delta p_j}{\Delta t_e} X_j^o \eta_{jN}$

Risk Unit tax: $t_{ru} (RU)(e_o) \eta_{ruee}\, ce + V t_{ru} [RU\, e(t_{ru})] > (1 + V) \frac{m}{1-m} \frac{\Delta p_j}{\Delta t_{ru}} X_j^o \eta_{jN}$

If the TI effect exceeds the RR effect as in the Parry (1995) and Bovenberg and de Mooij (1994) models, the environmental impact must be greater than this difference for a welfare gain to occur.

With the risk-based tax, the environmental impact (first) term shows that the RU weights the emissions base by a hazard value. While the Parry model and extensions are conducive for the analysis of one pollutant, risk-based taxation may regulate a number of chemical emissions. Specifically, the risk-based levy encourages individual firms to alter their emissions mix by taking into account the damage-weighting method.

The analysis is geared to estimating the change in activity that results from the implementation of an environmental tax policy. The reduction in emissions from the regulation will depend on responsiveness. If firms are responsive, the tax will reduce the target pollutant.[75] In the following experiment, the before-tax emissions level may or may not exceed the after-tax quantity for each tax base. Different assumptions are made concerning responsiveness for the purpose of observation. Along the lines of the literature, TI and RR effects are also calculated in order to speculate on an overall efficiency effect.

The relevant impacts must be compared to some baseline or reference environment in the absence of new regulation. The calculation of relevant values before the implementation of the tax policy calibrates the model and allows a comparison with after-tax results. Suppose the policy targets SIC 2812, Alkalines and Chlorine, in the Chemicals and Allied Products Industry Group. The regulated chemical is chlorine dioxide with a RU value of 1.05 (Appendix 4). In 1994 industry 2812 emitted 92,358 pounds of chlorine dioxide (TRI). The value of shipments from this industry was $2,728,000 in 1994. For the sake of continuity, the before-tax levels of emissions and output are normalized to equal one. If they change after tax implementation, output and emissions are assumed to vary by the same percentage. The environmental effect depends on the

[75] If efficiency gains are possible with the taxation of one pollutant, generalizations may be made.

magnitude of the change in emissions. A greater reduction will lead to a larger effect. In the absence of the environmental tax, the revenue-recycling and tax-interaction effects are non-existent and assume zero values. The results from tax implementation are therefore compared to a reference environment equal to zero for these two impacts. In the simulation experiment, the elasticity values come from the literature or are varied to resemble different assumptions.

What are the plausible values of the parameters in the Parry framework that will make the environmental effect exceed the difference between the tax-interaction and revenue-recycling effects and create an overall welfare gain? The following analysis addresses this question in the context of the risk-based tax. The revenue simulation implements rates that vary between $0.025 and $0.50 per RU. As a result, the simulation employs the rates $0.05 and $0.50. In the analysis that follows, *ce*, the value of marginal damage, will either exceed *t* or equal *t*. When *ce* exceeds the tax, it is assumed that the tax deviates from true marginal damages. When $t < ce$, the difference between the two is assumed to be ten percent.

Consider the other variables. With *V*, Parry incorporates the median, lower and upper bounds in his empirical application. The experiment below will do the same. The elasticities take on different values to represent alternative possibilities. The value for *m*, the labor tax wedge, also comes from Parry. The increase in the tax raises the marginal cost of output by the same amount. As Parry (1995, S-70) explains, the elasticity of demand with respect to leisure for particular forms of output are generally not available. As a result, this analysis assumes alternative values to take into account different degrees of substitutability between the output and leisure. Represented as η_{LN}, positive values mean the case when the dirty commodity and leisure possess different degrees of substitutability, while a negative η_{LN} means that output and leisure are complements. For an overall welfare gain, the environmental effect must exceed the difference between the TI and RR effects. Table 23 summarizes the relevant variables and their plausible values from the welfare

equations. Table 24 presents the calculations for the tax-interaction effect using the plausible numbers: $(1 + V)\ (m/1\text{-}m)(\Delta p_j/\Delta t)\ X_j^o\ \eta_{jN}$.

The value of the TI effect depends on η_{jN}, the compensated elasticity of demand for X_j with respect to the price of leisure. As the degree of substitution between leisure and the output increases, the value of the TI effect increases. It is important to note, however, that the TI effect is assumed to reduce welfare. In other words, the TI effect reduces overall welfare. But if the TI effect is negative, which occurs when η_{jN} assumes a value of -1.0, the TI impact contributes positively to welfare. In this latter situation, the higher tax reduces the after-tax wage, which in turn lowers the price of leisure. The demand for leisure decreases when the demand for the dirty commodity goes down when the two are complements. Individuals work more and real income goes up. In the Parry model, the TI effect contributes positively to welfare when the demand for leisure decreases.

To demonstrate the revenue-recycling effect with the risk-based levy, assume that the target emission is chlorine dioxide, with a RU value of 1.05. The risk-based system alters the tax base by a value for RU to account for the fact that certain chemical emissions have higher damage values.[76] Chlorine dioxide is an example. With the risk-based tax, the revenue-recycling equation is $V\ t_{ru}\ [RU\ e(t_{ru})]$. Table 25 presents the calculations for the RR effect, assuming plausible parameter values. The $RU\ e(t_{ru})$ column in Table 25 multiplies the emission quantity by 1.05.

[76] A "typical" risk-based tax system will include chemical emissions with RU values greater than or less than one.

Table 23
Relevant variables and plausible values for the efficiency equations

Variable	Description	Values	Variable	Description	Values
t_{ru}	risk tax levied on one risk unit	\$0.05, \$0.50	V	marginal welfare cost of labor tax revenues	0.14, 0.29, 0.59
e_o	emissions quantity in the absence of a tax	1	η_{ruee}	compensated own-price elasticity of demand for e (risk unit tax)	0.5 inelastic 1.0 unit elastic 1.5 elastic
$e(t_{ru})$	emissions quantity in the presence of the risk-based tax	function of the tax	Δp	change in price	same as Δt
RU	risk units	1.05	Δt	change in tax	same as Δp
c	value of the externality	either equals the tax or exceeds the tax	η_{jN}	compensated elasticity of demand for X_j with respect to the price of leisure	-1.0, 0.5, 1.0, 1.5
X_j^o	level of pre-tax output	1	m	wedge from leisure subsidy	0.43

Table 24
Tax-interaction effect

V	$m/1-m$: $m = 0.43$	X_j^o	η_{jN}	TI effect
0.14	0.75	1.00	-1.0	-0.855
			0.5	0.4275
			1.0	0.855
			1.5	1.2825
0.29	0.75	1.00	-1.0	-0.9675
			0.5	0.48375
			1.0	0.9675
			1.5	1.45125
0.59	0.75	1.00	-1.0	-1.1925
			0.5	0.59625
			1.0	1.1925
			1.5	1.78875

Table 25 incorporates the fact that the new level of emissions, $e(t_{ru})$, is a function of the tax rate. Depending on how the firm reacts to the tax, emissions could decrease. If a tax increase of $0.05 represents a five percent cost increase, emissions may fall 2.5 percent, five percent, or ten percent, depending on elasticity values. The revenue-recycling effect represents the value of tax revenue. With a high rate and large base, this impact will be high; however, as the rate increases, emissions fall, as firms try to avoid tax payment. Compared to Table 24, the RR values in Table 25 are lower than the TI results with the *substitution assumption* between leisure and the commodity. This result conforms to Bovenberg and de Mooij (1994) and Parry (1995).

Table 26 presents the environmental effect with different parameter values. The original emissions level, e_o, is multiplied by RU: t_{ru} $(RU)(e_o)$ η_{ruee} ce. Assume the same chemical emission as before, chlorine dioxide, with a RU value of 1.05. For the calculation of ce with each value of η_{ee}, ce is first assumed to exceed the tax rate by ten percent and then to equal the tax.

Table 25
Revenue-recycling effect

V	t_{ru}	$RU\ e(t_{ru})$	RR effect
0.14	$0.05	1.023	0.007161
		0.9975	0.006982
		0.945	0.006615
	$0.50	0.7875	0.055125
		0.525	0.03675
		0	0
0.29	$0.05	1.023	0.014833
		0.9975	0.014463
		0.945	0.013702
	$0.50	0.7875	0.07875
		0.525	0.076125
		0	0
0.59	$0.05	1.023	0.030178
		0.9975	0.029426
		0.945	0.027877
	$0.50	0.7875	0.23231
		0.525	0.15487
		0	0

Table 26
Environmental effect

t_{ru}	$RU\ e_o$	η_{ruee}	ce	Environmental effect
0.05	1.05	0.5	0.055	0.00144
			0.05	0.00131
		1.0	0.055	0.00288
			0.05	0.00262
		1.5	0.055	0.00433
			0.05	0.00393
0.50	1.05	0.5	0.55	0.14437
			0.5	0.13125
		1.0	0.55	0.28875
			0.5	0.2625
		1.5	0.55	0.43312
			0.5	0.39375

Using alternative values for the relevant parameters, the overall efficiency impacts of the risk-based tax are in Table 27. To narrow the analysis, Table 27 assumes the midpoint value for V, 0.29. In addition, η_{ruee} is assumed to equal one. In this case the tax rates of 0.05 and 0.5 are levied on one risk unit of chlorine dioxide emissions. Recall the relevant efficiency equation:

$$t_{ru}\,(RU)(e_o)\,\eta_{ruee}\,ce + V\,t_{ru}\,[RU'\,e(t_{ru})] > (1+V)\,\frac{m}{1-m}\,\frac{\Delta p_j}{\Delta t_{ru}}\,X_j^o\,\eta_{jN}\,.$$

This equation shows that a welfare gain occurs when the environmental (first term) and revenue-recycling (second term) effects exceed the tax-interaction effect (third term). Table 27 contrasts the environmental effect with the difference between the TI and RR effects:

$$t_{ru}\,(RU)(e_o)\,\eta_{ruee}\,ce > (1+V)\,\frac{m}{1-m}\,\frac{\Delta p_j}{\Delta t_{ru}}\,X_j^o\,\eta_{jN} - V\,t_{ru}\,[RU'\,e(t_{ru})].$$

When the TI effect is negative in Table 27, it represents an efficiency gain. In this situation, the positive RR effect is added to the TI effect. In all other cases, the important result entails the difference between the greater TI effect and the smaller RR effect.

Table 27
Overall efficiency impact

V	η_{jN}	t_{ru}	$RU e(t_{ru})$	η_{ruee}	ce	EE (1)	TI - RR (2)	(1) - (2)	EE>(TI -RR)?	Efficiency gain/loss
.29	-1.0	0.05	1.023	1.0	0.055	0.00288	-0.9823	0.98518	yes	gain
					0.05	0.00262	-0.9823	0.98492	yes	gain
			0.9975	1.0	0.055	0.00288	-0.9819	0.98478	yes	gain
					0.05	0.00262	-0.9819	0.98452	yes	gain
			0.945	1.0	0.055	0.00288	-0.9812	0.98408	yes	gain
					0.05	0.00262	-0.9812	0.98382	yes	gain
		0.50	0.7875	1.0	0.55	0.28875	-1.0462	1.33495	yes	gain
					0.5	0.2625	-1.0462	1.3087	yes	gain
			0.525	1.0	0.55	0.28875	-1.0436	1.33235	yes	gain
					0.5	0.2625	-1.0436	1.3061	yes	gain
			0	1.0	0.55	0.28875	-0.9675	1.25625	yes	gain
					0.5	0.2625	-0.9675	1.23	yes	gain
	0.5	0.05	1.023	1.0	0.055	0.00288	0.46891	-0.4660	no	loss
					0.05	0.00262	0.46891	-0.4662	no	loss
			0.9975	1.0	0.055	0.00288	0.46928	-0.0466	no	loss
					0.05	0.00262	0.46928	-0.4666	no	loss
			0.945	1.0	0.055	0.00288	0.47004	-0.4671	no	loss
					0.05	0.00262	0.47004	-0.4674	no	loss
		0.5	0.7875	1.0	0.55	0.28875	0.405	-0.1162	no	loss
					0.5	0.2625	0.405	-0.1425	no	loss
			0.525	1.0	0.55	0.28875	0.40762	-0.1188	no	loss
					0.5	0.2625	0.40762	-0.1451	no	loss
			0	1.0	0.55	0.28875	0.48375	-0.195	no	loss
					0.5	0.2625	0.48375	-0.2212	no	loss
	1.0	0.05	1.023	1.0	0.055	0.00288	0.95266	-0.9497	no	loss
					0.05	0.00262	0.95266	-0.9500	no	loss
			0.9975	1.0	0.055	0.00288	0.95303	-0.9501	no	loss
					0.05	0.00262	0.95303	-0.9504	no	loss
			0.945	1.0	0.055	0.00288	0.95379	-0.9509	no	loss
					0.05	0.00262	0.95379	-0.9511	no	loss
		0.5	0.7875	1.0	0.55	0.28875	0.88875	-0.6	no	loss
					0.5	0.2625	0.88875	-0.6262	no	loss
			0.525	1.0	0.55	0.28875	0.89137	-0.6026	no	loss
					0.5	0.2625	0.89137	-0.6288	no	loss
			0	1.0	0.55	0.28875	0.9675	-0.6787	no	loss
					0.5	0.2625	0.9675	-0.705	no	loss
	1.5	0.05	1.023	1.0	0.055	0.00288	1.43641	-1.4335	no	loss
					0.05	0.00262	1.43641	-1.4337	no	loss
			0.9975	1.0	0.055	0.00288	1.43678	-1.4339	no	loss
					0.05	0.00262	1.43678	-1.4341	no	loss
			0.945	1.0	0.055	0.00288	1.43754	-1.4346	no	loss
					0.05	0.00262	1.43754	-1.4349	no	loss
		0.5	0.7875	1.0	0.55	0.28875	1.3725	-1.0837	no	loss
					0.5	0.2625	1.3725	-1.11	no	loss
			0.525	1.0	0.55	0.28875	1.37512	-1.0863	no	loss
					0.5	0.2625	1.37512	-1.1126	no	loss
			0	1.0	0.55	0.28875	1.45125	-1.1625	no	loss

What observations result from Table 27? First and foremost, whether the environmental effect exceeds the difference between the TI and RR effects depends crucially on η_{jN}, the compensated elasticity of demand for X_j with respect to the price of leisure. When the dirty commodity and leisure are complements (and η_{jN} is less than zero), efficiency gains occur. Second, the bigger tax rate, 0.5, leads to higher efficiency gains when η_{jN} is negative or lower welfare losses when η_{jN} is positive, compared to the lower tax rate 0.05. This latter difference in the reflects the fact that, with welfare losses, the higher tax rate leads to a larger environmental impact and a smaller difference between the TI and RR effects. Third, the negative values for the difference between the TI and RR effects actually represent a positive welfare gain. (The negative TI effect means that leisure decreases when the environmental tax is levied. This effect contributes a positive efficiency value in the Parry framework.) As a result, when efficiency gains occur in Table 27 with a negative value for η_{jN}, all three efficiency effects contribute positively to welfare. However, under no circumstances are welfare gains positive using the substitute assumption between the commodity and leisure.

