Calculating the Costs of Environmental Regulation

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Abstract

Decisions concerning environmental protection hinge on estimates of economic burden. Over the past 30 years, economists have developed and applied various tools to measure this burden. In this paper, developed as a chapter for the *Handbook of Environmental Economics*, we present a taxonomy of costs along with methods for measuring those costs. At the broadest level, we distinguish between partial and general equilibrium costs. Partial equilibrium costs represent the burden directly borne by the regulated entity (firms, households, government), including both pecuniary and nonpecuniary expenses, when prices are held constant. General equilibrium costs reflect the net burden once all good and factor markets have equilibrated. In addition to partial equlibrium costs, these general equilibrium costs include welfare losses or gains in markets with preexisting distortions, welfare losses or gains from rebalancing the government's budget constraint, and welfare gains from the added flexibilty of meeting pollution constraints through reductions in the use of higher-priced, pollution-intensive products. In addition to both partial and general equilibrium costs, we also consider the distribution of costs across households, countries, sectors, subnational regions, and generations. Despite improvements in our understanding of cost measurement, we find considerable opportunity for further work and, especially, better application of existing methods.

Key Words: social cost, cost-benefit, cost-effectiveness, environmental regulation

JEL Classification Numbers: Q20, Q28, H41, L50, D58

Contents

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1. Introduction

The real and perceived economic costs associated with environmental protection are easily the greatest obstacles to cleaner air and water, improved preservation of ecosystems and biodiversity, and slower depletion of natural resources. Over the past 30 years, considerable effort has been directed at quantifying these costs and improving measurement methods. Aggregate estimates for the United States suggest that roughly 2% of gross domestic product (GDP) is spent on environmental protection.[1] Data for other countries are less comprehensive but suggest similar levels of expense.[2] More important than these aggregate cost estimates—which imply a decision whether to protect the environment or not—are increasingly frequent and detailed studies of the cost of specific initiatives.[3]

Despite such efforts, the accurate measurement of costs remains challenging. This is true conceptually—in terms of defining what we include as costs—and especially in practice, where studies use very different methodological approaches to estimate these costs.

Our goal is to create a taxonomy of the different methods and theories of cost measurement, to show how they relate to one another, and to explain how various empirical exercises fit in. Imagine, for a moment, that a policymaker, analyst, or citizen asks what we must give up in order to protect an endangered species, remedy a polluted area, or prevent future

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[1] See U.S. EPA (1990a) and Rutledge and Vogan (1994)

[2] See Table 13.1 in OECD (1999) and more recent estimates available on the OECD web-site http://www.oecd.org/ ("Selected Environmental Data"). The OECD data suggest that expenditures range from 0.6% to 2.0% of GDP among OECD countries.

[3] For example, see analysis by both EPA and the Energy Information Administration presented during hearings before the U.S. Senate (*Consideration of S. 556, Clean Power Act* 2001). Alternatively, see a recent survey of cost analyses by Harrington et al. (2000).

climate change. The answer first depends on the consequences we ascribe to an environmental action—direct compliance costs, forgone opportunities, lost flexibility, etc. The answer also depends on whether the policy will meaningfully influence the price of goods and services, in which case it will be necessary to consider the possible welfare consequences due to preexisting distortions in other (primarily factor) markets, government behavior, and changes in the terms of trade. Finally, the answer depends on whether concern exists about the "we" taken as a whole, or about all the pieces "we" comprise. Our chapter focuses on each of these questions in turn: partial equilibrium costs, general equilibrium effects and broad economic costs, and the distribution of burden.

The object of our analysis will be one or more relationships of the form

$$\text{costs} = C_i\left(\mathbf{a},\mathbf{z}\right), \tag{1}$$

where **a** is a parameter or vector of parameters describing an environmental policy, **z** is a vector of parameters summarizing the current economic equilibrium, and $C_i\left(\mathbf{a},\mathbf{z}\right)$ is the associated cost borne by agent *i*. The vector **z** should be viewed as the economic features agent *i* is likely to assume are fixed in doing a cost calculation—features such as input prices and output level for a business, or prices and income for a consumer.[4] Once computed, this cost $C_i\left(\mathbf{a},\mathbf{z}\right)$ might then be compared with the cost of an alternative policy, $C_i\left(\mathbf{a}',\mathbf{z}\right)$, compared with the cost of no policy, $C_i\left(0,\mathbf{z}\right)$, compared with the cost $C_j\left(\mathbf{a},\mathbf{z}\right)$ borne by another agent *j*, compared with an estimate of benefits, or used to compute a marginal cost $\partial C_i\left(\mathbf{a},\mathbf{z}\right)/\partial \mathbf{a}$ and then compared with marginal benefits.

In each of these cases, we can distinguish between partial and general equilibrium costs, and among costs borne by various agents, by considering both the endogeneity of the vector **z** and the enumeration of agents *i*. When **z** is held fixed and *i* is limited to the directly regulated agent(s), we ignore price and activity changes in other markets and measure only the direct compliance costs in a partial equilibrium. By instead considering the relationship **z**(**a**), we can evaluate costs when policy effects are transmitted to other markets and the economy equilibrates. By summing costs across agents that compose final demand, we arrive at a measure of the total

[4] We are necessarily vague about **z** at this point because the environmental policy necessarily disrupts the economic equilibrium. Holding **z** fixed, as described below, is meant to capture the notion of a partial equilibrium evaluation.

cost to the economy.[5] This is the notion of cost most often described by economists (e.g., Harberger 1971; Diamond and Mirrlees 1971).

Defining and estimating $C_i(\mathbf{a}, \mathbf{z})$ is far from trivial. It may be difficult for agents to appreciate fully the costs associated with **a**, such as the opportunity cost of borrowing employees from other tasks to focus on pollution control, or of building space for house abatement equipment. It also requires sophisticated and complete models of the economy to elucidate the relation $\mathbf{z}(\mathbf{a})$ as well as transmission of costs to agents not directly affected by environmental policy (owners of capital, product consumers, factor suppliers).[6] Addressing these questions, we have divided our discussion into three parts. In the next section we consider the various consequences of environmental protection for those who are directly regulated. This analysis typically holds the behavior in the remainder of the economy fixed and is therefore considered partial equilibrium analysis. In Section 3, we consider broad economic costs, taking into account the general equilibrium effects outside the market where environmental protection occurs and focusing on final demand. In Section 4, we consider in more detail the question of who bears the cost.

2. Environmental Protection Costs and Consequences: Partial Equilibrium

Popular debate over environmental protection often centers on the out-of-pocket expenditures or other negative consequences[7] directly associated with pollution reduction paid by firms, governments, and households. In this section we examine various ways these costs are defined and estimated in the economic literature, organizing the discussion around the affected agent and type of cost, and later returning to the common issues of uncertainty and discounting. For the moment, we ignore how good and factor prices are likely to adjust, both creating new welfare effects in other markets and shifting the burden to different agents—that is, we are

[5] Here we emphasize that total costs involve only components of final demand because costs to businesses are passed on to consumers in the form of higher prices or reduced factor (capital-labor) income. In an open economy, this will involve costs potentially borne by the foreign sector (McKibbin et al. 1999).

[6] For example, U.S. EPA analysis of recently proposed power plant regulation estimated costs of \$3.1 billion to \$6.9 billion without completely describing who bears these costs: power plant owners, factor suppliers (e.g., coal companies), or particular end-users (2001).

[7] Such consequences, including effects on jobs (Hahn and Steger 1990; Rosewicz 1990) or international competitiveness (Jaffe et al. 1995), can be converted to pecuniary effects. Other, more indirect costs are discussed below.

implicitly following a partial equilibrium approach and ignoring actual incidence. We return to these important issues in Sections 3 and 4, respectively.

The overwhelming focal point of the literature to date has been measurement of the direct compliance cost to firms. Given sufficient knowledge of production technologies, obtained through engineering studies or revealed market behavior, the cost of environmental protection described by (1) is a relatively straightforward calculation for firms. Studies of government compliance costs, while rarer in the literature, are similarly straightforward thanks to detailed budget documentation. Analyses of regulatory impacts on households are rarer still—and fraught with difficulty. It is hard to identify, let alone measure, the "technology" for pollution control activities and the immediate cost to households of environmental regulations.

2.1. Direct Compliance Costs

Environmental policies applied to firms either directly force changes in production methods (command-and-control policies) or provide incentives to do so by changing prices (market-based policies). The direct compliance cost is the change in production costs entailed by the policy. This cost will depend on the particular technological alternatives available to the firm.

Techniques for modeling and estimating production technology are well described in the economic literature (Caves and Christensen 1980; Färe 1988; Jorgenson 1986). Engineering models of pollution and abatement also exist (e.g., U.S. EPA 1990, 1985). Finally, surveys of business and governments can be used to provide direct estimates of pollution control costs (McGraw-Hill various years; Bureau of the Census 1973-1997; OECD 1999). All of these methods have been heavily used—and criticized—in the analysis of environmental protection costs.

A convenient way to understand the roles of these various costing techniques is to consider the channels through which pollution can be reduced and the data necessary to quantify relevant abatement costs. In some cases, pollution is associated with a particular input, and substitution away from that input will reduce pollution. In other cases, pollution reduction arises from changing the production process or installing "end-of-pipe" equipment to capture pollutants before they escape.[8] When input substitution is the primary mechanism for reducing emissions,

[8] End-of-pipe treatment often converts one type of pollution (air) into another (solid), raising the need for integrated pollution control policies. See Davies (2001)

or when process changes or end-of-pipe treatment has already occurred and the study is retrospective, historical data can be used to estimate relationships between observed or estimated emissions and production costs. However, absent relevant historical experience with pollution control or input substitution, econometrically estimable production models are unlikely to provide useful information about abatement costs.

In the case of prospective cost analysis where input substitution is not the primary means of pollution control, often the only way to estimate costs is to pose the question to engineers familiar with abatement technology. Early studies of specific environmental regulation followed this approach. These efforts, including Atkinson and Lewis (1974), Seskin, et al. (1983), Perl and Dunbar (1982), and Krupnick (1986), estimate the cost of alternative policies based on linear programming models of firm response using specific technology options enumerated by engineering experts. This approach is also used by the U.S. Environmental Protection Agency (EPA) in its regulatory impact analyses of proposed regulations (e.g., U.S. EPA 1985, 1992). Through 1979, the president's Council on Environmental Quality used engineering estimates to forecast aggregate pollution control costs in the annual publication *Environmental Quality*. Discussions of these types of analyses can be found in Tietenberg (1992) and more recently in Morgenstern (1997).

Although an engineering approach is most useful when tailored to the specific characteristics of an individual plant, it can be problematic when applied on a broad scale. This often involves estimates based on a "typical" plant that are then extrapolated to the entire industry. Technologies differ across plants, as do factor costs and even local (state and municipal) environmental regulations (CBO 1985). For this reason there has been some concern about the accuracy of this approach when applied to broad regulatory initiatives.