Policy Implications

Two policy implications result. First, if regulators are concerned with efficiency, they should target commodities that are complements or weak substitutes with leisure. Second, when it raises the tax rate to reflect environmental damage, a risk-based tax policy creates higher efficiency gains when η_{jN} is negative and lower efficiency losses when η_{jN} is positive when compared to a tax system that does not distinguish between the damage values of different pollutants.

Relation to Evaluation Criteria

The potential for tax-system efficiency losses to offset environmental quality reinforces the need to refer to the evaluation criteria and portfolio problem. In addition to tax-system efficiency losses, the risk-based levy will raise administrative (collection and enforcement) costs and compliance costs (evasion, avoidance and monitoring). In many situations the regulator will have to look to other benefits of the policy to justify regulation with risk-based taxation. Fortunately, the risk-based system presents the opportunity for higher levels of environmental quality. As it relates the risk measure to the damage of specific pollutants, the risk-based system attempts to internalize the pollution externality. Since it targets emissions, moreover, risk-based taxation improves environmental quality for everyone. In the *short term*, revenue payment will increase as more chemical emissions are included within the policy framework. In addition, the regulation of a large group of pollutants will decrease the potential for revenue instability during the different phases of the business cycle. However, the higher prices for more harmful emissions will continually encourage firms to decrease the use of these pollutants. In the *long term*, therefore, the regulation will tax away the portion of the base comprising the most harmful pollutants, or, at the very least, establish a pollution price for these chemical emissions.

TAXATION AND THE PRODUCTION PATHWAY

Suppose the Tennessee government wants to finance, with environmental taxation, a lower corporation net income tax on an industry group. The state targets this business tax in order to shift some tax burden from income to pollution. The target industry group, Chemicals and Allied Products—SIC Major Group 28—is chosen as a result of the group's large number of chemical

inputs, output, and emissions.[77] Table 28 shows the twenty-nine four-digit Standard Industry Classification (SIC) industries in Major Group 28.

Appendix 5 reveals the corporation net income tax (called an "excise" tax in Tennessee) raised 407.3 million dollars in 1994. This revenue figure applies since the emissions data also come from 1994. The tax rate is 6% on corporate net income in the state. The corporation net income tax raised 7% of total tax revenues for Tennessee in 1994 ($5314.1 million).

To implement revenue substitution, the governing authority would need the tax payment for the target industry group as a percentage for all industry groups; however, the Tennessee government will not reveal this number. It does not want to provide profit totals. Suppose Major Industry Group 28 contributes 10% of the total net income tax revenue for the state.[78] The group pays $40,730,000 (= $407,300,000 * 0.10) in net income tax revenue.

The tax base, or profits, equals tax revenue divided by tax rate (6%): $40,730,000/0.06 = $678,833,333.3. If the governing authority wants to lower the net income tax from 6% to 4% for the target industry group, tax revenues would equal $27,153,333 (=$678,833,333.3*0.04). The difference in tax revenue between the 6% policy and the 4% policy is $40,730,000 - $27,153,333 = $13,576,667. This latter number represents the amount of revenue the new environmental tax must raise for revenue substitution.

[77] The choice of one industry group for environmental regulation places the group at a competitive disadvantage relative to unregulated industries. This purpose of this presentation is to establish the framework for analysis of Parry's three effects, but not to consider the relative burden of target industries.
[78] This figure is probably an over-estimation.

Table 28
Standard Industry Classifications for Industry Group 28: Chemicals and Allied Products

SIC	Name	SIC	Name
2812	Alkalines and Chlorine	2844	Perfumes, Cosmetics, and other Toilet Preparations
2813	Industrial Gases	2851	Paints, Varnishes, Lacquers, Enamels, and Allied Products
2816	Inorganic Pigments	2861	Gum and Wood Chemicals
2819	Industrial Inorganic Chemicals, Not Elsewhere Classified	2865	Cyclic Organic Crudes and Intermediates, and Organic Dyes and Pigments
2821	Plastics Materials, Synthetic Resins, and Nonvulcanizable Elastomers	2869	Industrial Organic Chemicals, Not Elsewhere Classified
2822	Synthetic Rubber	2873	Nitrogenous Fertilizers
2823	Cellulosic Humanmade Fibers	2874	Phosphatic Fertilizers
2824	Humanmade Organic Fibers, Except Cellulosic	2875	Fertilizers, Mixing Only
2833	Medicinal Chemicals and Botanical Products	2879	Pesticides and Agricultural Chemicals, Not Elsewhere Classified
2834	Pharmaceutical Preparations	2891	Adhesives and Sealants
2835	In Vitro and In Vivo Diagnostic Substances	2892	Explosives
2836	Biological Products, Except Diagnostic Substances	2893	Printing Ink
2841	Soap and Other Detergents, Except Specialty Cleaners	2895	Carbon Black
2842	Specialty Cleaning, Polishing, and Sanitation Preparations	2899	Chemicals and Chemical Preparations, Not Elsewhere Classified
2843	Surface Active Agents, Finishing Agents, Sulfonated Oils, and Assistants		

Source: Office of Management and Budget. 1987. *Standard Industrial Classification Manual.* Washington: OMB.

To address the chemical pollution problem and raise money for revenue substitution, the governing authority could choose to tax different bases: inputs, output or emissions. The efficiency equations (from the Parry framework) demonstrate the impacts from the implementation of an environmental tax on each base along with a policy of revenue substitution. How would the overall efficiency costs and benefits differ if the regulator chooses to tax output, inputs, or emissions? The impact on environmental quality, in particular, would depend on how closely the tax base links to the source of pollution. If the levy provides an incentive for pollution reduction, a significant environmental impact may join the revenue-recycling effect to offset efficiency losses from the tax-interaction effect.

Suppose a policy maker had information concerning a target "pathway" in production for regulated industry.[79] A pathway in production means the technological relationship between inputs, output, and emissions. To produce a certain quantity of one product, firms employ a number of inputs (capital, labor, materials, energy, chemicals, etc.) and a level of technology. Emissions result from the production process. In equation form, output X is a function of inputs $Z_1, \ldots, Z_n$, where Z represents all inputs. Technology establishes a numerical relationship between inputs and output, which can be represented as:

$$X = q(Z_1, \ldots, Z_n).$$

Technology also establishes a relationship between emissions (E) and the production of output:

$$E = e(X).$$

The pathway, then, may be represented by substituting the first equation into the second:

[79] It is important to note here that information limitations restrict a full comparison of Parry's three effects when applying an environmental tax on different tax bases. A comparison between tax bases would entail a method to hold one of the three effects constant (such as the revenue-recycling effect with the same level of revenue generated for each base) for an environmental tax policy on each base and then compare the tax interaction and environmental impacts between policies. This is the approach in the subsection. The presentation in this subsection establishes a blueprint for analysis, but does not come up with a conclusion concerning the relative welfare impact of a risk-based tax.

$$E = \theta[X = q(Z_1, \ldots, Z_n)],$$

where the coefficient θ represents the technological capabilities of the firm. The equation shows that emissions result from the production process. In addition, it demonstrates the relationship between three potential environmental tax bases: emissions, output, and inputs. The following discussion first addresses the potential to calculate the welfare impact of an environmental tax on inputs, output or emissions. Second, the relationships between (1) inputs and output, and (2) emissions and production are discussed. Third, an analysis of the relationship between emissions and damage (and how it fits into the above pathway framework) follows.

In theory, an environmental tax policy could raise the same amount of revenue by taxing output, inputs or emissions.[80] With each base, the tax rate would vary in order to raise a target level of revenue. From the point of view of the firm, the choice of tax base would impact production decisions. The firm would not react the same way to a risk-based tax as opposed to an input tax. In addition, consumers would react differently to alternative levies. A product tax may impact behavior more directly than an emissions charge. As a result of these two factors, each tax would lead to a unique effect on environmental quality.

Assume, then, that it would be possible to implement an environmental tax on dirty inputs, output or emissions for target industry. In addition, the tax rate could be set to raise a certain level of revenue. In a policy of revenue substitution, the desired level of revenue from the environmental tax would correspond to the loss in revenue that would occur with a lower rate on a business or

[80] Depending on the target industries and production methods, certain tax bases would be more appealing from an environmental and/or administrative standpoint. Businesses in a target industry, for instance, may employ few dirty inputs. In this case, an environmental input tax may not be able to raise a desired level of revenue. In contrast, a target industry may emit a number of chemicals. With this latter case, the regulator could implement a risk-based tax.

other tax.[81] In the context of the Parry (1995) framework, the environmental tax on each base would create the same revenue-recycling effect by raising the same level of revenue.[82] What, then, would be the overall welfare impact (using the Parry framework) of individual environmental taxes on inputs, output or emissions? Keeping the revenue-recycling effect constant between tax bases, the overall welfare impact would depend on the tax-interaction and environmental effects.[83]

To analyze the pathway, three assumptions are necessary with respect to production technology for a "representative" firm.[84] The first assumption is that the firm possesses a linear production function. With this form of technology, output is produced in fixed proportions by the use of a certain number of inputs. In this case, the production function is homogeneous of degree one. As a result, it has constant returns to scale (Henderson and Quandt, 1980, 120-121).

Using similar notation from above, linear production may be characterized with a single output x produced from z inputs. The production process entails a set of coefficients, $a_1, \ldots, a_m$, that convey the input quantities necessary to produce final output. For any level of output, the quantity of each input is uniquely determined:

$$z_i = a_i x, \qquad i=1,\ldots, m.$$

[81] This could be a tax on capital or profits (for the firm), income (for the individual), or some other base. A different tax to target for revenue substitution besides the labor income tax would change the individual's budget constraint in the Parry framework, but would not alter the idea behind revenue substitution.

[82] For this to hold, V, the efficiency value per dollar of changes in tax revenues, must be constant. If pollution tax revenues are assumed to be small compared to the revenues of the tax source that is being lowered (for revenue substitution), changes in m, the wedge between the supply and demand prices of leisure, and leisure are small. In this case, we can assume V as constant (Parry, 1995, S-69). A revenue substitution policy that targets something other than the labor tax would change the formulation of V. This subsection assumes a constant V since the environmental tax revenue raised would be a small percentage of the revenue from the existing tax.

[83] We have an aggregation concern here. The Parry (1995) framework assumes an environmental tax on one output or input. The assumption used in the following presentation is that the three effects in Parry can be translated to a policy system that includes more than one taxable item, i.e. a number of different emissions.

[84] The firm is assumed to operate in a competitive market.

To produce one unit of output x, for example, the firm must employ a_i units of input. The maximum amount of output that the firm may produce from a set of inputs is

$$x = \min_i (z_i/a_i), \quad a_i > 0.$$

In other words, each input limits output. The quantity z_i, for example, will support an output level of z_i/a_i units; however, every other input must be available in sufficient quantity to produce this level of output. A certain amount of inputs may remain available after production as a result of scarcity of one input z_i. Therefore, the smallest ratio z_i/a_i determines the highest possible quantity of output (Henderson and Quandt, 1988, 121).

The second assumption concerns the relationship between output and emissions (e). The production of output leads to a certain level of emissions. This relationship may be characterized by a linear equation:

$$e_i = b_i x, \qquad i=1,\ldots, n,$$

where b_i represents one particular form of emissions (such as formaldehyde or chlorine dioxide.)

The third assumption involves the relationship between emissions and damage (d). The relationship between emissions and damage may be characterized by the following equation:

$$d = ce, \qquad c > 0, \quad e = \Sigma e_i.$$

As it is written, damage increases at a constant rate when emissions go up. A non-linear representation of damage could result from the analysis, but would not correspond with the risk-based framework. A non-linear damage function would pattern the analysis of a fully informed levy.

How restrictive are these assumptions concerning production linearity? The assumptions imply that production coefficients (for inputs in the production of output and emissions in the production of output) and damage coefficients (for emissions in the formulation of damage) are

fixed and technology is constant. Technology, however, may be characterized in different ways (Varian, 1992, Chapter One). The above equations do not capture the range of production possibilities that the firms in the target industries (discussed below) employ; therefore, the equations represent a simplifying tool to analyze the production pathway.

Given the linear relationships between inputs and output and output and emissions, tax rates may be calculated that raise the target level of revenue. To demonstrate this concept, consider the following input/output/emission pathway for the production of tinplate.[85] To produce 1000kg (one unit) of tinplate, the production process requires 3.9kg of zinc input. In addition, the production of this output leads to 0.01967kg of zinc emissions. In equation form,

$$z_1 = 3.9x,$$

where the zinc input = z_1 and tinplate output = x. Also,

$$e_1 = 0.01967x,$$

where zinc emissions = e_1. By levying an environmental tax according to weight, the policy maker could move between bases. For example, to raise \$10 in revenue with the tinplate/zinc pathway, a \$0.01 per kg tax rate on one unit of output would suffice: (1000kg/1 unit)(\$0.01kg) = \$10. In addition, a tax rate of \$2.56 per kg on the zinc input would raise \$10: (3.9kg)(\$2,56) = \$10. With emissions, a tax rate of \$508.38 per kg of emissions gives \$10: (0.01967)(\$508.38) = \$10. To tax risk units, the regulator needs the number of risk units per unit of emissions. If the emissions of different chemicals are measured per kg, the relative rankings for RU would be the same. From Appendix 4, the number of risk units for a unit of zinc emissions equals 0.52. If revenue = $(t_{ru})(RU)(e)$ for risk-based taxation, then the tax rate necessary to generate \$10 is \$977.67: \$10 = (\$977.67)(0.52)(0.01967).

[85] For additional information concerning the tinplate pathway and other pathways point to the EcoSite at www.ecosite.co.uk/depart/Data.html.

Table 29 summarizes of the four tax rates necessary to generate $10. A pathway with an alternative relationship between inputs, output, and emissions would yield different tax rates to achieve the target level of revenue. The tax rate on one RU is excessive compared to the others in Table 29. This rate results from the small quantity of zinc emissions resulting from the production pathway and the value for RU. This specific example is not intended for policy conclusions. It is meant to demonstrate the concept of the production pathway.

Table 29
Tax rates necessary to generate $10 for the tinplate pathway

Tax Base	Tax Rate
input	$2.56 per kg
output	$0.01 per kg
emissions	$508.38 per kg
risk unit	$977.67 per kg

The policy maker would need extensive pathway information for the twenty-nine different industries in Industry Group 28 to calculate tax rates on inputs, output and emissions to generate a target level of revenue with each base. Even with one industry, the regulator would need specific, firm-level pathway data. (The movement from one tax base to another is possible in limited cases, but would be difficult for an entire industry. Data restrictions prohibit a thorough treatment of the input/output pathway for each firm within Industry Group 28.) As a result of information limitations, the following discussion analyzes the risk-based system in conjunction with revenue substitution and alludes to other tax possibilities to demonstrate the production pathway.