More recently, there has been a large volume of work on the cost of reducing carbon dioxide emissions to mitigate the threat of climate change. Like the early analyses of abatement in the 1970s and 1980s, this work is prospective. Unlike these analyses, however, abatement is linked directly to reduced use of specific polluting inputs: fossil fuels, including coal, oil, and natural gas. This has allowed researchers to make use of historical information about fuel substitution to estimate the cost of emissions reductions, rather than rely on engineering estimates. Surveys of these efforts include Nordhaus (1991), Gaskins and Weyant (1993), and Weyant (1999).[9]

Alternatives to prospective cost studies have arisen with the collection of increasing amounts of data on observed or reported expenditures on environmental protection. Most notably, in 1973, the Bureau of the Census (1973-1997) began collecting and publishing data on Pollution Abatement Costs and Expenditures (PACE) based on surveys of individual manufacturing plants. The data provide aggregate industry-level estimates of the out-of-pocket expenses associated with environmental protection in the manufacturing sector. Although the survey approach is widely used, Portney (1981) points out the potential problems with it: sample size, response rate, and the difficulty among respondents with accurately distinguishing pollution control expenditures. Streitwieser (1995) similarly reports that question misinterpretation has been identified as the most serious deficiency of the PACE survey.

As these surveys have been refined and become more widely accepted, they have been combined with similar estimates from other sectors of the economy to estimate economy-wide costs, as shown in Table 1 (see also U.S. EPA 1984, 1990, 1999; Vogan 1996). These estimates represent the out-of-pocket expenses attributed to different agents in the economy, shown as consumers, business, and government. As noted by Schmalensee (1994), simply tallying these estimates to estimate total costs ignores many indirect costs and may double-count expenses that are not part of final demand. We return to this issue in Section 3.

When both prospective engineering estimates and retrospective reported expenditures are available for a particular regulation, a comparison between the two is possible. Harrington et al. (2000) do this and observe an interesting pattern of results. Among 18 regulations for which they were able to establish both prospective and retrospective cost measures, the prospective estimates were higher in two of three cases. They point out, however, that this frequently occurs because the amount of pollution control was lower than initially estimated, thereby reducing costs.

Returning to our focus on the functional relationship between environmental protection and cost, $C_i(\mathbf{a},\mathbf{z})$, both engineering and survey approaches are problematic in their inability to

[9] These surveys describe aggregate modeling exercises that incorporate elements of the next section on social costs. However, the exercises are based on production and consumer models that reflect private carbon-abatement costs based on fuel-input substitution.

capture uncounted costs or coincidental benefits—uncounted by either the engineer or the survey respondent.[10] Economists have proposed many reasons why these uncounted costs might be significant, as well as methods for revealing them. We now turn to these issues.

2.2. Indirect Costs and Revealed Cost Measures

Soon after the appearance of survey-based estimates of compliance costs in the literature, economists began postulating the existence of additional, uncounted burdens associated with environmental protection. For example, regulations that required large capital expenditures could arguably crowd other productive investments (Rose 1983). Or regulations that imposed tighter limits on new emissions sources could discourage investment in otherwise newer and more productive equipment (Gruenspecht 1982; Nelson et al. 1993). Finally, there is the general concern that environmental regulation reduces operating flexibility, slowing productivity improvements in general (Joshi et al. 1997; Boyd et al. 1998).

For example, many of these concerns about indirect or "hidden" costs have been applied to the New Source Review (NSR) program in the United States (U.S. EPA 2002). Under this program, new or substantially modified facilities must meet stringent emissions standards. By exempting old facilities, older plants become relatively more profitable, and firms tend to operate them longer rather than investing in new plants. Because of the murky definition of "substantial modification," firms may also underinvest in maintaining older plants for fear of triggering NSR. It is exactly these kinds of undesirable incentives and potentially large indirect costs that have encouraged greater reliance on market mechanisms in the United States and abroad (Gruenspecht and Stavins 2002).[11]

Distinct from these costs associated with unintended or distorted behavior, Schmalensee (1994) raises several additional concerns about measurement problems that plague survey estimates: No attempt is made to measure legal fees or paperwork costs. Nor can we easily capture the cost of operating restrictions, such as the increased logging costs associated with

[10] This is less of an issue in approaches based on measures of input substitution, such as carbon abatement costs from reduced fossil fuel use.

[11] For example, the acid rain trading program in the United States and the Kyoto Protocol developed under the United Nations Framework Convention on Climate Change.

spotted owl protection.[12] Finally, the potential to redesign entire production lines to prevent pollution while also improving efficiency makes it difficult to identify specific expenditures related to environmental protection. This is particular true as efforts turn increasingly to pollution prevention (Tietenberg 1998). It is unclear whether plant managers can realistically discern the pollution prevention component of expenditures on process changes and cleaner technologies embodied in new capital (Morgenstern et al. 1999). Instructions for the reinstituted 1999 PACE survey ask respondents to include "expenditures where environmental protection was the primary purpose," later adding that this should measure the incremental costs by comparing actual expenditures with that for "the alternative technology that would have been adopted were environmental protection not a consideration."[13]

In light of that difficulty, many empirical papers have sought revealed rather than stated measures of cost at the plant level. In an early example of this approach, Gollup and Roberts (1983) estimate the cost of lower sulfur dioxide emissions on electric power utilities. They use a cross-section of plant-level utility data to relate sulfur emissions to total generating costs. More recently, Coggins and Swinton (1996) examine the marginal cost of sulfur control among a smaller set of power plants in Wisconsin. Similarly, Pittman (1981) calculates the cost of controlling water pollution using a sample of pulp and paper mills in Wisconsin and Michigan. Using the same data, Färe et al. (1993) estimate plant-specific costs from the data. McClelland and Horowitz (1999) also examine water pollution control costs at pulp and paper mills, but they include information on the permitted versus actual emissions levels. Finally, Hartman et al. (1997) conduct a broad study of abatement costs in the manufacturing sector based on early PACE data that also includes abatement information.

All of those studies make use of environmental indicators to establish cost units. More often than not, however, data on actual emissions, abatement, or other environmental indicators are unavailable. This has led to a simpler effort to compare reported environmental expenditures with total production expenditures. Anything more than a one-to-one relationship would indicate indirect costs that remain uncounted in the reported environmental expenditures. Gray (1987)

[12] Spotted owl protection restricts the harvest of old-growth timber from federally owned lands in the Pacific Northwest. The magnitude and incidence of policy effects on such a regional timber supply restriction depend on market features distinct from production technology (Murray and Wear 1998). Schmalensee's (1994) point is that a lumber firm is unlikely to go to the trouble to estimate such costs in response to a Census Bureau questionnaire.

[13] From the general definition of pollution abatement activities and the additional definition for pollution prevention capital expenditures, 1999 Survey of Pollution Abatement Costs and Expenditures (Bureau of the Census 2000).

and Barbera and McConnell (1990) first studied this question using aggregate data, while subsequent work by Gray and Shadbegian (1994); Joshi et al. (1997); and Morgenstern et al. (1999) used detailed plant-level data. Although Gray and Shadbegian (1994) and Joshi et al. found evidence of uncounted costs, Gray (1987) and Morgenstern et al. did not. Barbera and McConnell are ambiguous.

The analyses relating measured environmental performance to overall production costs at the plant level provide the most convincing data concerning the expenditures associated with specific environmental benefits. They provide a revealed rather than reported measure of environmental amenity costs. In this way, they avoid the potential pitfalls of both uncounted categories and misallocated expenses associated with both engineering and survey estimates used to quantify the relation in (1). In the absence of environmental performance measures, the second group of papers offers insight into whether survey-based cost estimates should be trusted. Though not unanimous, many of those studies support the accuracy of survey measures. Survey accuracy is particularly relevant for estimates of national environmental expenditures, which abstract from environmental performance and lean almost exclusively on survey measures.

2.3. Negative Costs?

Although considerable work has explored the potential for indirect costs at the firm level, there has been limited work on the potential for indirect benefits to firms—that is, negative costs. Porter and van der Linde (1995a, 1995b argue that firms are not always operating efficiently and that environmental regulation can lead firms to recognize and correct these inefficiencies. This can lead to a significant indirect firm benefit in terms of increased productivity. DeCanio (1993) and Lovins (1996) present similar arguments. Based on engineering and econometric studies, they argue that significant inefficiencies—particularly in energy usage—exist throughout the economy.

Despite the obvious appeal of costless or even profitable environmental improvements, most economists remain skeptical. Palmer et al. (1995) respond that despite the presence of some cost-saving offsets, environmental regulation generally must increase costs and lower profit. They point to both surveys of plant managers and conversations with company officials indicating that the realized cost savings are small compared with the cost of environmental protection itself. While conceding that substantial savings might occur in a few cases, they conclude that the bulk of the empirical evidence supports nonzero costs.

The aforementioned work by Gray and Shadbegian (1994), Joshi et al. (1997), and Morgenstern et al. (1999) provides further evidence against this hypothesis. Although these authors were searching for uncounted costs, their methodology had the capacity to identify uncounted cost savings, or negative costs, as well. Morgenstern found evidence of significant cost saving in one of four industries (pulp and paper) relative to reported expenditures, but it was not sufficient to result in negative compliance costs overall.

2.4. Government Expenditures on Environmental Protection

Costs borne by private agents, such as businesses or consumers, must be determined by observation or questionnaires. In contrast, costs borne by governments (federal, state, and local) are part of the public record. For that reason, much less effort has focused on trying to measure these costs—they are simply compiled from appropriate government reports (e.g., U.S. Bureau of the Census 1997). National cost estimates conducted by various government agencies have differed in their treatment of specific programmatic areas of government expense, such as Superfund, solid waste disposal, drinking water, and other state and local mandates, as well as their allocation of capital expense across time (Jaffe et al. 1995; Schmalensee 1994). Despite these differences, the underlying data are generally considered an accurate measure of programmatic expense.

In addition to direct expenditures on pollution control and environmental protection, the government also spends resources on enforcement and monitoring. These costs, which are also measurable based on government budget information, are typically small (roughly 2%; see Table 1) compared with national expenditures on environmental protection (Vogan 1996).

2.5. Household Regulation

Regulations that effectively take income away from consumers result in direct costs to households.[14] Although the majority of out-of-pocket costs summarized in Table 1 accrue to businesses and government, some are borne directly by households. Those noted in the table refer to expenditures on emissions control devices for motor vehicles. Interestingly, this estimate associated with emissions controls ignores the annual cost of queuing for auto emissions

[14] Here, we ignore the potential increase in consumer prices associated with environmental regulation of firms—this is a general equilibrium and/or incidence issue that we return to in the next sections.

inspections. More generally, the cost associated with environmental programs (e.g., recycling) in terms of household time is ignored.[15] Embed these types of programs in models of household production (Morris and Holthausen 1994) and one sees how the efficiency of household production is diminished, resulting in loss of household utility.

Regulation can affect households in other ways. In the case of pesticide regulation, products that pose significant environmental and health threats can be banned. In such cases households may switch to higher-cost or lower-quality substitutes. A change in product "quality" reduces household utility and represents a real cost, but it is very difficult to quantify.

A third category of cost results from regulation that restricts household behavior. Although such regulation represents a small category at present, behavior-restricting regulations, particularly recreational restrictions to advance ecosystem goals, may grow in the future. For example, regulations that ban particular types of sport fishing, exclude motorized vehicles from sensitive habitats, or restrict all human intrusion into special areas cause the value of those recreational experiences to decline for those who otherwise would engage in them.

The last category of household costs we mention has to do with land use. Development restrictions on a piece of real estate cause the property value to decline and cause the property owner to suffer a capital loss. The restrictions might be imposed to protect an ecosystem, preserve environmental amenities (beaches, for example), or limit suburban sprawl. Increased concern over each of these issues suggests that land use restrictions will be an important regulatory tool for some time to come.

2.6. Uncertainty

Few analyses of environmental protection costs consider uncertainty in their analyses. Some perform a sensitivity analysis over key assumptions (e.g., Chapter 7 of U.S. EPA 1999). However, a sensitivity analysis merely shows how estimates change with alternative assumptions; it does not indicate what value to use for a benefit comparison or how decisions should be made.

Considerable work in statistical decision theory dating back to at least Wald (1950) and more recently Berger (1985) describes a straightforward, though computationally demanding,

[15] Of course, this is not inconsistent with the absence of leisure value in the National Income and Product Accounts.

approach to decisionmaking under uncertainty. By assigning probabilities to different outcomes, it is possible to compute expected costs. More generally, a policymaker willing to specify preferences over both uncertain costs and benefits—including risk aversion—can identify policies preferred in expectation.[16] Kolstad (1982) and Pizer (1999) provide environmental examples of this approach. A related issue, when both benefits and costs are considered jointly, is whether price policies are preferable to quantity policies. Price policies tend to lower both expected costs and expected benefits relative to quantity policies. Weitzman (1974) derives conditions where one instrument is preferred.

Dixit and Pindyck (1994) and Pindyck (1995) point out another issue partly but not wholly related to uncertainty: the irreversibility associated with some environmental policies. The potential exists for environmental protection activities to involve irreversible costs as well as for environmental consequences to involve irreversible damage. In the former case, the cost of a policy is raised by the forgone option value associated with waiting to see whether the environmental protection is necessary. In the latter case, the cost of a policy is lowered by the value of the option to prevent possibly irreversible damage. This option value could depend wholly on the passage of time but most often involves learning about uncertain outcomes. Computing the option value is further complicated when the choice of policy influences the learning process (Kelly 1999; Nordhaus 1997).

2.7. Discounting

Most cost analyses focus on current or annualized expenditures, as shown in Table 1. This facilitates a straightforward comparison with annualized benefits resulting from improved environmental quality. The emergence of global climate change as a major environmental policy issue forces us to rethink that approach. In particular, it forces us to consider comparisons of costs and benefits across very long periods of time, since climate change mitigation expenditures today yield benefits far in the future.

To compare costs (and benefits) over time, costs in the future are discounted to the present at a particular rate. The appropriate rate of discount has been subject to much debate (Lind 1982; Arrow et al. 1996; Bazerlon and Smetters 1999), with different rates leading to

[16] The notion of risk aversion captures the idea that we are not indifferent between a risky $10 gain and a certain $10 gain. In statistical decision theory, this amounts to defining a loss function; in utility theory, it reflects a choice of utility or welfare function.

dramatically different estimates of benefits and costs (Cline 1992). In the case of climate change, rates near zero support aggressive policy responses, with costs appearing small relative to benefits; rates exceeding 5% suggest modest policy responses, with costs appearing large relative to benefits. These observations have led to recent work on the consequences of uncertainty about the discount rate itself (Weitzman 2001; Newell and Pizer 2003). Even ignoring the question of the appropriate rate, some qualitative results remain. When the focus of climate change mitigation policy is a long-term concentration target, rather than a particular emissions profile, Wigley et al. (1996) demonstrate that discounting as well as adjustment costs favor some delay in mitigation relative to an immediate reversal in emissions growth.

In dynamic equilibrium models, much of the concern about discounting is embedded in the model specification via consumer preferences over time. As we will see in the next section, such models allow one to summarize the total cost associated with environmental controls across many periods, just as Table 1 summarizes the costs associated with environmental controls across different agents.

3. General Equilibrium Effects

The concepts described so far have focused on the costs associated with regulatory compliance by various economic actors—firms, governments, or households. Such analysis ignores the indirect impact that environmental protection activities in one market can have on activities in other markets, as well as feedback in the original market, as the economy equilibrates to these additional burdens. We are most often interested in the total cost *including* these indirect effects on welfare—effects that can sometimes be quite large relative to the more obvious consequences measured by the regulated entities. Stated another way, the question to ask about a proposed regulation or policy is the overall burden to the national economy, not just the sum of the direct costs measured in isolation.

Welfare effects in other markets arise because of preexisting distortions in those markets from current taxes or regulation. Such distortions—a difference between the point where the market would like to equilibrate, matching supply and demand at one price, and where the market does equilibrate in the presence of taxes or regulation—create deadweight loss. This deadweight loss is a cost that reduces the production possibilities of the entire economy. For example, taxes on labor income discourage people from working as much as they would if they received the full value of their time. When an environmental regulation is introduced, it can affect the existing deadweight loss in these already-distorted markets by shifting supply or demand and changing the magnitude of the deadweight loss in that market.

Consideration of equilibrium price changes can affect the original cost estimates for the directly regulated entities as well as welfare in other markets. For example, the cost of an electricity policy that encourages fuel substitution from coal to natural gas will rise if increased natural gas demand raises the price of natural gas versus a case where additional natural gas is available at the same price. On the other hand, the cost of the policy will fall if consumers demand less electricity as the price of electricity rises in response to the policy, versus a case where electricity demand remains constant. To the extent that distortions do not exist elsewhere in the economy, however, total costs can still be measured in the regulated sector even as the economy equilibrates.[17]

As we try to measure these total costs, this overall burden, we increasingly focus on the real decline in final demand—consumption, investment, and government spending—to avoid double counting (Schäfer and Stahmer 1989; Schmalensee 1994). Environmental regulations that raise the price of energy, for example, raise both the cost of energy and the cost of manufactured goods that are energy-intensive. If we count both the increase in manufacturing costs and the increase in consumer expenditures on higher-priced manufactured goods, we double-count the cost of the regulation. This has led to a focus on the consequences for real GDP in many analyses. Such a focus, however, fails to account for the consumption of leisure (and other nonmarket transactions) and ignores the distinction between investment in pollution control equipment and investment in productive capital. This means that typical analyses focused on "GDP effects" can be misleading, as highlighted below.[18]

Once the economy equilibrates, the change in final demand will include both the original costs discussed in the previous section (and passed on to households as either higher prices or

[17] That is, the marginal cost of incremental regulation in the regulated sector equals the marginal welfare cost of the incremental regulation measured across the entire economy. This is a consequence of the First Fundamental Welfare Theorem—a competitive market equilibrium is welfare maximizing and therefore the market value of the incremental regulation can be viewed as a shadow price. See, for example, Chapter 16 of Mas-Colell et al. (1995).

[18] Weyant and Hill (1999), for example, focus on GDP losses in analyzing global climate change policies. A later article in the same special issue (McKibbin et al. 1999) shows an example (Table 4) where GDP falls and consumption rises in 2020. Although GDP and consumption will generally move in the same direction, the quantitative differences raise interesting questions about how quantitative results should be presented—in terms of the more familiar (and popular) GDP measure or the more economically relevant consumption measure.

lower factor income), as well as indirect effects occurring in other markets. Consumer preferences provide a natural metric for valuing these changes in final demand. Consumers, in the guise of both purchasers of goods and owners of productive inputs (as well as the beneficiaries of investment returns and government provision of public goods), are the implicit if not explicit focus of most cost analyses.[19] A change in their well-being can be monetized by computing the equivalent variation in consumer income. Equivalent variation measures the change in income that would lead to the same change in a consumer's well-being as the policy under scrutiny.[20]

3.1. An Example

At this point, it is useful to consider an example to illustrate the different ways in which a partial equilibrium analysis—as well as GDP measures—can fail to correctly capture aggregate costs. We imagine a simple economy where a single consumer works for the economy's sole employer, a power plant that produces energy (x). The plant uses a linear technology to produce x_0 units of energy with $p_0 \cdot x_0$ hours of labor. We treat labor as the numeraire good, with a price of one, so costs, income, and prices are denominated in hours of labor. The central question is, how do we calculate the cost of an environmental regulation that changes the linear production technology so that it now costs $p_1 = p_0 + c > p_0$ to produce each unit of energy?

3.1.1. When Partial Equilibrium Is Right

The simplest approach would be to conclude that given current production x_0, the cost of the regulation is $c \cdot x_0$. Such an analysis might be the result of an impact analysis for a new regulation where little was known about the likely response of consumer demand for energy to higher prices. The calculation would be exactly right if in fact consumer demand were fixed. Why? With fixed demand for a good, any change in the price is equivalent to a lump-sum

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[19] That is, in the end what most analyses care about is the welfare of people (households). Given the indirect benefits to households of investment (in static models) and government spending, *ad hoc* assumptions are often made in general equilibrium analyses—for example, requiring investment and government spending to remain constant.

[20] Two alternatives to equivalent variation (EV) are discussed in the literature: compensating variation and Marshallian surplus. Freeman (1985) recommends EV because it represents the desired objective—a money equivalent of welfare change; see also Jorgenson and Slesnick (1985). The idea originated with Hicks (1942); Chipman and Moore (1980) show that EV represents an indirect utility function and is therefore appropriate for welfare comparisons.

transfer—the consumer always pays for the good and then uses leftover income to buy other goods on the margin. The added cost of energy $c \cdot x_0$ in this case is exactly like a $c \cdot x_0$ loss of income available for other goods. Graphically, we can see this in the left panel of Figure 1, with vertical demand and horizontal supply curves. Here the area above the price line—the consumer surplus—falls by $c \cdot x_0$ with the regulation, as indicated by the shaded area.[21]

Even in this case, where the simple partial equilibrium analysis works, measures like GDP can be misleading. GDP can be viewed as a measure of either factor income or marketed final demand—in this case, wage income and purchases of energy. Wage income rises in this example, as the consumer works more to support her fixed demand—that is, GDP *rises* in response to a costly regulation. A more careful analysis might note that real energy use remained constant, but in no case would a study of GDP reveal a decline because the quantity of marketed goods remains the same in this example, and GDP does not capture changes in nonmarketed goods, such as leisure.

Returning to the partial equilibrium analysis, input prices may not be perfectly elastic and demand may not be perfectly inelastic. For example, suppose the right panel in Figure 1 describes demand for energy. The original analysis based on fixed energy demand would have overstated costs by ignoring the flexibility of the consumer to substitute away from an increasingly expensive good. A better analysis might correctly predict—or, after the regulation is in place, observe—x_1 as the new demand for energy and conclude that costs equaled $c \cdot x_1$. This would reflect the current expenditures recorded by the power plant as being environmentally related.