Suppose a regulator targets the 158 chemical emissions in Tennessee from Davis (1994c). The regulator bans the ten most toxic, eliminates others that are severely restricted, and regulates the fifty remaining chemical emissions with the highest hazard values. The idea is to generate $13,576,667 in environmental tax revenue to use in the policy of revenue substitution. Data were

gathered for the fifty target chemicals. In the case of Industry Group 28, the risk-based (RT) environmental tax system would employ a tax rate of $0.8525575 on one risk unit to raise $13,576,667. Returning to the efficiency analysis, the welfare impact of this system would depend on the relative values of the environmental, tax-interaction and revenue-recycling effects.

What specific conditions would create an efficiency gain with risk-based taxation?

- With a target set of emissions, a sufficient rate would have to impact behavior at the margin; i.e., the tax would have to provide the incentive to alter production methods.
- By distinguishing between the hazard potential of target emissions and pricing the emissions accordingly, the risk-based system would have to provide the incentive to reduce the most harmful chemical emissions. For the sake of comparison, the risk-based system raises 10% of the revenue from the ten most harmful pollutants in the target set. A simple emissions tax system would raise 9% of the revenue from the same emissions.
- Over time, the risk-based policy would have to change the composition of emissions from the target industries to include less harmful pollutants. The change in emissions would increase environmental quality.
- The value of the environmental impact would have to exceed the difference between the TI and RR effects. The impact of the TI effect in particular depends on parameter assumptions, which may differ across target industries.

While this analysis does not conclude that the risk-based system would lead to efficiency gains, it establishes the conditions necessary for this result to occur.

ARE COSTS MINIMIZED WITH RISK-BASED TAXATION?

The regulator of the risk-based system would have information on damage values from the CCPCT index and quantities of each emission. It is impossible, however, to aggregate the individual damage functions (from each emission) into one total damage function for all emissions. As a result, the regulator cannot speculate on some optimal level of emissions for society in the absence of the economic value of total damages (and marginal abatement costs.)

The risk-based system encourages a change in the emissions mix of individual firms. This framework takes a practical policy approach. It addresses the following question: how could regulators encompass a set of chemical emissions with one policy? With the proposed framework, regulators would attempt to implement efficient policy by encouraging the reduction of the most harmful chemical emissions. Information limitations, however, would prevent the achievement of optimal emission reduction or even the identification of the optimal point.

This new incentive system approaches the optimality question from a different angle. Traditional environmental taxation analyses consider the marginal abatement costs and marginal benefits of the reduction of one pollutant. The optimal level of emissions reduction occurs where the two marginal functions intersect. But this point cannot be used as reference since the risk-based system includes a set of emissions. The risk-based tax system targets the benefits function for practical implementation reasons and the problem of estimating abatement costs.

As an incentive policy, the risk-based system allows firms rather than regulators to choose the best method for additional pollution control. Firms may treat the waste, change production methods, and take advantage of different abatement costs. Since businesses have flexibility in the abatement process with incentive policy, they choose the best method to incorporate the regulation into the production framework.

The risk system employs one tax rate to cover the set of chemical emissions. Facing one rate, firms with lower abatement costs decrease emissions more than firms with higher costs. In these circumstances, economics theory professes that total abatement costs are minimized when firms equalize marginal abatement costs in reaction to the tax. Even though marginal abatement cost functions are unknown to regulators, firms implement a least-cost solution. Given one tax rate on a risk unit, individual firms will adjust behavior to reflect its marginal abatement cost.

THE PREFERENCE FOR RISK-BASED TAXATION

The benefits of risk-based environmental taxation stem from the focus of the system. The framework attempts to target chemical emissions where net benefits exist for emission reduction. The regulator does not want to reduce the emissions of chemicals when marginal abatement cost exceeds any environmental benefits that would result from emissions reduction.

But, in the absence of information for abatement costs to cover regulated firms, policy makers must focus on the emissions that pose the highest risk to humans and the environment. This focus is the place for the chemical ranking methodology. The methodology allows a relative comparison of the damage potential of the target chemicals and the regulation of the most harmful emissions. With the evaluation of the damages of a number of pollutants, an aggregation problem prohibits the calculation of the optimal level of emission reduction, under jurisdiction in an nth-best world, for all of the chemical emissions. The risk-based system simplifies the problem by applying risk units to the target emissions.

CHAPTER 5

ISSUES OF POLICY IMPLEMENTATION AND CONCLUSION

Chapter five addresses an important topic concerning the implementation of risk-based taxation: policy jurisdiction. Regulation may assume national, state or local coverage, although the former two are probably more feasible. A problem with the administration of the risk-based tax system concerns pollution spillover. The pollution from target emissions will cross state lines and impose damage on humans and the environment. Nevertheless, the implementation sequence in chapter five proposes the adoption of the risk-based system at the state level. The problem of pollution spillovers across state lines will remain an issue for future research. In addition to developing a policy proposal, chapter five discusses the merits of risk-based taxation, contrasts this indirect policy instrument with a theoretical, direct method of regulation, evaluates the policy's costs and benefits and offers final comments.

THE MERITS OF RISK-BASED TAXATION

Risk-based taxation possesses many appealing characteristics. First, this policy offers a method to regulate a set of chemical emissions. With one tax base, the system encompasses the pollutants from the CCPCT index. The ranking methodology provides the basis for relative pricing. Formulating the risk-based framework, evaluating its merits, and calculating when the system serves as an effective policy constitute important contributions of this paper. With risk-based taxation, policy makers could use chemical rankings other than the CCPCT index, if the other systems prove suitable.

Second, to formulate relative risk values, the CCPCT algorithm uses toxicological endpoints for human and environmental impacts as well as exposure potential. This inclusive framework attempts to account for the pollution damage of each chemical emission by using different pathways. While limited in its formulation of site-specific pollutants, the risk-based methodology provides a relative ranking of the chemical emissions. This relative formulation allows the application of the risk unit.

Third, the risk-based system attempts to internalize pollution damage with an approximation of the damage function. Although an error term exists between the damage functions implied by the risk-based system and "true" damage functions, the system could provide welfare gains for society by encouraging the flow of cleaner chemical emissions. This tax system would impact environmental quality if the tax rate provides the incentive for pollution reduction. The extent that risk-based taxation affects related economic decisions[86] remains an issue for further research.

Fourth, like all environmental taxes, the levies in the risk-based system would raise revenue. This additional dividend of environmental taxation sets it apart from non-revenue raising instruments. Goulder, Parry, and Burtraw (1996) conclude that the implementation of environmental taxation in a revenue-neutral policy leads to the smallest tax-system losses when contrasted to other instruments that do not raise revenue. As chapter four explains, a state could raise a significant amount of revenue, depending on the tax rate applied to one risk unit.

Further research will identify the rate that will decrease the tax base for a particular industry. In addition, the research will calculate how increases in the rate affect the emissions mix of target firms. This future research will address the fifth—and very important—merit of the risk-

[86] These related economic decisions include the labor/leisure, intertemporal, intermediate input, and consumer good choices.

based tax system: the ability to enhance environmental quality by reducing the most harmful chemical emissions.

INDIRECT CONTRASTED WITH DIRECT RISK-BASED TAXATION

Chapter three explains that the risk-based tax methodology—with the CCPCT index—establishes an indirect framework for pollution control. The algorithm cannot account for specific levels of pollution damage in varying situations. Even though the algorithm includes a number of toxicological endpoints, including inhalation, oral, carcinogenic, and others, the framework does not entail site-specific estimation. Unpredictable meteorological conditions, the use of animals as surrogates for humans in the data set incorporated into the CCPCT index, an incomplete data set, a lack of site-specific estimates, different distances between pollution sources and human and animal populations, the inherent problems of using laboratory estimates for endpoints, and many other factors make the formulation of the relative hazard values an indirect calculation.

The Ideal World

In an ideal world, policy makers would have a direct method to ascertain the marginal damage from chemical emissions. Marginal damage from each pollutant would take into account specific meteorological and human conditions unique to each polluting facility and area under regulation. As the emissions of a particular chemical increased over time, an algorithm would show an equation relating damages to the quantity of emissions. The equation would have a coefficient to relate the two variables in a non-linear or linear manner:

$$\text{Damage} = f(\text{emissions}).$$

With a relationship between the two variables established, researchers could calculate the level of total damages by determining the area under the damage function at one level of emissions. This method—along with an economic valuation—would provide the damages associated with chemical emissions for each polluter under jurisdiction. As a direct method of regulation, policy makers could use total damage for a given period of time to levy a system of fully informed emissions taxes.

The reason that this direct method is unappealing from a policy perspective concerns the problem of gathering the necessary information. To calculate damage functions for a set of chemical emissions on a site-by-site basis, researchers would have to gather sufficient endpoint data unique to each location, including persistence and bioaccumulation. To assign economic values to the damages, the researchers would have to employ valuation techniques. If policy makers wanted to target a number of industries in a state, a team of researchers would require funding.

Improving the Algorithm

Improvements in the use of the algorithm involve repetition. Researchers could target a given industry and apply the algorithm to corresponding firms in different locations. This way, the researchers would understand the relationship between emissions and damage for a set of firms that employ similar production processes. The research would provide representative damage functions using the endpoints from the CCPCT algorithm. Repetitive use with familiar targets would familiarize researchers with the strengths and weaknesses of the algorithm and data-collecting techniques.

In addition, once researchers target a specific industry, they would improve their understanding by recalculating certain toxicological endpoints at different periods of time. For instance, the CCPCT index includes impact pathways for persistence and bioaccumulation in the calculation of exposure potential. Abiotic (the result of light or reaction with water) and biotic (the impact of living activity) transformations affect a chemical emissions' concentration and persistence in an environment. These transformations modify the parent chemical into a subsequent derivative. High levels of abiotic and biotic activity work to reduce persistence; however, conditions vary over time and space. The same quantity of a particular chemical emission will persist in the environment and affect living organisms differently, depending on the situation (Davis *et al.*, 1994c, 14).

Bioaccumulation describes a situation when an organism absorbs a chemical and takes it to a concentration that exceeds the level in the surrounding environment. Bioaccumulation is a function of a chemical's solubility and depends on the rate of metabolism in both aquatic and terrestrial organisms. A creature with high metabolism, for example, will lower its rate of chemical bioaccumulation faster than an organism with lower metabolism. Particular factors such as chemical solubility and metabolism, unique to each pollution situation, affect bioaccumulation. In sum, exposure depends on the amount of chemical emission, the medium of release (water, air), and environmental conditions that vary with different pollution situations (Davis *et al.*, 1994c, 14)

In addition to these physical circumstances, uncertainties with the use of the algorithm involve missing data. In the CCPCT index, the inhalation endpoint data for the human health effects have missing entries for many chemical emissions. As a result of a low volatility, either no reported air releases exist, or no experimental data are expected. Improvements in the calculations of the algorithm entail the collection of missing data for the toxicological endpoints.

Davis *et al.* (1994c) promise a yet-to-be-designed second confirmation tier for the CCPCT index. Recall that the first, or screening, tier relies on available information to include rather than eliminate chemicals of potential concern. The assessment of the first tier is designed to prioritize chemicals on the basis of risk evaluation. Adding to the database, the confirmation tier will assess the chemical emissions with complete information. This second tier will avoid false positives by identifying only the chemicals of concern. With the addition of the second tier, the algorithm will produce more robust results.

Another improvement in the algorithm involves the scoring for the chronic human health effect endpoint. Chronic human health effects include developmental, reproductive, mutagenic, neurotoxic effects and cancer. For cancer, the CCPCT index uses a value for the carcinogenicity of a chemical based on its weight-of-evidence (WOE) classification. As Davis *et al.* explain, "WOE classifications refer only to the amount and adequacy of the available evidence and not to the potency of the carcinogenic effect or the mechanisms involved" (Davis *et al.*, 1994c, 11). An improvement in this aspect of the scoring criteria may entail the use of something other than the semi-qualitative WOE data.

A final improvement involves additional research with the weighting mechanism of the algorithm. In its original form, the human health and environmental endpoints contribute additively to the formulation of total hazard. In addition, the exposure parameters contribute multiplicatively, although with a different scoring potential. It may be that each parameter within the algorithm should include a coefficient of influence according to its relative importance. Carcinogenic impacts, for example, may be more important than oral or inhalation. It may be the case that the relative importance of each endpoint depends on circumstance of application, i.e. region. Further research could address the issue of the relative importance of each parameter.

Additional Research

Since this dissertation represents the first attempt to formulate a risk-based tax policy to regulate chemical emissions, further research will enrich the analysis. First, additional research will address the issues of risk units and tax rate. The choice of the numeraire chemical emission initiates the regulation process. The important issue involving the "proper" tax rate follows. A regulator could use a scientific basis in formulating the tax rate by calculating the damage value (human health, environmental, and exposure potential parameters) of one unit of the numeraire pollutant. This way, the relative risk values of the other chemical emissions will correspond to the damage of one risk unit. For policy implementation, a regulator could use the initial, scientific value for a risk unit as a starting point to learn how the risk-based tax rates influence behavior, affect environmental quality and generate revenue.

Second, future research must address the fact that the risk-based system may have to operate alongside other forms of environmental policy, particularly technology requirements and emission standards. The goal of the risk-based system is to establish pollution prices and account for environmental damage. The tax rate on one risk unit that reduces pollution significantly in the absence of other environmental regulations may differ from the proper rate in a mixed policy setting. Future research could also consider whether production costs would decrease if government implemented the risk-based system in lieu of existing standards (e.g. the Clean Air Act.) Under the jurisdiction of the risk-based incentive policy, firms may achieve target levels of emission reduction at lower cost.

Third, the risk-based tax system could serve as an effective environmental policy for industries that emit a wide range of chemical emissions. Future research will determine which industries represent good targets for regulation. Standard Industry Classification 28—Chemicals

and Allied Products—represents a solid starting point. But many other manufacturing industries may qualify since they emit a large amount of chemical emissions.

Fourth, this dissertation does not consider the impact of the risk-based levy system on certain "macro" effects in the economy such as GDP, consumer expenditure, manufacturing output, service sector output, unemployment and distribution. A researcher could analyze one or more of these issues in a general-equilibrium framework. The potential to reduce real income would prohibit the implementation of the policy. It may be the case, however, that a cleaner environment would lead to long-term benefits that impact employment.

Fifth, additional research could attempt to determine if risk-based taxation has "double dividend" characteristics.[87] At a sufficient tax rate, a system of risk-based levies would influence firm behavior to the point where polluters change the composition of emissions and enhance environmental quality. This effect would satisfy the first dividend of environmental taxation. In addition, the tax system would generate a revenue stream over time. Even though the tax base would shrink when polluters substitute less-harmful emissions for emissions with higher damage values, the revenue stream could remain relatively stable over time, depending on tax elasticities. Polluters may not be able to substitute away from certain chemical emissions in the production process. With this potential stability, the revenues from the risk-based tax system could finance a reduction in another tax that distorts economic decision making. This process of revenue recycling could lead to tax system efficiency gains and the second dividend of environmental taxation.

Sixth, additional research must evaluate the aggregation potential of the Parry (1995) framework. Equations that would incorporate a set of emissions, output or input taxes in a revenue-recycling model and evaluate the three efficiency effects would contribute to the literature. In addition, the regulation of a particular set of industries would make it possible to determine the

[87] See Goulder (1995a) for a discussion of the double dividend.

potential for the substitution between consumption and leisure in a realistic model. This analysis would aid the calculation of the tax-interaction effect.

Finally, risk-based taxation would presumably burden industries more that emit a high percentage of the chemicals with large hazard values. A topic of research could involve the measurement of tax burden that results from the application of the risk-based policy. Since risk-based taxation requires payment according to chemical emission levels, the policy could affect industry competitiveness. Industries that emit a larger percentage of the most hazardous pollutants bear the burden of higher tax payment. An equation that entails tax payment divided by output would calculate relative industry burden. Industries could be compared at the state or national level. With appropriate output and emissions data, individual firms could be compared within one industry. The calculation of relative tax burden may justify some form of ecological tax reform (revenue substitution) to accompany the risk-based tax policy.

A PROPOSAL FOR ADOPTION OF RISK-BASED ENVIRONMENTAL TAXATION

Regardless of federal regulation, many reasons exist for states to implement a risk-based environmental tax system. This mechanism could serve state environmental and fiscal goals. At the state level, regulators could identify polluters and target effective policy. In addition, state regulation could provide policy makers with experience and familiarity that could be used at the national level. In other words, a risk-based taxation could build momentum for a national shift toward this form of regulation.

State governments already impose various environmental taxes on water effluent, motor fuel and utility sales (Bohm and Kelsay, 1992). Proposals to increase these taxes as a percentage

of total annual tax revenue have been on the agendas of many states. The implementation of the risk-based system would provide additional environmental tax revenue from a wide base.

Implemented at high-enough rates to impact environmental quality, risk-based taxes could produce interstate price differentials that could damage state competitiveness. Since states do not want to discourage investment with taxes, policy makers could include the risk-based levy system as part of a revenue-neutral tax package. The policy could then shift some tax burden from work, consumption and investment to pollution.

Systems of risk-based taxation at the state level do not preclude the possible future implementation of federal regulation. A policy of state-federal cooperation would help to enforce tax payment. Although the individual state would administer the risk-based tax system, state and federal coordination would aid in the process of administration.

Implementation Sequence

A state could implement the policy with the idea of improving environmental quality. To achieve this goal, the tax rate must give polluters the incentive to change their production processes over time. While decreasing the tax base, the system would generate revenue over the life of the policy. Setting the tax rate at a low level to sustain a stable revenue stream would not work towards the goal of reducing harmful chemical emissions.

The first implementation task entails the choice of target emissions. The CCPCT index covers 158 chemicals (140 from the TRI) and twenty-one high-volume pesticides (Davis, *et al.*, 1994c, 6). These chemical emissions would constitute the initial targets since ranking information exists. As the regulator gains insight into the risk-based system over time and hazard information becomes available for more pollutants, chemical emissions could be added or deleted. Since firms

already have to report TRI emissions each year, risk-based taxation would not impose new monitoring requirements.

Policy makers must evaluate substitution opportunities in the production process of target industry. Improvements in environmental quality would depend on the ability of firms to alter the mix of chemical inputs to use cleaner alternatives. Certain inputs, however, are crucial. Firms cannot substitute away from cadmium emissions, for example, in the production of cadmium batteries. The idea behind the risk-based system is not to reduce the total level of industry emissions, but lower the level of *damage* that the emissions impose on humans and the environment. If emissions remain constant after tax implementation, but the level of risk decreases for target populations and environments, then the system succeeds. At the moment, limited research is available that considers the opportunity to substitute cleaner emissions in the production process for their dirtier counterparts (see Davis *et al.*, 1994a).

Since the pollution problem entails chemical emissions, the second administrative task involves industry identification. Section 313 of the Superfund Amendments and Reauthorization Act (SARA) requires firms in SIC codes 20 through 39 (major manufacturing categories) to report to the TRI. The risk-based system, however, could apply to all firms that discharge target pollutants. (The inclusion of all polluters would increase the revenue calculations in Appendix 4.) Alternatively, the expansion of the tax base (to include more firms than those in SIC codes 20-39) would able regulators to lower the rate to achieve a target level of revenue.

The identification of specific sources allows the use of the risk unit as tax base. For one, the risk-based framework relies on accurate emissions data. Secondly, researchers could apply the CCPCT algorithm to the emissions of particular sites to improve the damage estimations. If polluters are difficult to identify, the risk-based system does not provide fair regulation.

The third task requires the establishment of an open dialogue between government and business. The dialogue must remain open throughout the life of the system. The state, for instance, would make clear the implications of the risk-based system, explain its policy relevance and elucidate the constitutional requirements. The taxes should be predictable and stable so firms can plan abatement strategies. Businesses in turn would provide insight into their technological capabilities, the effect of taxes on input costs and estimated behavioral response. A governing agency cannot determine the effects of the risk-based levy on behavior by exclusively looking at annual emission figures. Regulators need to know if the taxes are providing incentive for firms to change their production processes. While firms will argue for lower tax rates, their insight will aid the regulatory process.

Fourth, before tax implementation, researchers must model industry behavior with and without the risk-based tax system. By calibrating a model of firm behavior within an industry without the influence of risk-based taxes, a researcher would then be able to estimate how the new levy may subsequently affect firm behavior over a range of tax rates. This research would provide empirical basis for a single tax rate. The research would also forecast revenue, determine potential distribution consequences, calculate the expected value of benefits or damages avoided and determine how the new system may interact with existing regulation. The revenue estimations would aid the formulation of a revenue-neutral policy. The calculation of potential damages avoided would help to predict firm response. The modeling of the interaction between new policy and existing regulation would determine the extent of the internalization of the externality. The model could also consider whether the risk-based system would help to enhance environmental quality more effectively than existing regulation. The incentive policy, for instance, could lower the same level of emissions at a lower cost than technology standards.

Fifth, the environmental authority must establish the damage framework and tax rate. For instance, the regulator must determine the numeraire chemical to use to formulate relative risk units. In addition, the policy maker must consider how to formulate the tax rate on one risk unit. The state may want to achieve a certain level of emission reduction or raise a target level of revenue. Firms, in contrast, want to know their tax payment schedule ahead of time in order to make production decisions.

Sixth, the environmental authority must develop a timetable for reporting emissions and a mechanism for enforcement. If the tax rate applies to emissions over a course of a year, firms must know when to report emissions to the state. A framework could be established where regulated firms report their emission every month. In the system, the tax rate on one risk unit would remain constant so firms have an idea about their annual tax payment. The state must be confident that firms are not under-reporting the emissions of pollutants with high hazard values.

The seventh task involves the determination of whether the risk-based system constitutes a long-term policy. Indeed, short-run goals may conflict with a long-term objective. If regulators adopt a long-term perspective, firms will change their production processes, consider capital investment and adjust to the new environmental regulation. But with a myopic outlook, firms may not alter their production methods.

Eighth, the state government must include measures in the policy package that eliminate all non-price barriers to effective pollution abatement. Information restrictions concerning abatement technologies could limit adjustments in production. Other government regulations already in existence could restrict the effectiveness of the environmental tax system. Eliminating these conflicting policies would increase the elasticity of substitution between inputs and lead to a less-costly and shorter period of adjustment.

The ninth, and last, task involves the formulation of a revenue policy to incorporate into the overall policy package. Possible revenue uses include a contribution to the general budget, revenue recycling or earmarking. For the sake of political feasibility, the former possibility does not serve as an appropriate option. The possibility for revenue recycling, however, may aid the passage of the risk-based tax program. A legislator could sell the system as a method to shift some tax burden from labor to polluters. The revenue from the system could finance lower rates on existing income taxes. In this sense, the public would not perceive the risk-based system as new, green taxes, but part of a revenue-neutral policy package. The state could also earmark revenue to subsidize pollution abatement for regulated firms. The revenues could partially finance research and development.

When regulators have targeted industries, identified firms, established a set of emissions to regulate, opened dialogue with business, modeled firm behavior, forecasted revenue, formulated relative risk units for the chemical emissions, calculated an appropriate tax rate or schedule of rates, determined a monitoring schedule, prioritized the long term, eliminated non-price barriers, and set up a policy for revenue use, regulators may implement the risk-based tax system. As with any new environmental regulation, the extent of emission reduction is unclear. But risk reduction through the emission of less-harmful pollutants would favor the environment.

Implementation Problems

Since the policy constitutes an initial attempt at a risk-based tax regulation, problems could occur at different points in the implementation process. Regulators, however, could draw on existing experience with emissions taxation to help facilitate the process. The government must convince stakeholders and the general public that the chemical emission problem requires action

and the risk-based system represents an efficient solution. This problem of acceptability could arise from concern over a new tax. But the focus on emissions could help to convince the public. Government must assure business and the public alike that they will have feedback throughout the life of the policy.

Another problem could involve perception if businesses think the tax system will stifle industrial competitiveness. But the government could use the revenues to finance lower rates on existing taxes in a revenue-neutral policy. In addition, government could eliminate existing regulation that might conflict with the new incentive policy. This latter possibility means that the implementation of the risk-based tax system could actually *lower* production costs. Either method would ensure that risk-based taxation would not place undue burden on business.

For tax implementation, regulators must rely on a detailed description of the sectors involved. This knowledge base would largely come from industry, since tax administrators may not have input data. Reliance on business could lead to a situation where industry has undue power over the authorities. To avoid this occurrence, policy makers could implement the system slowly. During this time, government and business could work together to build the information base necessary to establish effective policy.

The regulatory authority must establish a philosophy to address the problems of compliance and enforcement. Even though firms are required by law to report emission content and levels, compliance could fall below 100 percent. In other words, firms could underreport emissions to lower tax payment. The problem would depend on the extent and persistence of under-reporting. As a result of this possibility, the government must enforce the regulation. Enforcement requires periodic checks of the firms to make sure the reports are accurate. Without a workable policy to address these issues, problems of compliance and enforcement could limit the effectiveness of risk-based taxation.

THE PORTFOLIO PROBLEM OF RISK-BASED ENVIRONMENTAL TAXATION

Neither effectiveness nor the concept of cost-effective provides a sole criterion to judge an environmental policy. Efficiency establishes one criterion, but additional aspects of environmental policy must be taken into account. When implemented along with a policy of revenue substitution, the risk-based tax system would create benefits in the form of environmental improvement and revenue-recycling effects. The costs include the efficiency losses of tax interaction. The theoretical discussion in chapter four explains under what circumstances the benefits may exceed costs in the context of efficiency. The overall impact of the risk-based system, however, also depends on the other evaluation criteria established in chapter one. From a regulator's perspective, the portfolio problem of environmental taxation means that the implementation of the risk-based system (as it is outlined above in the implementation sequence) entails the possibility of creating benefits to society at the expense of important policy costs. The net effect of the system depends on the economic value of each. An emphasis on environmental improvement and a relatively higher tax rate on harmful chemical emissions, for instance, may entail important administrative and compliance costs. A goal of implementing a policy of revenue substitution may maximize the revenue stream over time but sacrifice the potential for higher levels of environmental improvement. Considering the portfolio problem, the policy maker must determine whether the tax system creates more benefits than costs. In a benefit/cost framework, the following subsections return to the evaluation criteria of chapter one to critically assess the potential for risk-based taxation.

Benefits

The benefits of the risk-based levies include the potential for environmental enhancement, equity, production, incentive and revenue effects and certain aspects of administration. The target chemical emissions invariably lead to many forms of human health and environmental damage. Because the risk-based system implements relatively high tax rates on chemical emissions with large risk values, a reduction in the damage effects of target pollutants will result if the tax provides the incentive for a change in production methods. The benefits of a reduction in risk could be measured on an economic basis by improvements in human health (fewer cancer deaths, lower rates of mortality, etc.) and environmental quality (lower levels of bioaccumulation and persistence, etc.)

Fairness is another important issue in connection with environmental policy. An equitable regulation balances costs and benefits across all affected parties by appropriately distributing benefits and/or allowing those who benefit to pay an appropriate percentage of the costs. The risk-based tax would be judged inequitable if the individuals or regions that bear the burden of the policy do not share the benefits. If the policy stifles development by reducing employment over time, the value of enhanced environmental quality may not exceed the costs in a particular area. Since the system targets a reduction in harmful emissions, environmental benefits in the form of cleaner air, water and earth would accrue to all who live in an area affected by the target pollutants.

But the presence of the externality and the policy to correct it would affect the distribution of wealth in target regions. The polluter pays principle—the idea that those who pollute should pay for the damage—seems to apply in this case since the environmental regulation under consideration is new. It is not the case that residents are moving to an area with a factory and do not know about the existence of harmful chemical emissions. The tax, however, could reduce the

income of factory owners if they internalize the pollution externality.[88] Wealth could be redistributed to homeowners if housing prices rise as a result of a cleaner environment. The question to consider concerns whether the redistribution effect exceeds the value of the economic damages themselves. It is possible that the value of the tax paid by polluters could exceed the economic value of damages, including human health, property and environmental effects. This question is empirical in nature and arises in the presence of any environmental regulation. It is safe to say that the benefits of enhanced environmental quality would accrue to all, while polluters are forced to internalize the pollution externality partially or fully.

Higher levels of production efficiency could result from the regulatory system. By creating a price for chemical emissions, these pollutants become inputs in the production processes of target industry. The risk-based regulation forces firms to include emissions in their input matrix. An improvement in environmental performance by reducing the emissions of chemical inputs may entail expenses in either money or time. Abatement costs depend on the degree of importance of the chemical in production, the level of technology, capital/labor ratio and competitors' positions with respect to abatement. If the tax rate exceeds abatement costs, firms have the incentive to reduce emissions. The point is that chemical emission abatement resulting from the policy may force firms to account for pollution residuals. Firms have the opportunity to implement least-cost abatement technology and establish more efficient production processes. Higher levels of productivity that result from this environmental regulation would be an example of the "Porter Hypothesis" (Porter, 1991).

The risk-based system would generate a revenue stream over the life of the policy. The goal from an environmental standpoint is to decrease the tax base and reduce pollution; therefore, a state government would not enjoy the benefits of a stable revenue stream indefinitely. Increases in

[88] But if they change the production processes and become more efficient, costs could decrease.

tax-system efficiency, however, could involve the use of a revenue-neutral policy package in conjunction with the implementation of the risk-based system. The value of a revenue-recycling effect would depend on the revenue stream. The environmental levies in the risk-based system could finance lower rates on income or capital taxes at the margin, but would not be able to sustain the revenue substitution over time, as the environmental tax base would decrease. The benefits of the revenue-recycling effect would therefore be short-term in nature.