Such a calculation, an analysis of environmental expenditures after regulation is in place, is precisely the basis of the national cost estimates discussed in the first section, where firms are surveyed about their abatement expenditures.[22] Yet such analyses routinely ignore the loss of consumer welfare associated with reduced consumption of more expensive, regulated goods—in our example, reduced energy demand from x_0 to x_1. In this way, they entirely miss the regulatory costs associated with goods that are banned or whose regulated costs become so high that no

[21] Introductory economics teaches that changes in the area between the demand curve and the price line, referred to as consumer surplus, can be used to measure changes in consumer welfare as prices change. Intuitively, when the consumer is forced to pay more for the purchased good, it is as though that much income is being taken away, and as prices rise and demand falls, there will not be as much of an income loss for the next incremental price increase.

[22] For example, U.S, EPA (1990a), Rutledge and Vogan (1994), OECD (1999).

production occurs and no control costs are reported. We can fix the calculation in our example relatively easily: the true welfare cost is the area under the demand curve between $p = p_0$ and $p = p_1$, or $c \cdot \frac{1}{2}(x_1 + x_0)$; that is, the change in consumer surplus.[23] Note that again, the estimated change in GDP will likely be positive unless the decline from x_0 to x_1 is large. That is, the change in GDP equals $c \cdot x_1 - (x_0 - x_1)\, p_0$.

3.1.2.When Partial Equilibrium Is Wrong

So far, the partial equilibrium cost analysis of environmental regulation in our simple model has done well when the consumer's response is properly captured. However, we have yet to consider what happens when distortions, especially tax distortions, already exist in other markets. In this case, we have to consider not only costs in the regulated market, but also changes in existing costs measured in those other markets.

Figure 2 shows the consequences of a new environmental regulation in the energy (left panel) and labor markets (right panel) in the presence of a preexisting tax on labor income. When the regulation is implemented, employment decreases from l_0 to l_1 while energy demand decreases from x_0 to x_1. The shaded region shows deadweight loss in both markets.[24] The important thing to recognize is that the deadweight loss in the labor market changes as employment shifts from l_0 to l_1. That is, the environmental regulation not only creates a cost in the energy market where it is imposed, it also influences the cost of existing policies (a labor tax in this case) in other markets.

One can make the example more precise algebraically. Consider the following utility function that underlies Figure 1 and Figure 2:

$$u(x,l) = -\frac{(x^* - x)^2}{2} + (L - l) \qquad (2)$$

[23] Here the demand schedule is linear. The welfare analysis is not correct unless we are careful to use compensated (holding utility constant) rather than uncompensated (holding income constant) demand schedules. Note that as the price goes up by an initial increment *dp*, the cost of maintaining the current level of well-being rises—which we correctly measure as $x_0\, dp$. Now, we want to continue to measure the cost of maintaining that *same* level of well-being as the price rises by the next increment, not the cost of maintaining the new, *lower* level of well-being associated with a higher price and fixed income. Therefore, the demand schedule must hold welfare, not income, constant. When preferences are quasi-linear (e.g., the marginal utility of one good equals unity), income and utility do not affect demand for any good other than the quasi-linear good, and both compensated and uncompensated demand schedules are the same.

[24] In both cases, we show *compensated* labor supplies, as remarked in the preceding footnote.

where x is demand for energy, l is supply of labor (so $L - l$ is demand for leisure), and x^* is a parameter describing a level of satiation with energy. The consumer faces a budget constraint that now allows for distinction between the wage paid by the employer (unity) and the wage received by the consumer (w):

$$p \cdot x + w \cdot (L - l) = w \cdot L + T \tag{3}$$

where expenditures on energy, $p \cdot x$, plus implied expenditures on leisure, $w \cdot (L - l)$, must equal the value of the labor endowment, $w \cdot L$, plus any income transfer T from the government . With the wage paid by the power plant equal to one, we have $w = 1 - t$ where t is the labor tax. The first-order conditions imply that

$$x = x^* - \frac{p}{w} \text{ and } l = \frac{p}{w}\left(x^* - \frac{p}{w}\right) + L - \frac{e}{w} = \frac{p}{w}\left(x^* - \frac{p}{w}\right) - \frac{T}{w} \tag{4}$$

providing a solution for energy demand and labor supply. This, in turn, allows us to write expenditure and indirect utility functions using (2) and (4)

$$\begin{aligned} e(p,w,u) &= p\left(x^* - \frac{p}{w}\right) + w\left(u + \frac{1}{2}\left(\frac{p}{w}\right)^2\right) \\ v(p,w,e) &= -\frac{1}{2}\left(\frac{p}{w}\right)^2 + \frac{e}{w} - \frac{p}{w}\left(x^* - \frac{p}{w}\right) \end{aligned} \tag{5}$$

where u is utility, v is the indirect utility function, and e is total expenditures, including leisure, and equal to $w \cdot L + T$ in (3).

Previously, we asserted that the cost to consumers without a labor tax (when $w = 1$) equals $c \cdot \tfrac{1}{2}(x_1 + x_0)$ in Figure 1. Therefore, we should be able to see the following relation:

$$e\left(p_0, 1, u_{\text{no labor tax, no regulation}}\right) - c \cdot \tfrac{1}{2}(x_1 + x_0) = e\left(p_0, 1, u_{\text{no labor tax, regulation}}\right) \tag{6}$$

That is, if we took away $c \cdot \tfrac{1}{2}(x_1 + x_0)$ in income from the untaxed, unregulated expenditure level—keeping the prices the same—utility would fall to the same level created by the regulation. How can we see this in our model? Without taxes, $e\left(p_0, 1, u_{\text{no labor tax, no regulation}}\right)$ equals $e\left(p_0 + c, 1, u_{\text{no labor tax, regulation}}\right)$ equals L. That is, the equilibrium expenditure level always equals the income level, and without taxes this always equals L.[25] Therefore,

[25] Note that the equilibrium expenditure in the face of particular policy equals the expenditure function evaluated at the price and utility resulting from the particular policy; it necessarily equals $w \cdot L + T$.

$$e\left(p_0, 1, u_{\text{no labor tax, no regulation}}\right) - e\left(p_0, 1, u_{\text{no labor tax, regulation}}\right) =$$
$$e\left(p_0 + c, 1, u_{\text{no labor tax,regulation}}\right) - e\left(p_0, 1, u_{\text{no labor tax, regulation}}\right)$$

But this equals $c \cdot \frac{1}{2}\left(x_1 + x_0\right)$ through manipulation of the expenditure function in (5), and so (6) is true.[26]

With a preexisting labor tax, the calculation is slightly more complex. Again, we want to compute the change in income equivalent to the utility loss associated with the regulation—but now with a labor tax present,

$$e\left(p_0, w, u_{\text{labor tax, no regulation}}\right) - ? = e\left(p_0, w, u_{\text{labor tax, regulation}}\right)$$

analogous to (6). But unlike the untaxed case in (6), where equilibrium expenditures always equaled the same level of income *L*, equilibrium expenditures now equal $w \cdot L + T$, where *T* changes as we move from the unregulated to the regulated equilibrium. In equilibrium, we know that $T = t \cdot l$ because the government transfer must equal the tax revenue. Using this, we have

$$e\left(p_0, w, u_{\text{labor tax, no regulation}}\right) + t \cdot \left(l_1 - l_0\right) = e\left(p_0 + c, w, u_{\text{labor tax, regulation}}\right)$$

That is, the taxed equilibrium expenditure without regulation, plus the change in tax revenue associated with the regulation, equals the taxed equilibrium expenditure with regulation. As before, $e\left(p_0 + c, 1, u_{\text{labor tax,regulation}}\right) - e\left(p_0, 1, u_{\text{labor tax, regulation}}\right)$ equals $c \cdot \frac{1}{2}\left(x_1 + x_0\right)$, so we have

$$e\left(p_0, w, u_{\text{labor tax, no regulation}}\right) - e\left(p_0, w, u_{\text{labor tax, regulation}}\right) = c \cdot \frac{1}{2}\left(x_1 + x_0\right) + t \cdot \left(l_1 - l_0\right) \qquad (7)$$

Intuitively, welfare cost equals the cost measured in the partial equilibrium energy market analysis, plus the change in employment times the tax wedge (e.g., the difference between the marginal benefit and marginal cost of an hour worked).

How does this compare with simpler analyses? As before, we could imagine assuming that the before-regulation output level $x = x_0$ remains fixed, that the after-regulation output level $x = x_1$ remains fixed, or that output changes based on the compensated demand schedule. With per unit regulatory costs *c*, we would estimate partial equilibrium costs of $c \cdot x_0$, $c \cdot x_1$, or $c \cdot \frac{1}{2}\left(x_1 + x_0\right)$, respectively (the latter corresponds to the shaded area in the left panel of Figure 2). All of these estimates ignore costs in the labor market arising from the existing labor tax.

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[26] Note that $\left(p_0 + c\right)\left(x^* - \frac{1}{2}\left(p_0 + c\right)\right) - p_0\left(x^* - \frac{1}{2}p_0\right) = \left(2x^* - 2p_0 - c\right) = c \cdot \frac{1}{2}\left(x_0 + x_1\right)$. It is also evident by the definition of the compensated demand curve in Figure 2, which equals the derivative of the expenditure function with respect to price.

How does this compare with a general equilibrium calculation based on Figure 2? One could imagine using Figure 2 to compute the cost not only in the energy market but in the labor market as well—as we suggested at the beginning of this section. This would not be entirely correct, either. The compensated demand curves used for welfare analysis in these diagrams measure welfare consequences for *price changes in their own markets*. They cannot be used to accurately measure welfare changes in one market arising from price changes in another market because they are computed holding those other prices fixed. That is, adding the shaded region between $l = l_0$ and $l = l_1$ in the right panel of Figure 2 to the shaded area in the left panel of Figure 2 yields too small an estimate.

Table 2 summarizes the variations we just went through and the consequences that would be measured based on various notions of costs. The first line shows the costs that might show up on a survey where firms were asked to report their expenditures on environmental protection. This ignores the change in production level that may be a result of the regulation. An accurate partial equilibrium analysis, on the second line, shows what a good analysis of the regulated market would yield. The third line shows how a model trained to measure GDP impacts would register costs (measurable as either the change in final goods sold or the change in production factors purchased). Finally, the fourth line shows the true welfare costs, measured as the loss of income that would lead to a welfare level equivalent to the imposed regulation—the equivalent variation in income.

The key observation is that no measure of costs correctly measures welfare costs in all cases *except* the general equilibrium measure. This illustrates the first main point of Section 3. Namely, both the measurable costs to the firm in a single market as well as changes in GDP can be misleading measures of the real cost to consumers. Costs to the firm measured in a partial equilibrium analysis ignore general equilibrium changes in output and prices and miss potential welfare changes from altered production levels and distortions in other markets. Changes in GDP are misleading because they ignore the consumption of leisure and treat regulatory expenses as if they directly benefited consumers—as if they were expenditures on food or housing.

The remainder of this section discusses both approximate and exact methods of measuring the costs of environmental regulation based on a general equilibrium model. This discussion builds up to the second main point of this section: an accurate measure of the total cost of environmental regulation depends on a complete model of technology, preferences, and government behavior. Different assumptions about these features of the economy can lead to different estimates of the impact of environmental regulation.