From the perspective of administration, the system would be able to adapt to a changing policy environment. If a regulator wants to alter the tax rate on one RU in an iterative process to account for pollution damage or impact behavior at the margin, the framework would allow for adjustments. Chemical pollutants could also be added or deleted without a loss in continuity. The policy of risk-based taxation draws on the requirement of the EPA that firms must report to the Toxics Release Inventory. This use of existing policy means that firms would not have new reporting requirement. Data collection would not present a high administrative expense.

Costs

A proposal to implement the risk-based system at the state level must include estimated administrative costs. Fortunately, the system uses data already required by law, a straightforward tax rate schedule and annual collections. Initial administrative costs include the identification of target emissions, risk units and the tax rate. Over time, the governing authority would have to enforce the regulation and maintain open dialogue with business. Regulators would have to make sure that businesses continue to report all target emissions. Since it would target (at the very least) SIC industries 20-39, enforcement costs would exist as long as firms have the incentive for under-compliance. Problems of evasion and/or avoidance may exist in industries that emit a relatively higher proportion of the chemicals with greater damage values. Relative to other industry, these

businesses would make higher tax payment. These expenses could jeopardize the potential for overall policy benefits, but would invariably result from the nature of the tax framework (Alm, 1996).

Administrative costs would also include outlays to continue the research of techniques designed to calculate pollution damage. This expense would continue over time as chemicals are added to the regulatory system and site-specific estimations are required. For target chemical emissions, industries and firms, government would have to estimate whether the CCPCT index provides responsible measures for risk units for each pollutant. This activity would entail (contracted) fieldwork,[89] further estimation of the damage values of individual pollutants and comparison of the estimated values with the CCPCT index. Another small cost includes data collection for the state. While not extensive in scope since large manufacturing firms must report to the TRI every year, the state would have to establish a data bank to list the industries, firms and chemical emissions that operate under the jurisdiction of the policy.

A potential cost to certain regions involves the issue of pollution source and destination. The place where chemical pollutants are emitted is not necessarily the location where damage occurs, as wind and water may carry pollution across state lines. Yet the policy is intended to fall under the category of the polluter-pays-principle. If implemented at the state level, the risk-based policy could place certain regions at a competitive disadvantage since they may or may not be affected by harmful effluent. In a time when states offer tax preferences to attract large business, the presence of a new environmental policy would not enhance the ability to entice new industry. As a result, the implementation of the risk-based system at the state level could serve as an initial attempt at regulation in a movement to a national policy.

[89] The Center for Clean Products and Clean Technology at The University of Tennessee, for instance, could update and expand their data bank and continually apply the algorithm to individual pollutants.

The risk-based system also incurs regulatory, production and monitoring costs to the firm and incidence and tax system costs to society. Polluters must pay the tax as long as they emit regulated chemicals. As a result of tax implementation, firms may have to alter production methods to lower tax payment. This adjustment could entail capital investment and new technology for a cleaner production process. In addition, polluters may alter the mix of emissions. This switching involves substitution costs in the form of research and development. Monitoring costs depend on chemical coverage and the reporting schedule.[90] If the risk-based policy operates within the framework of the TRI, new monitoring costs will not exist. However, if businesses are not currently required to report to the TRI, new costs would appear.

From the perspective of society as a whole, industrial areas that emit more pollutants with higher damage values would bear the burden of the tax system more than "cleaner" regions. This potential for an uneven pattern of incidence exists with many new environmental policies (Lesser *et al.*, 1997, 187-189). To the extent that poorer residents occupy the industrial areas, two important "fairness" issues would result from policy implementation. First, employment reduction and higher housing values (rents) could reduce the quality of life for many poorer residents. Second, by relocating away from industrial areas, residents who are able may already protect themselves from any environmental problems. If the risk-based system is implemented as new regulation and does not replace existing policy, higher production costs borne by consumers would detract from overall benefits.

In addition, tax-system efficiency losses could occur. With a policy of revenue substitution, environmental taxation creates a tax-interaction effect. This welfare impact includes the potential increase in production costs and labor market distortions. If the revenues from the risk-based system are used to finance a lower income tax, a loss in real income could occur as the

[90] If the risk-based system replaces direct control, compliance costs could decrease.

environmental tax base decreases over time and cannot fully finance a lower labor tax. For the sake of political feasibility, the revenues from the risk-based system may have to finance a lower rate on a business tax. In a revenue-neutral setting, a lower tax on capital coupled with the risk-based levies could increase investment and reduce the emissions of chemical pollutants.

Existence of Net Benefits?

The application of the evaluation criteria to the risk-based regulation above elucidates the fact that important costs accompany the benefits of policy implementation. After policy implementation, would net benefits result over time? The answer depends on the scope of the policy and the portfolio choices of the regulator. Within the framework of the system, the regulator could emphasize different criteria. Table 30 reproduces the evaluation checklist from chapter one. Based on the above discussion, each criterion could lead to a cost, benefit or neither over time.

If the regulator intends to implement the risk-based system as a long-term policy, a sufficient tax rate could encourage pollution abatement. This impact would lead to important environmental benefits. As it stands, the value of environmental improvement is crucial for net benefits to occur. As with any tax, the potential for administration, compliance and efficiency costs exists. If a benefit/cost framework is used to justify the policy, lower emission levels and a large economic value placed on environmental improvement could offset the costs that would invariably result. But since many chemical emissions have high damage values per unit as compared to criteria air pollutants, it may be that environmental improvement represents the most important portfolio criteria for many regions, and may justify the administrative, compliance and efficiency costs that result.

Table 30
Criteria checklist for risk-based taxation

Criteria applied to risk-based taxation	Benefit	Cost	Criteria applied to risk-based taxation	Benefit	Cost
Environmental impact	X		Compliance		X
Efficiency			Monitoring		X
Tax interaction effect		X	Evasion		X
Revenue recycling effect	X		Avoidance		X
Equity	X		Revenue	X	
Administration		X	Yield	X	
Adaptability	X		Adequacy		
Collection	X		Short-run stability	X	
Enforcement		X	Elasticity		
Source/destination		X			

FINAL COMMENTS

The risk-based environmental tax system applies to pollution situations with chemical emissions. The idea is to embed, for target firms, a pollution price for chemical emissions into the matrix of input costs. In the absence of research that formulates marginal damage functions for target pollutants, the risk-based system provides an indirect measure of damage. By quantifying chemical emissions by risk units, a single tax rate applies a different charge to individual pollutants and forces firms to internalize the externality partially or fully. The main contribution of this paper consists of the development of the risk-based tax system as a proxy for a fully informed regulation. By formulating a linear approximation of true marginal damages, the risk-based policy establishes prices for chemical emissions in the decision matrices of target manufacturing firms.

Policy evaluation involves the accuracy of the risk-based system. When does the regulation provide accurate estimates of marginal damage? This result occurs when damage from target pollutants is relatively stable as emissions increase. How realistic is it to expect stable marginal damages? This occurrence probably happens infrequently, although it is difficult to

speculate on the shapes of the damage function of many chemical pollutants. The impact depends on a number of factors, which include method of disposal, proximity to humans and animals, synergy between pollutants and many others. What may we conclude, in the presence of non-linear damage, about the accuracy of risk-based taxation? The regulation breaks down in terms of pricing the externality accurately when marginal damage for target pollutants differs greatly from a linear function. This result implies that the risk-based system has a difficult time making tax rates equal true marginal damages. But, given the information limitations, the risk-based system provides a framework to assign prices to a set of pollutants.

In the implementation of the new incentive policy, the regulator may choose to emphasize different portfolio criteria. Presumably, efficiency may be the highest priority in the form of environmental improvement, but important costs exist that must be considered, including administration, compliance and tax-system effects. It is important to note that the emphasis of an individual criterion will impact the effectiveness of the regulation. From an environmental perspective, the priorities of the policy maker are therefore crucial for policy success. Placing a high value on environmental achievement in the long-term would help to justify the policy.

While the analysis in this chapter argues that a state could operate the system, the EPA could oversee a national policy. The risk-based framework provides incentives for industry to change the chemical emissions mix in the production process. The policy also discourages the emission of pollutants that register high damage values in the CCPCT index. The system is flexible in that policy makers could apply it, in addition to direct regulation, to provide incentive for pollution reduction. Risk-based taxation could also replace direct control. From an administrative standpoint, the risk-based system allows extensive coverage of a number of chemical emissions. For the firm, the system allows flexibility in the abatement process. The risk-

based policy may have to include some form of revenue substitution to keep the tax payment of target industry constant.

This paper has developed three major points. First, the risk-based framework provides a logic for the regulation of chemical emissions. The application of the risk unit allows the regulator to encompass a number of pollutants within the jurisdiction of the policy and apply one tax rate. Second, the system focuses on the damage function for individual pollutants in the absence of marginal abatement cost information. This contribution results from the goal of establishing practical environmental policy. Third, the risk-based system serves as an appropriate policy in lieu of a fully informed emissions tax when the regulator wants to regulate a number of chemical emissions, the polluters are easily defined, and target chemical emissions display stable marginal damages.

The implementation of risk-based taxation, however, remains a portfolio problem for the regulator. Policy implementation would generate specific costs to accompany the benefits of environmental improvement. The problem is that certain pollutants may lead to marginal damages that are so non-linear in nature that the risk-based policy does not implement accurate prices. But in these instances the new policy still establishes the pollutants as factors in the production matrix of target firms. Returning to the quote by Turner, Pearce and Bateman (1993) at the beginning of this dissertation, a practical policy method in the absence of full information is to levy environmental taxes on target pollutants according to the relative magnitude of emission damage. Levying rates in this manner is the objective of the risk-based system. While raising potential administrative, compliance and efficiency costs from the perspective of society, the extent that the system provides welfare gains depends on the level and value of pollution reduction.

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APPENDICES

Appendix 1

THE STATE OF ENVIRONMENTAL TAXATION

Environmental taxes are appropriate public policies in many pollution situations. Many pollution problems exist, however, that prevent the use of emissions or risk-based taxation. Appendix 1 discusses environmental taxes in OECD countries and discusses the situations applicable for different environmental tax bases.

CURRENT ENVIRONMENTAL TAXES

An important tax implementation issue concerns policy jurisdiction. Should national, regional, or local governments levy environmental taxes? "The scope of the taxing authority affects the efficiency and effectiveness of any tax strategy" (OECD, 1995b, 38). Municipalities, for instance, are in the best position to address local environmental problems. Municipal waste user charges serve as an example. But as pollution floats downstream or crosses borders through the air, the problems become regional and better addressed by state or national authorities. Table 31 lists environmental instruments in OECD countries at the national level by taxable base.

Because some environmental tax bases are not obvious, this subsection lists bases and accompanying rates. This section demonstrates the relative dearth of environmental emissions taxes as compared to environmental output and input taxes. Most of the tax tables also elucidate when environmental taxes elicit incentive response. Note that the different countries earmark the revenue for particular projects or the general budget.

Table 31

Overview of environmentally related taxes in OECD countries: national level

Environmental Taxes	Australia	Austria	Belgium	Canada	Denmark	Finland	France	Germany	Greece	Ireland	Italy	Japan	Mexico	Netherl	Norway	Portuga	Spain	Sweden	Switzer	Turkey	UK	USA	Total
Motor Fuels:																							
Leaded/unleaded (differential)	x		x		x	x	x	x		x	x		x	x	x	x	x	x	x	x	x		17
Diesel (quality differential)					x	x									x			x					4
Other Energy Products:																							
Carbon/energy tax					x	x					x			x	x			x	x				7
Sulfur tax															x			x					2
NOX charge							x									x		x					3
Vehicle Related:																							
Sales/regis tax diff (cars)			x		x	x			x	x	x		x	x	x	x		x	x	x		x	14
Road/regis tax diff (cars)			x	x	x			x		x	x	x	x	x	x		x	x	x	x	x		15
Emission diff (cars)		x			x			x						x									4
Agricultural Inputs:																							
Fertilizers		x				x									x			x					4
Pesticides					x	x									x			x					4
Soil protection			x											x									2
Air/Other Goods:																							
Air pollution							x				x					x		x				x	5
Batteries			x		x											x		x					4
Plastic carrier bags					x																		1
Disposable containers			x		x	x									x	x		x					6
Tires				x	x																	x	3
CFCs and/or halons	x	x			x																	x	4
Disposable razors			x																				1
Disposable cameras			x																				1
Lubricant oil charge						x	x				x				x								4
Oil pollution charge	x					x																	2
Air/Water/Waste:																							
Noise charges	x		x				x	x				x		x	x	x			x				9
Sewage charges			x					x						x									3
Waste disposal charge		x	x		x		x			x	x			x		x	x		x	x	x		12
Hazardous waste charge		x	x			x		x							x	x						x	7
Total:	4	5	12	2	13	10	6	6	1	4	7	2	3	9	12	9	3	12	6	4	3	5	

Terminology: "diff" = differential; Sources: OECD, *Implementation Strategies for Environmental Taxes*, 1995; and OECD, *Managing the Environment: The Role of Economic Instruments*, 1994

Table 32 shows that revenue from tax differentiation goes into general budgets. The tax rates, moreover, are similar between countries. In Germany, the Netherlands and Norway the market share of unleaded fuel increased dramatically a few years after the implementation of the tax differential. Canada phased out leaded fuel beginning in 1990 (OECD, 1994a).

Table 32[91]
Tax differentiation between leaded and unleaded petrol

Country	Tax rate	Incentive effects	Revenue use
Austria	ECU 0.03/1	---	General budget
Belgium	ECU 0.03/1	Market share from 0 to 32.5% in 1989-1990	General budget
Canada	ECU 0.01/1	Leaded phased out since December 1990	General budget
Denmark	ECU 0.07/1	Market share is 75%	General budget
Finland	ECU 0.07/1	Market share attained 70% in 1992	General budget
France	ECU 0.06/1	Market share rose to 25% in 1991	General budget
Germany	ECU 0.04/1	Market share from 11 to 88% in 1986-1992	General budget
Ireland	---	---	General budget
Netherlands	ECU 0.05/1	Market share from 32 to 75% in 1989-1992	General budget
Norway	ECU 0.07/1	Market from 17 to 55%	General budget
Portugal	ECU 0.05/1	---	General budget
Sweden	ECU 0.04/1	Market share attained 55% in 1991	General budget
Switzerland	ECU 0.04/1	Market share attained 65% in 1992	General budget
UK	---	---	General budget

Symbol: --- = no information available

The leaded/unleaded tax differential seeks to encourage the use of a cleaner fuel alternative by raising the price of dirtier fuel. As an output/input tax, this levy indirectly addresses the exhaust problem from vehicles. It indirectly discourages the use of fuel inefficient automobiles.

Carbon taxes (Table 33) are levied according to the carbon content of fuels and on top of existing charges. Denmark, Norway, and Sweden recently adopted the carbon tax. In Denmark, the national government levies a CO_2 tax on all types of CO_2 emissions sources except gasoline,

[91] As of March 7, 1998 1 ECU = 1.09 U.S. dollars.

natural gas, and biofuels. Based on the CO_2 content of each fuel at combustion, the tax rate is 100 DKr[92] per ton of CO_2 (OECD, 1994a).

Table 33
Explicit and implicit carbon charges

Country	Rate per ton CO_2	Implicit rate on existing taxes	Incentive: intended	Incentive: actual
Denmark	ECU 5.5-11.1	ECU 26.3	+	..
Finland	ECU 1.1	ECU 19.1	#	..
Italy	ECU 1.7	ECU 39.9	..	..
Netherlands	ECU 0.4	ECU 15.9	--	..
Norway	ECU 13.8 on coal ECU 15.7 on fuel oils ECU 40.6 on gasoline ECU 40.6 on natural gas	ECU 32.5	+	..
Sweden	ECU 37.9	ECU 38.2	+	#

Symbols:
+ = yes
-- = no
.. = no data available
# = unclear

The carbon charge seeks to price the negative externality of CO_2 emissions. An input tax that links directly to CO_2 emissions, this levy eases the process of administration as existing tax mechanisms allow policy makers to add a carbon charge to other energy payments, such as a household's energy bill.

Table 34 shows that, in Sweden, the sulfur tax decreased the sulfur content of oil by about 30% from 1990 to 1992. In addition, the level of emissions fell from the burning of coal and peat in this country. An input tax, this charge targets the composition of energy. The increase in relative price encourages an emission reduction, like in Sweden. While a government could require a certain energy content for any particular form of fuel, a tax uses a market incentive to price the fuel content.

[92] In March, 1998 Denmark's currency, the Krone, fluctuated around 6.9 to the dollar.

Table 34
Charges on sulfur content of fuels

Country	Charge base	Incentive: intended	Incentive: actual	Revenue use
Norway	S-weight % of coal	+	#	General budget
Sweden	S-weight % of oil, coal and peat	+	+	General budget

The output charges in Table 35 vary widely between countries. Greece reports that a major change occurred in the new car market (OECD, 1994a). Manufacturers equip all new cars with catalytic converters. In addition, consumers of these new cars receive a scrapping refund. These two policies reduce automobile emissions. The sales tax differential in the United States, moreover, discourages the purchase of gas-guzzling cars. Other charges discourage the use of cars that do not meet predetermined standards for environmental quality. In these situations, the governing authorities of the different countries could implement a direct regulation that requires automobile manufacturers to produce cars that satisfy certain criteria, such as having catalytic converters. The taxes, however, raise revenue. The sales tax differentials discourage the use of environmentally unfriendly cars.

The tax differential policies described in Table 36 take into account harmful emissions, but use different tax bases. In Germany, the tax rate depends on an index of harmful emissions such as carbon monoxide, nitrogen dioxide, and hydrocarbons. By distinguishing between pollutants and equipment that affect environmental quality, Austria, Germany, and the Netherlands are creating prices for the use of the environment. Taxes are appropriate to change the relative prices of equipment and emissions to favor environmentally-friendly activity.

Table 35
Car sales tax differentials

Country	Tax base and rate
Belgium	Cars not satisfying emission standards: ECU 314-419
Canada	Automobile weight: ECU 17-34
Finland	Cars not equipped with catalytic converter: ECU 714
Germany	Cars not equipped with catalytic converter: ECU 238-478
Greece	Degree of compliance with emission standards
Japan	Tax deductions for cars with low emissions, electric cars and cars on alternative fuels
Netherlands	Cars not satisfying US or EC emission standards: ECU 301-603
Norway	Cars not equipped with catalytic converter: ECU 774
Sweden	Degree of compliance with emission standards: ECU 237-474
USA	Cars with fuel efficiency below 22.5 miles/gallon: ECU 655-4587

Table 36
Emissions differential

Country	Tax base	Tax rate
Austria	Equipment with catalytic converter	Dependent on weight class
Germany	Cylinder capacity and emissions	ECU 3.6/100 cc
Netherlands	Emissions	--

The input/output taxes described in Table 37 distinguish between different fertilizers according to chemical content. The more hazardous fertilizers receive higher charges. Because all kinds of fertilizer are allowed in these countries, the taxes serve as appropriate policy to price the fertilizers for environmental reasons. The taxes are levied on quantities of the fertilizer products at the retail level; policy makers could implement a risk-based tax in addition to the charges that already exist, by distinguishing between the different pollutants. The fertilizer charges discourage product purchase and may indirectly influence farming methods.

The decrease of phosphorus in fertilizer in Finland may be due to additional policy instruments besides the environmental tax. Response has been highest in Sweden to this charge. Sweden earmarks the revenues for subsidies and environmental protection which influences consumer response (OECD, 1994a).

Table 37
Charges on fertilizers

Country	Charge base, rate and % of price	Incentive: intended	Incentive: actual	Revenue use
Austria	N and P content: ECU 0.31, ECU 0.18 and ECU 0.09 per kg	--	--	Subsidies, environmental expenditure
Finland	N and P content: ECU 0.41 and 0.27/kg (5-20% of price)	+	#	Agricultural subsidies, general budget
Norway	N and P content: ECU 0.13 (19% of price) and ECU 0.24/kg (11% of price)	+	#	General budget
Sweden	N and P content: ECU 0.07 and 0.14/kg (10% of price)	+	+	Subsidies, environmental protection

Symbols: N = nitrogen
P = phosphorus

In Table 38, the pesticide charge on active ingredients in Sweden amounts to about 5 percent of price. The levy per treated hectare is an additional charge of about 20 percent of price.

Interestingly, the Swedish Board of Agriculture concludes that the price elasticity of pesticide demand is between 0.2 and 0.5 (OECD, 1994a). These figures mean that an increase in price leads to a small decrease in quantity. Like with the charge on fertilizer, Sweden earmarks the revenues from the charge on pesticides for environmental purposes. Norway, on the other hand, uses the revenue as part of the general budget. In Norway, the pesticide charge raises the price of the product at the wholesale level. As a result, this levy is no different from another output charge from the standpoint of tax policy. In Sweden, policy makers levy a tax that discourages pollution content and economic behavior.

Table 38
Charges on pesticides

Country	Charge base and rate	Incentive: intended	Incentive: actual	Revenue use
Norway	13% on wholesale price	+	..	General budget
Sweden	ECU 0.9/kg active ingredient ECU 3.6 per treated hectare	+	#	Environmental expenditure

Soil protection charges in the Netherlands and Belgium (Table 39) focus on the number of livestock in targeted regions. Belgium and the Netherlands have a surplus manure charge on quantities that exceed environmentally acceptable loads per hectare. The governments of these two countries use the revenue for manure transport, processing, and storage. It would be difficult for policy makers to levy the tax on a per-ton basis. Many policy problems would exist with compliance and enforcement.

In Table 40, the charges on air pollution in France, Japan, Portugal, Sweden, and the United States demonstrate that many tax bases are possible. Sweden experienced success since the

Table 39
Soil protection charges

Country	Charge base	Incentive: intended	Incentive: actual	Revenue use
Belgium	P and N in manure: number of livestock	-	..	Manure policy
Netherlands	P in manure: number of livestock	+	..	Manure policy

Symbols: P = phosphorus
N = nitrogen

Table 40
Charges on air pollution

Country	Charge base and rate	Incentive: intended	Incentive: actual	Revenue use
France	Acidifying emissions: ECU 19.0/ton (SO_2, NO_X, H_2S, N_2O, HCl)	-/+	..	Air quality control
Japan	SO_2 emissions: rates differ regionally between ECU 0.5-4.5	+	#	Compensation of health damage
Portugal	SO_2, NO_X	..	..	Air quality control
Sweden	NO_X emissions of energy producers: ECU 4.7/kg NO_2	+	+	Rebated to energy producers
USA	Criteria pollutants, >= ECU 16.4/ton	-/+	..	Air quality control

levy encouraged compliance before the implementation of sharper standards in 1995. As emissions charges, these levies generally apply to energy producers. Because the revenues from the taxes help to fund different aspects of environmental policy and control, the taxes represent appropriate environmental policies. Authorities levy the air pollution charges of Table 40 on a per-unit basis on a finite number of identifiable sources.

The air pollution charge introduced in 1985 in France applies to waste incineration facilities with capacities over three tons per hour and plants that emit over one hundred and fifty tons per year of sulfur, nitrogen oxide, hydrochloric acid compounds, non-methane hydrocarbons, solvents, and other organic compounds. The Environment and Energy Agency administers the charge of 150/ton (francs)[93] on 1400 facilities (OECD, 1994a).

The battery charge of Table 41 finances collection, treatment and recycling in Denmark, Portugal and Sweden. The battery taxes have in common the implementation procedure. Paid at the retail level, the taxes are levied according to quantity or weight. In the case of batteries, a deposit/refund system would serve as an efficient policy. The authority would want to reduce batteries in landfills. In this case, a deposit/refund system could serve as an appropriate policy.

Table 41
Charges on batteries

Country	Charge base and rate	Incentive: intended	Incentive: actual	Revenue use
Denmark	NiCd batteries: ECU 0.2/piece	-	-	Collection, treatment and recycling
Portugal	Lead batteries: ECU 1-5/piece	-	-	Collection, treatment and recycling
Sweden	Lead batteries > 3 kg: ECU 3.8/kg HgO_X batteries: ECU 2.7/kg NiCd batteries: ECU 1.5/kg	-	-	Collection, treatment and recycling

[93] In March, 1998 the franc fluctuated around 6.1 to the dollar.

For the countries in Table 42, low lubricant oil charges do not affect behavior. The levies finance waste oil treatment in Finland, France, and Italy and contribute to the general budget in Norway. This charge, classified as an input or output tax, is a traditional revenue source for the four countries. Since Finland, France, and Italy use the revenues to fund waste oil treatment, the entire policy (including revenue use) helps to enhance environmental quality.

Table 42
Charges on lubricant oil

Country	Rate	Incentive: intended	Incentive: actual	Revenue use
Finland	ECU 0.04/kg	-	-	Waste oil treatment
France	ECU 0.009/lr	-	-	Waste oil treatment
Italy	ECU 0.003/lr	-	-	Waste oil treatment
Norway	ECU 0.055/lr	-	-	General budget

The CFC charges in Table 43 apply to different tax bases. Before the phase-out program in the United States, for instance, Congress enacted a tax on CFCs (in 1989) that included eight different chemicals subject to different rates: CFC-11; CFC-12; CFC-113; CFC-114; CFC-115; Halon-1211; Halon-1301; and Halon-2402. The tax was formulated by first multiplying $1.37 per pound and then by an ozone-depleting factor assigned to each chemical. The factors represented varying levels of environmental harm (Barthold, 1994, 137). In the United States, the CFC charge applied to the domestic production of CFCs, imports of the chemicals, and imports of products that contained the chemicals. As a result, the CFC charge was a tax on output.

In Table 44, the aircraft noise charge raises revenue for noise abatement and does not provide the incentive to change behavior in these countries. This tax is a Pigouvian levy when it attempts to price the social damage caused by noise pollution, or the taxable base. The tax on waste disposal to land fills and waste incinerators in Table 45 usually operates alongside other waste policies, so incentive effects are difficult to isolate.

Table 43
Charges on ozone-depleting chemicals

Country	Charge base and rate	Incentive: intended	Incentive: actual	Revenue use
Australia	CFCs: ECU 0.07/kg	-	-	CFC phase out
Austria	Appliances containing refrigerants	-	-	Environmental expenditure
Denmark	CFCs and halons: ECU 3.3/kg	#	#	General budget, environmental expenditure
USA	Ozone-depleting chemicals: ECU 2.0-2.4/kg	#	+	General budget

Table 44
Aircraft noise charges

Country	Charge base	Incentive: intended	Incentive: actual	Revenue use
Belgium	Type of aircraft, time of day	-	..	Noise abatement
France	Noise characteristics	-	..	Noise abatement
Germany	Noise class of aircraft	+	#	Noise abatement
Japan	Weight and noise level of aircraft	-	..	General budget
Netherlands	Weight and type of class	-	..	Noise abatement
Norway	Noise characteristics	+	..	No net revenue
Portugal	Weight of aircraft	-	..	General budget
Switzerland	Noise characteristics	-	..	Noise abatement

Table 45
Taxes on waste disposal

Country	Tax rate	Incentive: intended	Incentive: actual	Revenue use
Austria	ECU1.4/ton	+	#/+	General budget
Belgium	ECU 3.1-18.9/ton	+	..	Environmental expenditure
Denmark	ECU 14.4/ton	+	#/+	General budget
France	ECU 2.5/ton	#	..	Treatment and recycling
Italy	..	..	..	..
Netherlands	ECU 6.2/ton	#	..	General budget

In addition to the charges in Table 45, improvements in environmental quality are due to technological improvements in waste treatment, increases in user charges in some countries, and regulations on waste removal and processing. Regulators usually levy the tax on the weight of waste supplied to incinerators and landfills. An unintended side effect of the waste generation charge is that polluters sort out reusable material and the waste stream becomes more polluted. As it stands, the waste generation fee taxes emissions appropriately on the basis of weight.

Policy makers have set the rates for the hazardous waste charges (Table 46) too low to influence behavior (OECD, 1994a). In the US, the hazardous waste charge funds the Superfund cleanup program. In many cases, hazardous waste isolates in its own environment. Countries that regulate this form of pollution also process the waste to ensure environmental quality. While the waste may impact its immediate vicinity, its effects on a surrounding environment are often limited. This is not the case for countries that do not regulate hazardous waste.

Table 46
Charges on hazardous waste

Country	Charge base and rate	Incentive: intended	Incentive: actual	Revenue use
Austria	ECU 12/ton	..	..	Clean-up of contaminated sites
Belgium	ECU 8-63/ton	..	..	Environmental expenditure
Finland	ECU 270/ton on average	#	..	Waste processing
USA	..	-	-	Clean-up of contaminated sites

The following discussion presents an overview of sub-national taxes despite limited information. As Table 47 shows, many sub-national levies lead to different revenue uses. Tables 48 through 50 list the bases and rates for three levies presented in Table 47.[94] Table 49 shows when incentive responses occur and Table 50 lists how governments use the revenues.

[94] For many of the taxes, detailed information is not available. The information on the taxes in this subsection comes from OECD (1994a).