3.2. Extended Market Analysis

A central failure of simple partial equilibrium analysis is its inability to handle price and output changes in related markets. Therefore, one solution is to extend the analysis to markets related vertically or horizontally (Just et al. 1982; Whalley 1975; Kokoski and Smith 1987). A vertically expanded analysis would involve computing the input factor supply schedules, rather than treating input prices as fixed, and computing consumer and producer surplus loss in those markets. In the horizontal case, this would involve computing a demand schedule for the output of the firm undertaking environmental activities, rather than assuming output remains fixed, and then using that schedule to compute a change in surplus.

Although such extensions would seem to offer improvements, that is not the case empirically. Whalley (1975) computes the effect of capital tax reform in the United Kingdom and finds that extending the partial equilibrium to include horizontal effects worsens most of the welfare estimates relative to the true social costs. In an environmental example, Kokoski and Smith (1987) examine the cost of unmitigated global warming. In separate horizontal and vertical extensions to a simple partial equilibrium analysis, they find that both are considerably worse when only one market is affected by global warming. When multiple markets are affected by global warming, the extensions do better.

In our earlier example, summarized in Table 2, an extended market analysis helps. By considering the substitution opportunities in the leisure market, an extended market analysis would properly capture the welfare effects in the cases without taxes. With taxes, however, this approach fails to capture welfare loss in the labor market. An increase in the consumption of leisure (and decrease in labor supply) due to substitution exacerbates the deadweight loss associated with the labor tax. These effects are ignored by the extended market analysis.

3.3. Approximating Losses in Other Markets

An alternative to these limited extensions is to consider an approximation based on the effect in all markets, focused on deadweight loss. Both Diamond and Mirrlees (1971) and Harberger (1971) provide guidance, suggesting expressions of the form:

$$\text{cost} = C_j\left(\mathbf{a}\right) - C_j\left(0\right) - \sum_{i \neq j} t_i\left(X_i\left(\mathbf{a}\right) - X_i\left(0\right)\right), \tag{8}$$

where as before the vector **a** parameterizes the policy, $C_j\left(\mathbf{a}\right) - C_j\left(0\right)$ is the loss of social surplus in the market bearing the direct out-of-pocket costs, t_i measures preexisting taxes in each of the remaining markets $i \neq j$, and $X_i(\mathbf{a}) - X_i(0)$ indicates the change in output in each of those markets due to the policy.[27]

Note that when we apply (8) to our example in Section 3.1, it delivers the correct answer with a preexisting labor tax. In addition to the partial equilibrium cost in the energy market, shown in the left panel of Figure 1 and equal to $c \cdot \frac{1}{2}(x_1 + x_0)$, we have costs in the labor market equal to the labor tax t times the change in labor supply, $l_1 - l_0$. This yields the correct welfare cost $c \cdot \frac{1}{2}(x_1 + x_0) + t \cdot (l_1 - l_0)$. With a single representative consumer and a convex production technology, we can develop a more explicit expression for (8). Suppose **y** is the vector of (net) goods produced by the firm, and **x** is the vector of (net) goods consumed by the consumer. We can define **Y** as the matrix of first derivatives of **y** with respect to producer prices **p** and **S** as the matrix of compensated first derivatives of **x** with respect to consumer prices **q**.[28] Suppose **a** is simply a description of the amount of pollution abated—a scalar *a*—and let abatement of that pollutant be good *j* in the above description. Consider a small change in pollution abated, *da*. If we assume that the existing distortions $\mathbf{t} = \mathbf{q} - \mathbf{p}$ do not change, we have $d\mathbf{x} = \mathbf{S} \cdot d\mathbf{p}$ (assuming no change in utility) and $d\mathbf{y} = \mathbf{Y} \cdot d\mathbf{p}$ with $d\mathbf{p} = d\mathbf{q}$. If we also assume that the existing excess supply (government and/or foreign supply), $\mathbf{y} - \mathbf{x}$, remains constant, we have $d\mathbf{y} - d\mathbf{x} = e_j da$ where e_j is unit vector *j*. Combining these relations, we have (since taxes are fixed)

$$d\mathbf{p} = d\mathbf{q} = (\mathbf{Y} - \mathbf{S})^{-1} e_j da$$

The actual change in consumption would be given by

$$d\mathbf{x} = \mathbf{S} \cdot d\mathbf{p} = \mathbf{S}(\mathbf{Y} - \mathbf{S})^{-1} e_j da \qquad (9)$$

Normally, marginal changes in consumption have no effect on welfare since, at the margin, producer cost equals consumer benefit. In the presence of distortionary taxes, however, consumer benefit exceeds producer cost by the tax rate. Increased consumption will raise

[27] See Equation (92) in Diamond and Mirrlees; Equation ($5'''$) in Harberger. Diamond and Mirrlees present marginal conditions for optimal provision of a public good, where their marginal cost expression equals the derivative of the right-hand side of (8) with respect to **a**, replacing marginal partial equilibrium costs C_j' with the derivative of an aggregate production function with respect to the regulated good G_j (they use *k* as the regulated good). In this way, we view Equation (8) as a discrete approximation to the cost side of Diamond and Mirrlees, and in turn an approximate metric for comparison with monetized benefits. The comparison with Harberger is more straightforward, as he is already focused on a discrete approximation. The only difference is that his expression for the cost of a tax in market *j*, $\frac{1}{2} T_j^* \Delta x_j$ is replaced by our cost of the regulation in market *j*, $C_j(\mathbf{a}) - C_j(0)$.

[28] Note that this approach requires strict convexity in production and preferences.

welfare; decreased consumption will lower welfare. To compute the change in welfare, we multiply change in consumption (9) by the tax rate:

$$\mathbf{t}\cdot d\mathbf{x} = \mathbf{t}\cdot\mathbf{S}\left(\mathbf{Y}-\mathbf{S}\right)^{-1} e_j da$$

This is exactly the second term in (8) as derived by Bruce and Harris (1982) in their analysis of cost-benefit criteria for evaluating small projects.[29] Rather than applying a marginal cost-benefit analysis to the incremental cost of producing *a*, $C_j'\left(a\right)$, this suggests that we should base our analysis on

$$C_j'\left(a\right) - \mathbf{t}\cdot\mathbf{S}\left(\mathbf{Y}-\mathbf{S}\right)^{-1} e_j\,, \tag{10}$$

adjusting for surplus changes in other markets. Diewert (1983) extends this approach even further, considering the effect of import tariffs for a small open economy.

As the preceding discussion suggests, it is not easy to estimate $X_i\left(\mathbf{a}\right) - X_i\left(0\right)$, the effect of a particular policy in every market. This requires a local model of general equilibrium effects, captured by **Y** and **S** in (10). However, as Harberger (1971) notes, the only real concern will be those markets that have large distortions (t_i) and are significantly affected by the policy (large ΔX_i). Interestingly, this bit of intuition has pervaded a large volume of literature on the importance of general equilibrium analysis over the intervening 30 years.[30]

3.4. General Equilibrium Analysis

Efforts to include social costs outside the directly regulated market lead naturally to general equilibrium cost analyses. Where the previous approach in (10) uses linear approximations of preferences and technology to work out new equilibria, a general equilibrium model works directly with utility and profit functions, maximizing both subject to market equilibrium conditions. A general equilibrium model can also be used to consider alternative assumptions about taxes and spending (where the previous approach assumed fixed tax rates and fixed government purchases).

[29] See Equation (7) in Bruce and Harris (1982).

[30] In a recent article on the use of "Harberger triangles," Hines (1999) notes that despite the criticism that analysis based on such triangles ignores general equilibrium effect, Harberger himself consistently emphasized the importance of spillover effects in distorted markets.

General equilibrium analyses can involve both numerical simulation and analytic results. Analytic work on general equilibrium problems has focused on establishing conditions for second-best optima and the prices appropriate for marginal project evaluation. Just as Harberger and others revealed the importance of costs associated with tax distortions, this work has revealed the importance of fiscal responses to environmental and other public policies. It is always possible—and frequently necessary—to adjust taxes or spending in response to nonfiscal policies. The necessity arises from the need to maintain the government budget constraint as the tax base changes. These adjustments can have significant welfare consequences.

Consider a simple model with a single consumer, a single firm, and a government. The consumer has endowment $\mathbf{k}$ and utility $u(\mathbf{x})$, where both $\mathbf{k}$ and $\mathbf{x}$ are N element vectors spanning the good and factor space of the economy. The firm possesses a constant-returns-to-scale technology with unit profit function $\pi(\mathbf{p})$ that is operated at an arbitrary scale α, provided profit is nonnegative based on producer prices $\mathbf{p}$. Finally, the government collects taxes on commodity transactions between the consumer and firm. The government also purchases a fixed bundle of goods $\mathbf{g}$ with any government balance (negative or positive), resulting in a transfer to the consumer b. Letting $e(\mathbf{p} + \mathbf{t}, u)$ be the expenditure function dual of $u(\mathbf{x})$ for consumer prices $\mathbf{p} + \mathbf{t}$, we can write the equilibrium conditions as:

Household budget: $e(\mathbf{p} + \mathbf{t}, u) = b + (\mathbf{p} + \mathbf{t}) \cdot \mathbf{k}$ **(11)**

Government budget: $\mathbf{t} \cdot \mathbf{x}(\mathbf{p} + \mathbf{t}, u) = b + \mathbf{p} \cdot \mathbf{g}$

Zero profit: $\mathbf{p} \cdot \mathbf{y}(\mathbf{p}) = 0$

Market clearing: $\mathbf{y}(\mathbf{p})\, \alpha + \mathbf{k} = \mathbf{g} + \mathbf{x}(\mathbf{p} + \mathbf{t}, u)$,

where utility u, producer prices $\mathbf{p}$, output scale α, and government transfers b adjust to reach an equilibrium. By Walras law, one of the budget or market-clearing equations is redundant. To fix the nominal variables $\mathbf{p}$, $\mathbf{t}$, and b, one price must also be fixed as numeraire. A typical approach is to fix first good as numeraire ($p_1 = 1$) and to drop the market-clearing constraint on that first good. This model is a simplified version of the kinds of models used for analytic and numerical work on public good provision in a second-best general equilibrium setting (Diamond and Mirrlees 1971; Diewert 1983; Stiglitz and Dasgupta 1971; Bovenberg and Goulder 1996; Rutherford 1999).

An important observation about these equilibrium conditions is that to close the model, it is necessary to make an additional assumption about how the government meets its budget constraint. In all, the model (11) involves $N + 1$ endogenous variables—$\mathbf{p}$ (except p_1), u, and α—

and $N + 2$ equations (recall that one is dropped by Walras law). One additional degree of freedom must be introduced among the otherwise fixed government variables, $\mathbf{t}$, $\mathbf{g}$, or b. The approximation in (10) assumed that taxes and government purchases were fixed, thereby endogenizing government transfers b by default. Alternatively, b can be fixed and the tax rates can be chosen optimally subject to the budget constraint. Or, any one of the tax rates can be allowed to adjust and b can be fixed.[31] When we consider policy evaluations, these different assumptions often have important consequences for the implied welfare change.