Table 47
Environmental taxes at the sub-national level in OECD countries

Environmental Taxes	Australia	Austria	Belgium	Canada	Denmark	Finland	France	Germany	Greece	Ireland	Italy	Japan	Mexico	Netherla	Norway	Portugal	Spain	Sweden	Switzer	Turkey	UK	USA	Total
Vehicle Related:																							
Sales/regis tax diff (cars)				x																			1
Municipal vehicle tax																x							1
Agricultural Inputs:																							
Fertilizers																						x	1
Soil protection																						x	1
Air:																							
Air pollution				x																			1
Other Goods:																							
Batteries				x																			1
Packaging				x																		x	2
Non-refillable beverage containers				x																			1
Car Tires				x																		x	2
Disposable diapers				x																			1
Lubricant oil charge																						x	1
Water Charges/ Taxes:																							
User charges for sewage and sewage treatment	x		x	x	x	x	x	x		x	x			x	x	x	x	x	x		x	x	17
Water effluent charges	x		x	x			x	x						x		x	x					x	9
Water quality fee																						x	1
Waste Disposal:																							
Municipal waste user charges	x	x	x	x	x	x	x	x		x	x	x		x	x	x	x	x	x	x	x	x	20
Tax on waste disposal	x							x														x	3
Hazardous waste charge	x																					x	2
Total:	5	1	3	10	2	2	3	4	0	2	2	1	0	3	2	4	3	2	2	1	2	11	

Terminology: "regis" = registration
Sources: OECD (1994a) and OECD (1995a)

Sewage management (Table 48) occurs at the local level. Local governments charge households and firms for water usage and pollution load. The municipalities that charge per-load as opposed to a flat rate experience higher costs. The per-unit charge operates as an efficient tax since it represents a payment for a service. Sewage is a form of waste that most countries isolate and process. The goal is to restrict any effects on human health and the environment. The user charges help pay for this waste processing.

The effluent charges on households and firms in Table 49 are independent of any wastewater treatment. Taxes are appropriate policies with water effluent. In all of the countries, governments use the revenue for water quality and/or subsidies for pollution abatement. Two provinces in Canada levy a water effluent charge on the basis of pollution load. The rates vary according to the toxicity levels of the different pollutants. In Belgium, local governments levy the fee on pollution equivalents. The French tax firms that exceed a certain number of "inhabitant equivalents." This measurement includes a standardized menu of nitrogen, phosphates, suspended matter, toxic materials, and dissolved salts (OECD, 1994a).

The water pollution taxes in France and the Netherlands help to enhance environmental quality. In France, for example, six river-basin agencies operate five-year plans for industries that dump pollution into the waterways. These plans identify safe levels of water pollution and the establish coefficients that determine payments. By levying per-unit charges, the government raises a significant amount of revenue. However, the representatives of industry make sure that the revenues return as subsidies for abatement technology. In fact, the subsidy scheme was the factor that convinced companies to agree to the pollution levies in the first place. The water pollution taxes have improved water quality in France (Andersen, 1994).

Table 48
User charges for sewage and sewage treatment

Country	Charge	Method of Measurement
Australia	H: Water usage	FR/AM
	F: Water usage, pollution load	--
Belgium	H: Water usage	FR/AM
	F: --	--
Canada	H: Water usage	FR/AM
	F: Water usage, pollution load	FR/AM
Denmark	H: Water usage	FR
	F: Water usage, excess pollution load	AM
Finland	H: Water usage	FR/AM
	F: Water usage, excess pollution load	AM
France	H: Water usage	--
	F: Water usage	--
Germany	H: Water usage	--
	F: Water usage, pollution load	FR/AM
Ireland	--	--
Italy	H: Water usage	--
	F: --	--
Netherlands	H: Flat rate	FR
	F: --	--
Norway	H: Water usage	FR
	F: Water usage, excess pollution load	FR/AM
Portugal	H: Water usage	FR
	F: Water usage	FR
Spain	H: Water usage	AM
	F: Water usage	AM
Sweden	H: Water usage	FR/AM
	F: Water usage, excess pollution load	AM
Switzerland	H: Water usage	FR/AM
	F: Water usage	FR/AM
UK	H: Water usage	FR/AM
	F: Water usage, pollution load	
USA	H: Water usage	--
	F: Water usage, pollution load	FR/AM

Symbols:
H = households
F = firms
FR = flat rate
AM = actual measurement

Table 49
Water effluent charges

Country	Charge base	Incentive: intended	Incentive: actual	Measurement method
Australia	F: Pollution load	+	#	AM
Belgium	H: Pollution load	+	#	AM
	F: Pollution load	..	..	FR/AM
Canada	F: Pollution load	+	#	AM
France	F: Pollution load	+	+	AM
Germany	F: Pollution load related to discharge standard	+	#	AM
Netherlands	F: Pollution load	+	+	AM
Portugal	H: Pollution load	..	..	FR
	F: Pollution load	..	..	--
Spain	F: Pollution load	..	..	AM
USA	F: Pollution load	..	..	..

Symbols:
+ = yes
- = no
.. = no data available
# = unclear

In the Netherlands, moreover, public authorities levy significant charges on the discharge of organic pollutants in the waterways and make it profitable for the companies to reduce emissions. The Dutch do not use a pure levy system. They earmark the revenues to finance public sewage plants and the industrial control of discharges. This revenue policy aids companies in the implementation of pollution abatement technology and helps local communities with sewage processing. Like in France, the Netherlands' pollution charges help to enhance water quality (Andersen, 1994).

The household user charges in Table 50 represent payments for a service. While most are flat rates, eight country municipalities have experimented with charges per volume or weight and some report incentive responses.

Table 50
Municipal waste user charges

Country	Charge base and rate	Revenue use
Australia	H: FR(AM) F: FR(AM)	Di
Austria	..	..
Belgium	H: FR(AM) F: ..	Co
Canada	H: FR F: AM	Co + Di
Denmark	H: FR(AM) F: AM	Co + Di
Finland	H: AM F: AM	Co + Di
France	H: FR(AM) F: ..	..
Germany	H: FR F: AM	Co + Di
Ireland	..	..
Italy	H: FR F: ..	..
Japan	H:FR(AM) F: ..	Co + Di
Netherlands	H: FR(AM) F: ..	Co + Di
Norway	H: FR, ECU 44-219/year F: AM, ECU 11-44/ton	Co + Di
Spain	H: FR F: FR	Co + Di
Sweden	H: FR, ECU 107/year F: FR(AM)	Co + Di
Switzerland	H: FR(AM) F: FR(AM)	Co + Di
UK	H: - F: AM	Co + Di
USA	H: FR(AM) F: ..	Co + Di

Symbol: FR(AM) = FR in most municipalities, AM in some
Note: Flat rates often depend on household size.
Co = collection
Di = disposal

Appendix 2

The Lagrangian function is formed as follows:

$$L = U^a(X^a, Y^a, Z) + \lambda_1(U^b(X^b, Y^b, Z) - U^b) + \lambda_2(f^X(K^X, L^X, S^X) - X^a - X^b) + \lambda_3(f^Y(K^Y, L^Y, S^Y) - Y^a - Y^b) + \lambda_4(L - L^X - L^Y) + \lambda_5(K - K^X - K^Y)$$

The choice variables are X^a, X^b, Y^a, Y^b, L^X, L^Y, K^X, K^Y, S^X, S^Y.

The following are the first-order maximization conditions. For example, L_{Xa} means that the Lagrangian function is differentiated with respect to X^a.

1. $L_{Xa} = dU^a/dX^a - \lambda_2 = 0$
2. $L_{Xb} = \lambda_1\, dU^b/dX^b - \lambda_2 = 0$
3. $L_{Ya} = dU^a/dY^a - \lambda_3 = 0$
4. $L_{Yb} = \lambda_1\, df^X/dL^X - \lambda_3 = 0$
5. $L_{LX} = \lambda_2\, df^X/dL^X - \lambda_4 = 0$
6. $L_{LY} = \lambda_3\, df^Y/dL^Y - \lambda_4 = 0$
7. $L_{KX} = \lambda_2\, df^X/dK^X - \lambda_5 = 0$
8. $L_{KY} = \lambda_3\, df^Y/dK^Y - \lambda_5 = 0$
9. $L_{SX} = (dU^a/dZ)(dZ/dS^X) + \lambda_1\,(dU^b/dZ)(dZ/dS^X) + \lambda_2\, df^X/dS^X = 0$
10. $L_{SY} = (dU^a/dZ)(dZ/dS^Y) + \lambda_1\,(dU^b/dZ)(dZ/dS^Y) + \lambda_3\, df^Y/dS^Y = 0$

Recall in equations nine and ten that dZ/dS^X and dZ/dS^Y equal one.

To derive the first Pareto efficiency condition, we substitute equation 1 into 2 and 3 into 4. Since λ_1 is the same in each of the remaining equations, we have the following:

$$(dU^a/dU^X)/(dU^a/dU^Y) = (dU^b/dU^X)/(dU^b/dU^Y)$$

To derive the second Pareto efficiency condition, we set equation 5 equal to 6 and equation 7 equal to 8. Substituting the remaining equations into each other results in the following:

$$(df^X/dL^X)/(df^X/dK^X) = (df^Y/dL^Y)/(df^Y/dK^Y)$$

The derivation of the third efficiency condition uses all ten first-order conditions. First, (1) is set equal to (2) and put into (9):

$$dU^a/dZ + \frac{(dU^a/dX)(dU^b/dZ)}{dU^b/dX} = -\lambda_2 df^X/dS^X$$

This equation is divided by dU^a/dX:

$$(dU^a/dZ)/(dU^a/dX) + (dU^b/dZ)/(dU^b/dX) = -\frac{\lambda_2 (df^X/dS^X)}{dU^a/dX}$$

Since from (1) $\lambda_2 = dU^a/dX$:

$$-\frac{(dU^a/dZ)}{(dU^a/dX)} - \frac{(dU^b/dZ)}{(dU^b/dX)} = df^X/dS^X$$

Second, set (3) equal to (4) and put into (10):

$$dU^a/dZ + \frac{(dU^a/dY^a)(dU^b/dZ)}{dU^b/dY^b} = -\lambda_3 df^Y/dS^Y$$

This equation is then divided by dU^a/dY^a:

$$\frac{dU^a/dZ}{dU^a/dY^a} + \frac{dU^b/dZ}{dU^b/dY^b} = \frac{-\lambda_3 df^Y/dS^Y}{dU^a/dY^a}$$

Since from (3) $\lambda_3 = dU^a/dY^a$:

$$-\frac{dU^a/dZ}{dU^a/dY^a} - \frac{dU^b/dZ}{dU^b/dY^b} = df^Y/dS^Y$$

This leaves the third Pareto efficiency condition:

$-\Sigma MRS_{ZX} = df^X/dS^X$ for good X. (The same holds true for Y.)

The externality problem occurs in this example as individuals and firms do not operate under the jurisdiction of environmental regulation. In other words, a pollution price does not exist. Specifically, in a decentralized system, the individual maximizes utility subject to income I:

$$\max U(X, Y, Z) + \lambda(I - P_X X - P_Y Y)$$

The choice variables are X and Y. The first-order conditions reveal that the *first efficiency condition* is satisfied:

$$U_X - \lambda P_X = 0$$

$$U_Y - \lambda P_Y = 0$$

The MRS of X for Y equals the price ratio (which in turn equals the MRS for the other person.)

The firm maximizes profits minus costs in a competitive setting, but does not consider the pollution externality imposed on society:

$$\max P_X f^X(K^X, L^X, S^X) - W_K K^X - W_L L^X$$

The choice variables are K^X, L^X, and S^X. The first-order conditions reveal that the *second efficiency condition* is satisfied:

$$P_X \, df^X/dK^X - W_K = 0$$

$$P_X \, df^X/dL^X - W_L = 0$$

So the MRTS of K for L equals the price ratio of these factor inputs (which in turn equals the MRTS of K for L for the other good.)

However, because the firm does not account for the externality imposed on society, the *third efficiency condition* does not hold:

$$df^X/dS^X = 0.$$

Here the government must intervene and implement a per-unit emissions tax on all firms that pollute. For the individual firm the maximization equation becomes:

$$\max Pf(K^X, L^X, S^X) - tS^X - W_K K^X - W_L L^X$$

where t is the per-unit emissions tax. This environmental tax leads to the satisfaction of the *third efficiency condition*:

$$df^X/dS^X = t.$$

What should t equal? The per-unit tax should equal $-\Sigma MRS_{ZX}$ and the level of marginal damage at the optimal level of emissions. The emissions tax internalizes the pollution externality at the optimal level of emissions.

The optimal level of emissions (E_o) occurs where the marginal damage function intersects the marginal abatement cost function (Figure 10).[95] Damages (D) and costs (C) are measured on the vertical axis. Damages increase as emissions increase.[96] (The marginal damage function [MD] does not have to be linear.) The marginal abatement cost function (MAC) increases as emissions decrease. (It is more expensive for firms to reduce additional emissions.) To the right of E_o, marginal damages exceed marginal abatement costs. In this area, society would benefit from a *reduction* in emissions since the costs of abating a unit of emissions are less than the damages from the same unit. To the left of E_o, marginal abatement costs exceed marginal damages. In this area, society would benefit from an *increase* in emissions because the costs of abating one unit of emissions are greater than the marginal damages. If the MAC and MD functions are known, the optimal Pigouvian tax equals t_o. Of course, policy makers cannot establish the optimal level of emissions because of information limitations and uncertainty with respect to the MAC and MD functions (Adar and Griffin, 1976). As a result, policy makers levy emissions taxes to establish a target level of intended behavior or raise revenue.

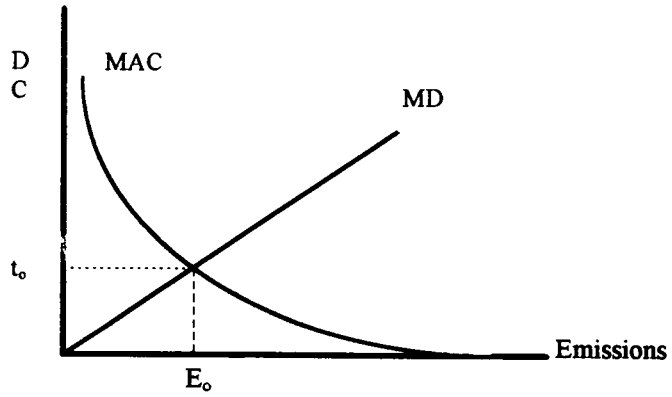

Figure 10
The optimal level of emissions

[95] This figure and discussion on the optimal level of emissions draws on the graph and presentation in Kahn (1995, 58).

[96] Different shapes of the marginal damage function are discussed in chapter three along with the factors that impact the shapes.

Appendix 3

Tables 51 and 52 present the toxicological endpoints in the algorithm. The descriptions of the endpoints following the tables are from page ten of Davis *et al.* (1994c).

Table 51
Toxicological endpoints

Type of Effect	Type of Toxicity	Toxicological Endpoint
Human Health Effects	Acute	Inhalation LC 50 (a)
	Acute	Oral LD 50 (b)
	Chronic	Carcinogenicity (c)
	Chronic	Other Specific Effects (d)
Environmental Effects		
Terrestrial Animals	Acute	Oral LD 50 (b)
Fish	Acute	LC 50 (e)
	Chronic	NOEL (f)

(a) The concentration of a substance in air that will kill half of a group of rodents when inhaled continuously for a specific period of time. Data from tests of eight hours or less were used; these data were scaled on a linear basis to be equivalent to a four hour test by: LC_{50}@4hr = (LC_{50}@ t hrs) x (t hrs)/4 hrs. (The term LC stands for lethal concentration.)

(b) The concentration of a substance that will kill half of a group of rodents within 14 days when administered orally as a single dose. (Dose is expressed as mass of chemical per mass of animal body weight. The term LD stands for lethal dose.)

(c) Based on the EPA or International Agency for Research on Cancer (IARC) weight-of-evidence classification.

(d) Includes positive evidence of mutagenicity, developmental effects, reproductive effects, other chronic effects and neurotoxicity.

(e) The concentration of a chemical in water that causes death in fifty percent of the fish in a ninety-six-hour test.

(f) No observable effect level (NOEL): the highest dosage administered that does not produce toxic effects, estimated from LC_{50} data.

Table 52
Exposure parameters

Persistence	Bioaccumulation
Biological oxygen demand (BOD) half-life (a)	Aquatic bioconcentration factor (BCF) (b)
Hydrolysis half-life (c)	

(a) The number of days required to biodegrade a chemical such that its BOD in water is reduced by half, as predicted using QSAR.

(b) The ratio of the concentration of a chemical in an aquatic organism to its concentration in water.

(c) The number of days required for the amount of a chemical to be reduced by half through hydrolysis in water, at pH 7, as predicted using QSAR.

Appendix 4

Table 53

Revenue from a risk-based tax and simple emissions tax

Tax Rate/RT	Chemical	# Firms	Pounds	Hazard Sum		RU	Revenue (RT)	Revenue (ET)
0.025	cadmium compounds	1		187.06	100	3.23	0	0
	arsenic compounds	4		153.9	82	2.65	0	0
Total Revenue RT	terbufos	0		151.6	81	2.61	0	0
2238947	hexachloro-1,3-butadiene	0		145.6	78	2.52	0	0
	polychlorinated biphenyls	0		133.1	71	2.29	0	0
% Revenue from	trifluralin	0		118.1	63	2.03	0	0
20 Most Toxic	hexachlorobenzene	1		116.1	62	2	0	0
0.07	trichlorobenzene	1		115.7	62	2	0	0
	chromium compounds	40		114	61	1.97	0	0
Tennessee 1994	2-nitropropane	0		113	60	1.94	0	0
	formaldehyde	12	171539	111.9	60	1.94	8300	4288.475
Tax Rate/ET	cobalt compounds	6	71555	111.3	59	1.9	3405	1788.875
0.025	lead compounds	29	46787	111.1	59	1.9	2226	1169.675
	nickel compounds	34	8160	110.9	59	1.9	388	204
Total Revenue ET	anthracene	1	500	108.3	58	1.87	23	12.5
2890590	diaminotoluene	0	0	107.4	57	1.84	0	0
	hydrogen fluoride	6	271605	103.4	55	1.77	12047	6790.125
% Revenue from	di phthalate	15	19994	102.9	55	1.77	887	499.85
20 Most Toxic	chlorothalonia	0	0	99.6	53	1.71	0	0
0.03	24-D	1	255	98.5	53	1.71	11	6.375
	2,4-dinitrophenol	1	1820	97.6	53	1.71	78	45.5
	1,3-dichloropropene	0	0	97.6	52	1.68	0	0
	epichlorohydrin	1	703	96.9	52	1.68	29	17.575
	decabromodiphenyl oxide	5	4014	96.5	51	1.65	165	100.35
	biphenyl	6	169937	96.5	51	1.65	6989	4248.425
	copper compounds	44	34716	96.4	51	1.65	1428	867.9
	hydrogen cyanide	3	71046	96	51	1.65	2922	1776.15
	styrene	44	2831122	93.6	50	1.61	114158	70778.05
	dibutyl phthalate	3	8211	93.1	50	1.61	331	205.275
	2,4-dinitrotoluene	0	0	92.8	49	1.58	0	0
	1,3-butadiene	2	2350	92.8	49	1.58	93	58.75
	naphthalene	8	35658	87.9	47	1.52	1352	891.45
	tetrachloroethylene	9	232747	86.4	46	1.48	9234	5818.675
	vinyl chloride	0	0	85.5	46	1.48	0	0
	chlorpyrifos	0	0	83.7	45	1.45	0	0
	chlorprene	1	3997	83	44	1.42	142	99.925
	n-nitrosodiphenylamine	0	0	82.6	44	1.42	0	0
	acrylamide	3	2096	81.9	44	1.41	74	52.4
	112-trichloroethane	31	1826452	81.8	44	1.41	64963	45661.3
	4-nitrophenol	0	0	81.5	44	1.4	0	0
	nitric acid	26	60582	81	43	1.4	2114	1514.55
	hexachloroethane	0	0	80.3	43	1.38	0	0
	titanium tetrachloride	1	950	79	42	1.36	32	23.75
	captan	0	0	79	42	1.36	0	0

Table 53 (continued)

	trichloroethylene	14	923715	78.7	42	1.36	31317	23092.875
	4,4'-methylenedianiline	0	0	78.6	42	1.35	0	0
	sulfuric acid	111	4,501,576	76.6	41	1.32	146548	112539.4
	1,2-dichlorobenzene	0	0	75.8	40	1.31	0	0
	ammonia	60	2,639,420	74.5	40	1.28	84711	65985.5
	dichlorobenzene	1	215	73.1	39	1.26	7	5.375
	vinylidene chloride	0	0	72.7	39	1.25	0	0
	methyl parathion	0	0	72	38	1.24	0	0
	chlorine	51	4342430	72	38	1.24	134691	108560.75
	ethylene oxide	4	17851	71.7	38	1.24	551	446.275
	cumene	4	18130	71	38	1.22	555	453.25
	1,2-dichloroethane	1	3	70.7	38	1.22	0	0.075
	cyclohexane	6	71079	70.2	38	1.21	2150	1776.975
	bis adipate	10	126843	68.8	37	1.19	3759	3171.075
	phenol	19	243126	67.3	36	1.16	7049	6078.15
	di-n-octyl phthalate	0	0	67.1	36	1.16	0	0
	phosphoric acid	73	2436124	66.4	35	1.14	69685	60903.1
	malathion	0	0	66.3	35	1.14	0	0
	cresol	5	29324	66.2	35	1.14	836	733.1
	phosphorus	6	6829	66	35	1.14	194	170.725
	acrylonitrile	6	3857	66	35	1.14	110	96.425
	p-cresol	1	1866	65.8	35	1.13	53	46.65
	carbon tetrachloride	2		65.3	35	1.13	0	0
	44-isopropylidenediphenol	2	265	65.1	35	1.12	7	6.625
	chlorophenols	0	0	64.9	35	1.12	0	0
	chloroform	2	92888	64.8	35	1.12	2593	2322.2
	benzene	9	132237	64.5	34	1.11	3674	3305.925
	1,4-dichlorobenzene	0	0	62.3	33	1.07	0	0
	p-xylene	3	276717	62	33	1.07	7391	6917.925
	hydrochloric acid	75	5372033	61.4	33	1.06	142095	134300.825
	chlorine dioxide	2	13406	61	33	1.05	352	335.15
	barium compounds	18	122392	60.6	32	1.04	3195	3059.8
	atrazine	0	0	59.5	32	1.03	0	0
	toluene-2,4-diisocyanate	13	13482	58.9	31	1.01	342	337.05
	chlorobenzene	2	8890	57.8	31	1	221	222.25
	cyanazine	0	0	57.2	31	0.99	0	0
	cumene hydroperoxide	2	17588	57	30	0.98	432	439.7
	n,n-dimethylaniline	2	8402	56.6	30	0.98	205	210.05
	picric acid	0	0	56.3	30	0.97	0	0
	molybdenum trioxide	0	0	55.7	30	0.96	0	0
	manganese compounds	35	4930380	55.7	30	0.96	118306	123259.5
	bromomethane	3	264940	54.9	29	0.95	6266	6623.5
	ethylbenzene	22	357574	54.1	29	0.93	8334	8939.35
	metan sodium	0	0	53.4	29	0.92	0	0
	1,2,4 trimethylbenzene	10	426341	53.4	29	0.92	9808	10658.525
	methylenebis	44	2130002	52.6	28	0.91	48266	53250.05

Table 53 (continued)

	aniline	1	5565	51.9	28	0.89	124	139.125
	xylene	131	6432133	51.4	27	0.89	142426	160803.325
	acetaldehyde	7	549661	50	27	0.86	11840	13741.525
	alachlor	0	0	49.6	26	0.85	0	0
	catechol	2	83	49.6	26	0.85	2	2.075
	propionaldehyde	1	112009	49.4	26	0.85	2384	2800.225
	o-xylene	4	173802	48.7	26	0.84	3646	4345.05
	carbaryl	0	0	47.6	25	0.82	0	0
	propylene oxide	1	8	46.9	25	0.81	0	0.2
	m-xylene	4	34805	46.4	25	0.8	698	872.625
	1,1,1-trichloroethane	32	1158652	45.6	24	0.79	22761	28966.3
	ethyl dipropylthiocarbamate	0	0	45.1	24	0.78	0	0
	zinc compounds	83	467559	44.7	24	0.77	9004	11688.975
	acrylic acid	5	14666	43.8	23	0.75	277	366.65
	butyl benzyl phthalate	0	0	41.9	22	0.72	0	0
	allyl chloride	1	1207	41.5	22	0.72	22	30.175
	maneb	0	0	41.3	22	0.71	0	0
	carbon disulfide	5	27895400	40.2	21	0.69	483093	697385
	butylate	0	0	39.4	21	0.68	0	0
	chloroethane	1	346	38.6	21	0.67	6	8.65
	benzoyl chloride	1	533	37	20	0.64	8	13.325
	metolachlor	0	0	36.5	19	0.63	0	0
	dichloromethane	33	2709658	36.5	19	0.63	42607	67741.45
	asbestos	3		36.4	19	0.63	0	0
	toluene	139	14632031	36	19	0.62	226924	365800.775
	isopropyl alcohol	3	141779	36	19	0.62	2199	3544.475
	diethyl phthalate	2	18680	35.5	19	0.61	286	467
	nitrobenzene	0	0	35	19	0.6	0	0
	pyridine	1	870	34.7	19	0.6	13	21.75
	hydroquinone	2	17943	34	18	0.59	263	448.575
	propylene	5	12930	33.4	18	0.58	186	323.25
	terephthalic acid	0	0	32.7	17	0.56	0	0
	ethylene	43	2498652	31.4	17	0.54	33799	62466.3
	2-methoxyethanol	2	20069	31.3	17	0.54	271	501.725
	maleic anhydride	8	4370	30.2	16	0.52	57	109.25
	zinc	96	5732167	30	16	0.52	74082	143304.175
	vinyl acetate	3	1435	29.4	16	0.51	18	35.875
	isobutyraldehyde	1	27300	28.6	15	0.49	396	682.5
	dimethyl phthalate	5	6145	28.4	15	0.49	75	153.625
	chloromethane	3	33069	27.9	15	0.48	397	826.725
	2-ethoxyethanol	1	500	27.7	15	0.48	6	12.5
	phthalic anhydride	3	630	27.6	15	0.48	7	15.75
	1,4-dioxane	2	126900	27.4	15	0.47	1498	3172.5
	diethanolamine	10	9811	25.9	14	0.45	109	245.275
	butyraldehyde	1	69130	25.9	14	0.45	771	1728.25

Table 53 (continued)

	methyl ethyl ketone	70	3,974,032	24.2	13	0.42	41430	99350.8
	glycol ethers	78	2,871,548	22.6	12	0.39	27957	71788.7
	methyl methacrylate	11	606,272	21.6	12	0.37	5641	15156.8
	12-dichloropropane	0	0	20	11	0.34	0	0
	methyl isobutyl ketone	35	950,320	19.8	11	0.34	8106	0
	n-butyl alcohol	29	686932	18.4	9.8	0.32	5445	17173.3
	metribuzin	0	0	18	9.6	0.31	0	0
	antimony compounds	14	20753	17.4	9.3	0.3	156	518.825
	aluminum	14	89984	15.6	8.3	0.27	605	2249.6
	ethylene glycol	35	910779	14.3	7.6	0.25	5611	22769.475
	thorium dioxide	0	0	20	11	0.34	0	0
	ammonium nitrate	5	732	11.9	6.4	0.21	4	18.3
	freon 113	0	0	10.9	5.8	0.19	0	0
	acetonitrile	2	170202	10.4	5.6	0.18	763	4255.05
	acetone	0	0	8.1	4.3	0.14	0	0
	carbonyl sulfide	2	2075900	6	3.2	0.1	5366	51897.5
	methanol	61	4529372	4.8	2.6	0.08	9366	113234.3
	ammonium sulfate	5	1201500	4.4	2.4	0.08	2277	30037.5
	tert-butyl alcohol	5	16409	4.3	2.3	0.07	30	410.225
	methyl tert-butyl ether	4	10740	3	1.6	0.05	14	268.5
	glyphosate	2	70569	1.9	1	0.03	58	1764.225
	sec-butyl alcohol	2	70569	0.9	0.5	0.02	27	1764.225
	Total						2238947	2890590.3

Appendix 5

Table 54
Tennessee tax collections in 1994 (millions)

Tax Source	1994	Tax Source	1994
Sales and Use	2923.9	Special Petroleum	51.0
Gasoline	504.0	Inheritance & Gift	42.9
Excise	407.3	Alcoholic Beverage	26.3
Franchise	238.1	Mixed Drink	23.3
Gross Receipts	161.3	Beer	13.5
Motor Vehicle Regis	159.5	Business	11.8
Privilege	122.7	Motor Vehicle Title	9.1
Motor Fuel	105.7	Severance	0.9
Income	96.6	Coin Amusement	0.3
Tobacco	77.4	Filing Fees	0.0

Source: Center for Business and Economic Research,
The University of Tennessee

VITA

Thomas Robert Sadler was born on August 7, 1969 in Evanston, Illinois. He attended public schools in Macomb, Illinois, and graduated in 1987 from Macomb Senior High School. He attended Western Illinois University from August, 1987 to May, 1991, where he earned a Bachelor of Arts degree in economics, summa cum laude. Having enrolled at the University of Missouri in August of 1991, he graduated with a Master of Arts degree in economics in May of 1993. While taking classes at the University of Missouri, he taught principles-of-economics discussion sections. He wrote a thesis on transition economies in Eastern Europe and the former Soviet Union. The author entered the University of Tennessee in August, 1993 and earned a Ph.D. in economics in August, 1998. During the Ph.D. program, he taught undergraduate classes in introductory and intermediate economics and worked as a research assistant at the Energy, Environment, and Resources Center.

The author is currently an Assistant Professor of Economics at Pace University in New York City, where he has been teaching environmental economics, international economics and public finance since September, 1997. Along with his wife, Holly Ann Stovall, he enjoys taking walking tours of Manhattan, visiting art museums, biking in Central Park and listening to live music.