Using this model, the cost associated with providing some incremental amount of a public good $d\mathbf{g} = e_j de$ can be computed two ways. In an analytic model, we can maximize u subject to the above constraints. If γ is the Lagrange multiplier on the government budget constraint, and ρ_j is the Lagrange multiplier on the jth market-clearing constraint, the social cost of the incremental public good will be

$$\left(\gamma p_j + \rho_j\right) de \tag{12}$$

In a numerical simulation model we can simulate the change du as we increase $\mathbf{g}$ by $d\mathbf{g}$. Using a money-metric utility scaling (Samuelson 1974) such that $e(\mathbf{q}, u) = u$ at the initial equilibrium, du can be interpreted as the equivalent variation cost of the policy for a small $d\mathbf{g}$.

Stiglitz and Dasgupta (1971), Ballard and Fullerton (1992), and others use the analytic approach to derive results with optimal taxes and no transfers. They show that the cost of a public good equals its price multiplied by one plus the marginal excess burden of the tax system. From expression (12), optimal tax policy must imply that ρ_j is proportional to p_j if this rule is going to hold for all goods j. This is similar to the results of Bovenberg and de Mooij (1994) and Bovenberg and Goulder (1996), who consider optimal Pigouvian taxes on pollution in the presence of distortionary taxes. Work by Bruce and Harris (1982) and Diewert (1983) reveals the form of ρ_j when taxes are instead exogenous and transfers adjust. In contrast to Stiglitz and Dasgupta, they find ρ to be a matrix-weighted average of producer and consumer prices.[32] Boadway (1975) provides guidance when some taxes are fixed and other taxes adjust.

[31] In this case, we can measure the excess burden of a particular tax by varying b.

[32] This result can be seen in (10) by taking $C'(a)$ to be a producer price p and replacing t with $q - p$.

3.5. Numerical Analysis

Analytical models are limited by the need for tractability. Numerical models, however, are limited only by computational power. An early example of numerical general equilibrium analysis is Ballard et al. (1985), who use a numerical general equilibrium model of the U.S. economy to estimate marginal excess tax burdens ranging from 17 to 56 cents on the dollar. Environmental applications of numerical general equilibrium models include Conrad and Schröder (1991), Bergman (1991), Whalley and Wigle (1991), and Böhringer and Rutherford (1997). Bergman considers the loss of gross national product (GNP) and marginal control cost of SO_2, NO_x, and CO_2 in Sweden. Conrad and Schröder use a computable general equilibrium model to estimate the carbon tax required to generate a specified annual emissions reduction and then measure the impacts on GNP and SO_2 and NO_x emissions in Germany. Böhringer and Rutherford also consider carbon policies in Germany, but unlike Bergman and Conrad and Schröder, they focus on welfare consequences. Similarly, Whalley and Wigle consider global consequences of CO_2 reductions with an emphasis on welfare as well as changes in patterns of trade.

When the control costs used in general equilibrium analyses are based on surveys of regulatory expense, there is often little information concerning the removal process or input substitutions used to reduce emissions. Consequently, most models make simplifying assumptions about abatement technology: either input usage for abatement is proportional to overall input usage[33] (Jorgenson and Wilcoxen 1990; Hazilla and Kopp 1990), or abatement involves only capital and labor (Ballard and Medema 1993). Recent work by Nestor and Pasurka (1995) demonstrates the potential problems with that approach, using detailed information on abatement technology (Schäfer and Stahmer 1989). They find that although simplifying assumptions about abatement technology lead to similar conclusions about which sectors are most adversely affected by regulation (based on declines in output), the magnitudes are significantly different. The proportional input model tends to underestimate output changes, but the capital-labor model overestimates output changes. This occurs, the authors argue, because the primary factor (capital-labor) intensity of the actual expenditures lies between the primary factor intensity implied by the proportional input and capital-labor models.

[33] This is equivalent to assuming that environmental regulation is a Hicks-neutral productivity effect.

3.6. Environmental Policy versus Public Good Provision

The question of optimal provision of public goods motivated many of the analytic exercises we have mentioned (Diamond and Mirrlees 1971; Diewert 1983; Stiglitz 1979). These studies focused on the problem of procuring market goods for public use and on the appropriate way to value those goods, leading to Equation (8) and subsequent variations. In contrast, environmental regulation focuses on nonmarket goods and frequently involves the manipulation of property rights. For example, emissions standards may restrict the right of firms to emit above a certain rate without any direct market consequence.

The intuition behind Equation (8) is the same in both cases, however. There is one market where the public policy imposes primary losses of social surplus. In the first case, increased government consumption of a particular good raises production costs in that market in order to increase output. In the second case, environmental regulation raises production costs in the regulated firms' output market while the output level remains the same. The first term in (8) reflects these direct costs. In response to these changes, there will be equilibrating effects in other markets. Tax-induced discrepancies between the marginal cost and marginal benefit of goods in those markets lead to welfare changes reflected in the second term in (8).

When market mechanisms are used to reduce pollution, we can focus on the pollution market directly, rather than the output market for regulated firms. The initial factor supply curve in this market is flat and passes through the origin—equilibrium emissions set marginal abatement costs equal to zero. The direct cost of any regulation is the change in area under the pollution demand curve or the integral $\int p(a)\,da$, where a measures the quantity abated and $p(a)$ is the market price at abatement level a (or the tax required to induce abatement level a). Indirect welfare changes, as before, are computed as the sum of changes in consumption times the tax rate in each market.

The use of market-based mechanisms—tradable permits and emissions taxes—changes the otherwise straightforward application of earlier results from public finance. In addition to encouraging abatement and creating pollution control costs, these mechanisms also associate costs with uncontrolled emissions. These costs reflect the value of permits or taxes associated with unabated emissions. In other words, firms must pay for their inframarginal emissions, not just the emissions they reduce. Yet these costs on inframarginal emissions are not really costs at all—they are transfers to whoever owns the emission property rights. This could be firm owners, consumers, or the government. In either case, this transfer is an ancillary feature of market-based environmental regulation that is not present in the standard case of public good provision.

Goulder et al. (1998) discuss the different outcomes created by market and nonmarket mechanisms.

Goulder et al. (1997) echoes our earlier analysis and notes that environmental regulations can lead to large deadweight losses in the labor market—distinct from the out-of-pocket costs of environmental regulation—if labor taxes are high and labor supply declines (even a small amount) in response to the regulation. This is an important point to bear in mind when considering the ability to raise government revenue (via permit auctions or tax payments). Revenue raised by market-based policies can reduce the social cost of environmental policies if the new revenue is used to offset tax distortions elsewhere in the economy.

Recognition that market-based regulatory mechanisms can be used to raise government revenue, and that this revenue can be used to reduce existing taxes and lower the economy's overall tax and regulatory burden, has lead to an explosive amount of research into so-called revenue recycling. This research in turn has lent additional credibility to arguments for the auctioning of emissions permits rather than allocating them gratis.

3.7. The Double Dividend

The potential for revenue-raising environmental regulation to both reduce pollution and lower the cost of the tax system has been referred to as a double dividend. Specifically, the second dividend is the reduction in tax distortions caused by the replacement of undesirable taxes on labor and capital with environmentally beneficial taxes on pollution (Pearce 1991). However, this remains an empirical question: setting aside any environmental benefit, can the substitution of taxes on pollution for existing taxes in fact lower the distortionary cost of the tax system as a whole?

Goulder (1995b) points out that there are several versions of the double dividend. The weak version argues that the tax swap (green taxes replace conventional taxes) raises welfare relative to an identical environmental tax policy that returns the revenue lump-sum. In contrast, the strong version argues that swapping environmental taxes for a typical or representative distortionary tax (e.g., labor) is either welfare-neutral or welfare-improving relative to the *no-policy* alternative.

The weak form of the double dividend hypothesis is relatively uncontroversial. As Goulder explains, as long as existing taxes are indeed distortionary, using new revenue to reduce such distortions—versus returning it lump-sum—must reduce the distortion and therefore raise welfare relative to the case where the revenue is returned lump-sum.[34] This is the main point of Goulder et al. (1997). The stronger forms, however, have provoked considerable debate. An important insight comes from Bovenberg and de Mooij (1994), who demonstrate that the hypothesis hinges on the sign of the uncompensated elasticity of labor supply. A negative elasticity supports the hypothesis; a positive elasticity rejects it. With most empirical studies favoring positive values (Hausman 1985), this has been viewed as substantial evidence against the strong version of the double dividend hypothesis.[35]

Nonetheless, the Bovenberg and de Mooij result is based on a relatively simple model that ignores many features of the economy and the tax code. Bovenberg and van der Ploeg (1996) consider a model with involuntary unemployment and find more support for the strong double dividend. Simulation results from more complex economic models summarized in Goulder (1995b) lean against the strong version of the double dividend hypothesis, but the evidence is mixed. Most of the results indicate that substitution of environmental taxes for other taxes lowers welfare, though several indicate marginal losses or welfare improvements. Therefore, despite a general sentiment against the strong double dividend hypothesis, the question remains open.

3.8. Dynamic General Equilibrium Analysis

Static models are limited by the fact that they provide a snapshot of policy consequences at a single point in time. Since the cost of an environmental policy may vary over time, a single snapshot can be misleading. Static models are also limited by their inability to model savings and investment correctly. Savings and investment decisions reflect a trade-off among consumption in different periods of time. Since investment is a significant portion of final demand, static models may not even present an accurate snapshot of the period they attempt to describe, if there is a large impact on investment—a point made via the example at the beginning of the section.

[34] Although the efficiency properties of lump-sum versus distortionary taxes are uncontroversial, the equity properties are not. Kaplow (1996) provides an example of such concerns.

[35] A recent paper by Jaeger (2000) argues against the Bovenberg and de Mooij result but has not been widely accepted.

One approach to dynamic analysis is solving a system of linked static equilibrium models, as demonstrated by Ballard et al. (1985) and Conrad and Schröder (1991). These models involve a myopic, intertemporal trade-off where consumers assume the current return to capital investment is fixed. True dynamic general equilibrium models, however, solve these problems by simultaneously satisfying market equilibrium and budget constraints in every period.

In our model (11), three modifications are required. First, we index quantities and prices over both goods and time. Second, we have to consider utility over both goods and time, resulting in an intertemporal expenditure function. Third, our endowments **k** are fixed in the zeroeth period but otherwise evolve based on investment **i**, an additional element of final demand. The modified model becomes

$$\text{Household budget: } e(\mathbf{p} + \mathbf{t}, u) = b + (\mathbf{p}_0 + \mathbf{t}_0) \cdot \mathbf{k}_0 \tag{13}$$

$$\text{Government budget: } \mathbf{t} \cdot \mathbf{x}(\mathbf{p} + \mathbf{t}, u) = b + \mathbf{p} \cdot \mathbf{g}$$

$$\text{Zero profit: } \mathbf{p}_t \cdot \mathbf{y}(\mathbf{p}_t) = 0$$

$$\text{Market clearing: } \mathbf{y}(\mathbf{p}_t)\ \alpha_t + \mathbf{k}_t = \mathbf{g}_t + \mathbf{x}_t(\mathbf{p} + \mathbf{t}, u) + \mathbf{i}_t$$

$$\text{Capital accumulation: } \mathbf{k}_{t+1} = (\mathbf{I} - \Delta)\mathbf{k}_t + \mathbf{f}(\mathbf{i}_t),$$

where **I** is the identity matrix and Δ is a diagonal matrix of depreciation rates. Note that the household and government budget constraints are now intertemporal constraints, based on the intertemporal prices **p**. The vector-valued function **f**() describes the tranformation of investment goods into new capital. This could be a simple linear relationship describing how outputs in one period become inputs the next period. Or it could be a convex relationship representing the adjustment costs associated with rapid increases in the capital stock. Finally, we note that this general specification allows for multiple capital stocks that may be malleable between uses.

Models of the form (13) are not generally solvable using analytic means and are instead solved by numerical algorithms (Codsi et al. 1992; Lipton and et al. 1982; Rutherford 1999). Hazilla and Kopp (1990) and Jorgenson and Wilcoxen (1990) provide the best examples of dynamic general equilibrium modeling applied to environmental regulation. The main finding of those studies is that static productivity losses due to environmental regulation are amplified by the long-term effect on capital accumulation. Intuitively, environmental regulation lowers the marginal product of capital. In the long term, this leads to a lower capital stock, decreased output, and reduced welfare. The additional cost of this accumulation effect on welfare can be as much as 40% above the static cost that ignores changes in the capital stock.

When we consider the transition between unregulated and regulated regimes, assumptions about capital stock malleability become important. Models are frequently categorized as either "putty-putty" when they assume a single capital stock that can be moved costlessly between uses (see, for example, Hazilla and Kopp 1990 and Jorgenson and Wilcoxen 1990) or "putty-clay" when they assume specialized capital stocks that must be retired and replaced (Atkeson and Kehoe 1999) or incorporate convex adjustment costs (Goulder 1995a). This is especially relevant for recent applications concerning the prospective costs of climate change mitigation efforts (Jorgenson and Wilcoxen 1993; Goulder 1995a; Weyant and Hill 1999; McKibbin et al. 1999). Although Jorgenson and Wilcoxen require taxes of $25 per ton of carbon emissions in 2020 to stabilize emissions at 1990 levels, Goulder requires a tax of $50 per ton. This partially reflects Goulder's adjustment cost specification vis-à-vis the putty-putty specification in Jorgenson and Wilcoxen. The possibility of significant adjustment costs due to less malleable capital stocks emphasizes the importance of dynamic efficiency in long-term stabilization efforts, referred to as "when flexibility" by Wigley (1996).

3.9. Other "Costs" in General Equilibrium Models

The attention given to specific areas of public concern—productivity growth, employment, and trade—has led to an emphasis on these issues as alternative measures of "cost." Jorgenson and Wilcoxen (1990), for example, focus on the growth consequences associated with environmental regulation rather than welfare measures—even though their general equilibrium model can measure welfare effects. Other studies employ nongeneral equilibrium models to compute these effects. Christainsen and Havemen (1981) survey a collection of studies on the productivity slowdown of the 1970s and conclude that perhaps 8–12% of the slowdown can be attributed to environmental regulation. A more recent survey of the literature by U.S. Office of Technology Assessment (1994) concluded that as much as 44% might be attributable to environmental protection in individual industries. Looking at specific industries, Barbera and McConnell (1990) attribute 10–30% of the slowdown to environmental regulations. More recent concerns have included trade impacts (McKibbin et al. 1999) and labor consequences (Morgenstern et al. 1998).

4. Distribution of Costs

Cost and cost-benefit analyses of environmental regulation tend to focus on aggregate consequences. Yet these consequences are not evenly distributed among all members of society. For many reasons, the distribution of consequences and, specifically, costs is important.

First and foremost, most societies have some desire for equality and fairness. This is revealed by the existence of, for example, civil rights, progressive income taxes, and welfare programs. Equality can be defined in many ways and is subject to intense debate. Regardless of the definition, however, the costs associated with a particular regulation or program can be held to the definition and evaluated. The same evaluation can also be used to design compensatory schemes to redress identified inequalities.

In cases of international environmental policy, the distribution of costs across countries is a point of negotiation. Notions of equality and fairness play an important role in the design of international agreements, with the additional complication that countries participate voluntarily. This makes a careful analysis even more important, as no supranational authority exists to resolve differences.

Even without a desire for fairness, there are frequently practical considerations that motivate a focus on distribution. When costs are narrowly focused on a few stakeholders—such as restrictions on Northwest logging companies to protect wildlife habitats, or restrictions on chemical companies manufacturing chlorofluorocarbons to protect stratospheric ozone—those stakeholders are likely to be more vocal than when costs are spread over a diffuse group. It often makes sense for policymakers to work with these stakeholders to design policies that at least minimize their distress.[36]

Practical concerns often become political concerns, especially when costs are concentrated geographically. Concentration of costs may arise because of a concentration of polluting raw materials (coal mines in West Virginia), a concentration of energy-intensive or pollution-intensive productive activities (steel mills in Pennsylvania), or a concentration of consumers of energy-intensive or pollution-intensive final goods (heavy use of home heating oil in New England). All of these effects are ameliorated to the extent that environmental benefits are primarily local, raising questions of environmental federalism (Oates and Schwab 1996).

[36] These observations are attributed to Olson (1965).

Finally, environmental policies frequently transcend generations, with one generation bearing the costs and another the benefits (e.g., current mitigation efforts to prevent future global warming). Unlike the previous distributional concerns, there are currently no advocates for future generations to draw attention to equity, practical, or political issues. Nonetheless, concern over this generational equity continues to influence the policy process.

In the remainder of this section, we summarize different approaches for measuring the distribution of regulatory costs broken down by household, sector, region, and generation.

4.1. Impacts by Household

Given the pervasiveness and magnitude of environmental regulation, one would think that comprehensive studies of the cost and benefit distribution of these policies would be bountiful. Ironically, the contrary is true. What does exist is a small set of fairly straightforward analyses of price changes and the effect of those changes on household budgets, coupled with an even smaller number of sophisticated general equilibrium studies.

A recent study by Metcalf (1999) provides one kind of analysis. Metcalf uses input-output tables to construct estimates of the pollution content of different commodities purchased by consumers. Under a proposed green tax reform that would tax consumption in proportion to its pollution content, he can then estimate effective rates on each commodity. Finally, he constructs consumption bundles differentiated by income, marital status, and age based on the Consumer Expenditure Survey.[37] Combining these calculations, he is able to compute the tax burden for different demographic groups. In a report by the Congressional Budget Office (2000), similar calculations are carried out concerning the cost to different income groups of a $100-per-ton carbon tax. Bull et al. (1994) consider the impact of both a carbon tax and a Btu tax on households differentiated by income and Census region.[38]

These studies indicate that pollution taxes in isolation tend to be regressive, with poorer households spending a larger fraction of their income on environmental taxes. As Metcalf points out, compensatory reductions in income and payroll taxes targeted at poorer households can

[37] Sabelhaus (1996) discusses use of the Consumer Expenditure Survey; Lawson (1997) discusses input-output tables.

[38] There are four Census regions for individuals in urban areas (West, Northeast, South, and Midwest), plus rural.

offset the regressive effect. Of course, alternative tax reforms could further heighten the regressivity.[39]

Dowlatabadi et al. (1995) consider a different question. Instead of broad consumption bundles, they focus on differences in energy consumption. They analyze energy price increases arising from carbon taxes and trace the impact of these price increases through three alternative household energy demand models, which vary by region. Results of this exercise suggest that costs (in terms of household energy expenditures) between the lowest- and the highest-cost region differ by as much as 45%.

4.2. Households, General Equilibrium, and Social Welfare

The preceding examples provide cost estimates differentiated by household but ignore the general equilibrium considerations discussed in Section 3. General equilibrium analysis of distributional impacts is complicated by both data and computational requirements. Two approaches have emerged. One considers restricted models of household preferences that (owing to the restrictions) can be combined easily into a model of aggregate behavior (though not necessarily via a representative agent; see Jorgenson et al. 1980). The general equilibrium model can be solved using the simplified model of aggregate behavior, and then individual demand and welfare can be recovered afterward. Jorgenson et al. (1992) use this approach to study the impact of carbon taxes on the economic welfare of 1,344 distinct household types. Similarly, Aasness et al. (1996) consider the effect of environmental taxes in Norway.

A second approach is to model explicitly the behavior of distinct households. In our earlier general equilibrium model (11), we need to specify distinct endowments for each household. We then add one equation (a household budget constraint) and one endogenous variable (utility) for each additional household, leading to a modified model,

Household budgets: $e_j(\mathbf{p} + \mathbf{t}, u_j) = b_j + (\mathbf{p} + \mathbf{t}) \cdot \mathbf{k}_j$ **(14)**

Government budget: $\Sigma_j\, \mathbf{t} \cdot \mathbf{x}_j(\mathbf{p} + \mathbf{t}, u_j) = \Sigma_j\, b_j + \mathbf{p} \cdot \mathbf{g}$

Zero profit: $\mathbf{p} \cdot \mathbf{y}(\mathbf{p}) = 0$

Market clearing: $\mathbf{y}(\mathbf{p})\, z + \Sigma_j\, \mathbf{k}_j = \mathbf{g} + \Sigma_j\, \mathbf{x}_j(\mathbf{p} + \mathbf{t}, u_j)$,

[39] Both Bull et al. (1994) and Metcalf (1999) argue that the regressivity declines when one considers the lifetime incidence of these taxes.

where the definitions are identical to (11) except that household utility (u_j), expenditures (e_j), lump-sum transfers (b_j), and demand ($\mathbf{x}_j$) are now indexed over households j.

The down side to this approach is that computational constraints limit the number of distinct households. For that reason, it is not usually applied to disaggregate analysis. Such models are, however, the standard approach to multiregion, international general equilibrium analyses, discussed in the next section.

In addition to modeling the welfare of distinct households, Jorgenson et al. (1992) also consider the question of how to quantify in an internally consistent manner society's aversion to inequality. They develop a social welfare function with many desirable properties and then use this function to measure the social cost, inclusive of equity concerns.[40] Depending on the aversion to inequality, they find a welfare loss of $187 billion to $249 billion—suggesting that equity concerns alone might increase the social cost of policies by a third.

4.3. Multicountry Analysis

Modeling distributional effects across countries or regions is entirely analogous to measuring distributional effects across households. Regions have their own endowments and then trade on international markets. Thanks to recent interest in modeling mitigation efforts to reduce global climate change, a host of such disaggregated global models exist. A recent special issue of *Energy Journal* (Weyant and Hill 1999) presents results from more than a dozen models simulating the consequences of the Kyoto Protocol.

The Second Generation Model (MacCracken et al. 1999), for example, has been used to study the impact of different implementations of the Kyoto Protocol. In particular, the model is capable of estimating regional costs associated with various assumptions about international trading of emissions permits and the supply of additional emissions rights from developing countries. They demonstrate that substantial cost savings—for all countries—are possible with international permit trading. Similar analyses are carried out by McKibbin et al. (1999), with a particular emphasis on trade flows and exchange rate appreciation. They argue that the Kyoto Protocol leads to significant exchange rate appreciation in developing countries, fed by an increase in capital inflows and possibly the export of emissions credits. While boosting

[40] These properties include unrestricted domain, independence of irrelevant alternatives, positive association, nonimposition, and cardinal full comparability (see Jorgenson and Slesnick 1983).

developing country income, this has a Dutch disease effect, whereby the higher exchange rate makes their exports less competitive on world markets. They report a decline in developing countries' exports of approximately 20%. Other international models with regional detail have been used to address costs associated with capital malleability (Jacoby and Wing 1999), incentives for regional participation (Peck and Teisberg 1999), and cost-effectiveness (Nordhaus and Boyer 1999), among other topics.

4.4. Impacts by Sector

Within the United States (and we expect elsewhere), the economic analysis of a proposed environmental regulation begins with a quantification of the direct compliance cost. To the extent that most major regulations have focused on commercial activities, these compliance costs are borne by production sectors of the economy. As a consequence, production sector studies of regulatory cost are quite common, considerably more so than household-level studies.

The vast majority of these studies consider only the costs borne by the directly regulated sector(s). Since 1973, the Pollution Abatement Cost and Expenditure Report has provided detailed information on expenditures by different sectors in the United States (Bureau of the Census 1973-1997). Since 1975, the Bureau of Economic Analysis has supplemented that report with its own analysis (Vogan 1996). Germany has reported even more detailed information, beginning in 1980 (Schäfer and Stahmer 1989). These studies consistently identify the same sectors as spending the most per dollar of revenue: steel, petroleum refining, plastics, and paper, to name a few.

In many cases this is adequate for the purposes of assessing differential impact. However, in the case of large regulatory programs, especially programs that affect important economic sectors (e.g., energy), secondary impacts can be significant. In a series of papers beginning with Hazilla and Kopp (1990) and Jorgenson and Wilcoxen (1990), large-scale, multisector, computable general equilibrium (CGE) models were used to examine the costs of environmental regulation and to assess the distribution of those costs across different sectors of economy. Both Hazilla-Kopp and Jorgenson-Wilcoxen assessed the impact of the U.S. Clean Air and Clean Water acts. In addition to estimates of macroeconomic changes, these structurally similar models enabled one to model changes in total cost, output, employment, and capital accumulation resulting from environmental regulation, and to do so in a dynamic setting.

The results of these models revealed the wide range of sectoral impacts brought about by major regulatory programs. One example illustrates the point. Hazilla and Kopp report that in

1990 the output of the finance, insurance, and real estate sector was almost 5% lower than it would have been in the absence of the Clean Air and Clean Water regulations and that employment declined 2.5%. These are costs borne by a sector that had no direct compliance cost whatsoever.

New CGE models take the analysis of regulation and sectoral impacts into the realm of policy design. Current debates over global climate change and the policy measures needed to restrict the emissions of greenhouse gases acknowledge that, at least in the near term, these polices will be focused on the combustion of fossil fuels—most notably coal consumed in electricity generation. Efforts to restrict carbon emissions severely will impact the coal mining and processing sector, electricity utilities, and to some extent, other fossil energy industries. A recent study by Bovenberg and Goulder (2000) uses a CGE model to examine the impact of carbon taxes. In particular, it determines the level of compensation that would be required to compensate capital owners in the energy sector for their losses. Climate policies that generate revenues (e.g., carbon taxes or auctioned permits) could then be combined with compensation schemes to address directly the sectoral cost distribution issue.

It is important to bear in mind that not all environmental regulations have negative sectoral impacts. Cap-and-trade permit policies can generate large rents for regulated sectors if permits are allocated on a gratis basis. The point has recently been made by Bovenberg and Goulder (2000).

4.5. Impacts by Region

Many studies of distribution by household also consider regional effects. Jorgenson et al. (1992), Dowlatabadi et al. (1995), and Bull et al. (1994) all discuss impacts at broad regional levels (between four and nine regions) in the United States. More detailed impacts of broad, national policy initiatives are difficult because detailed data are lacking. Of course, case studies of narrow regulatory efforts are often sufficiently detailed to identify actual payees (e.g., Deck 1997).

A related line of work has explored whether environmental regulation influences the choice of plant location. Such choices could have significant local economic impacts that would appear in few, if any, of the cost measures discussed so far. Studies by Bartik (1988), Low and Yeats (1992), and Crandall (1993) suggest that firms are sensitive, in general terms, to cost variations among states when deciding where to locate new facilities. However, there is little direct evidence of a relationship between stringency of environmental regulation and plant location choices. In an analysis that includes measures of environmental stringency, Bartik found that neither measures of expenditures nor emissions standards had significant effects on plant location decisions. These results are similar to those of Levinson (1996) and McConnell and Schwab (1990), although Levinson did find that the locations of new branch plants of large multiplant companies in pollution-intensive industries were somewhat sensitive to differences in regulations. In contrast, a recent study by Gray (1996) finds that states with more stringent regulation (measured by a variety of state-specific measures) have fewer plant openings.

Rather than directly examining plant location decisions, other work has compared rates of manufacturing employment growth—not just new plants—in attainment areas versus nonattainment areas. Papers by Henderson (1996) and Kahn (1997) found relatively lower growth rates in manufacturing employment in nonattainment counties, compared with those that attained the air quality standard. Becker and Henderson (1997) found that environmental regulation reduced births and increased deaths in nonattainment areas, shifting polluting activity to cleaner areas. With a similar approach, Greenstone (1997) estimates an annual loss of about 8,000 jobs in nonattainment areas over the period 1972–1987. Importantly, his estimates assume that employment growth at polluting plants in less regulated areas is an appropriate control group from which to infer the likely change in employment in the absence of regulation.

Pollution control is not the only source of environmental regulation. When one looks to the future, one sees a growing demand for land use restrictions, fueled in part by environmental concerns and in part by other motivations. By their very nature, land use restrictions are local or regional and therefore might be expected to give rise to regional economic disparities—though without the controversy of federally dictated policy. For example, northern Virginia (the area nearest Washington, D.C.) is the Silicon Valley of the East. It has experienced rapid employment and income growth fed by an equally rapid commercialization of once-rural countryside. Had severe restrictions on land use been in place 15 years ago, one can argue that the current economic environment might have been negatively affected. The exact effect of current restrictions on future growth is a topic of intense debate (Webster and Wu 1999a, 1999b; Alavalapati et al. 1996). In any case, it seems likely that current restrictions will have some regional economic cost.

4.6. Intergenerational Issues

Many environmental issues, such as hazardous waste disposal, habitat and species preservation, and global climate change, involve consequences that span generations. Although standard economic theory discounts future consequences quite rapidly, ethical arguments suggest a more equal treatment.[41] Arrow et al. (1996) summarize these opposing views, but the issue remains unresolved. Recent advances in economic modeling have allowed one to model discounting within generations without imposing discounting across generations, providing a formal framework for analysis (see overlapping generations models of Howarth 1998; Bovenberg and Heijdra 1998). Again, the underlying question of how to compare intergenerational utility remains unsettled.

The significance of this issue cannot be understated, as small differences in the discount rate lead to enormous differences in valuation. This feature alone has led to the observation that ordinary market fluctuations in interest rates could lead to the use of lower rates—even distinct from the intergenerational concern (Newell and Pizer 2003; Weitzman 1998). Given the significance of intergenerational welfare in environmental policy, we leave a more complete discussion of the topic to Chapter XXX of this handbook.

5. Conclusions

Accurate analysis of the social cost of environmental regulation requires a sophisticated application of welfare economics. This analysis begins with attention to nonpecuniary costs and unmeasured consequences at the firm and household level, continues with general equilibrium effects—including tax distortions, revenue recycling, and capital accumulation—and concludes with particular attention to the distribution of costs and consequences. Each part of this analysis has important lessons to offer both the practitioner and the researcher.

Without a randomized experiment to understand the consequences of new regulation, it is impossible to speak confidently about the costs borne by firms. There are reasonable arguments suggesting that surveys and engineering studies could both under- and overestimate compliance costs. Case studies, econometric analyses, and retrospective analysis remain inconclusive, although one could interpret this to mean that such measures are, on average, unbiased.

Economy-wide analyses identify two important effects ignored by even the most thorough direct cost analysis: welfare losses associated with increased tax distortions and the potential to offset at least some of these losses with revenue-raising environmental taxes. The

[41] Ramsey (1928) called the use of anything greater than a zero percent discount rate between generations "indefensible."

former concern, trumpeted by economists for more than 30 years, remains outside the scope of ordinary policy analysts. The latter concern, recently picked up by environmental advocates (Repetto and Austin 1997), remains poorly understood outside—and sometimes among—environmental economists.

Finally, most cost analyses ignore the differential cost of environmental regulation based on demographics, especially income. Costs are not borne evenly, and many regulations appear to be regressive. When revenue from environmental taxes is used to cut incomes taxes, especially on capital, the effect is acutely regressive. The same is true when marketable permits are provided gratis to owners of capital. An interesting question is whether, given these concerns, the recent trend toward market-based regulation over command-and-control approaches improves the well-being of all consumers.

Despite the difficulties, one often hears the refrain that costs are easier to estimate than the benefits of environmental regulation, or that regulatory cost estimates are simple data collection and accounting exercises (as performed by the Department of Commerce and published in the PACE reports). This disparity of view is due, we believe, to the prevailing opinion among many in the policy community, and some trained economists as well, that the cost of environmental regulation is synonymous with the private compliance cost contained in self-reported surveys and engineering studies. The theory and evidence we present in this paper are meant to dispel that myth and to argue that, indeed, the private compliance costs may be only a fraction of the true social cost of regulation. Furthermore, unless one takes a very simplistic view of the policymaking process, the inability to measure benefits accurately does not diminish the importance of accurate cost measurements.

As we look toward the future of regulatory cost analysis, several topics stand out as both practical concerns and important research topics. Although private costs seem to be measured adequately by surveys and, to a lesser extent, engineering studies, social costs consistently deviate from these costs because of interactions with distortionary taxes in other markets as well as the effects of various recycling schemes (in the case of revenue-raising policies). Further, most environmental policies have important equity consequences—regionally, sectorally, demographically, and intertemporally—that should not be ignored. The most challenging area for future work, however, will likely be improved understanding and modeling of the process of technological change.[42]

As pollution control moves away from end-of-pipe abatement and toward pollution prevention and process changes, it becomes increasingly difficult to identify the operating costs associated with environmental protection. Even worse, as we turn to research and new technology to provide cleaner alternatives to polluting activities, the cost of these research and development activities is virtually unknown. In retrospective studies, it is difficult to know what improvements may have occurred elsewhere in the absence of environmentally focused R&D activities. In prospective studies, it is difficult to know how much improved technologies will cost.

Consider the example of global climate policy and, particularly, efforts to reduce carbon dioxide emissions. Economists would agree that in general, effective policies to limit carbon dioxide emissions should raise the private of cost of carbon emitted into the atmosphere in the short run (through the use of tax or permit systems). The price rise will stimulate conservation of carbon-containing fuels and provide incentives for the development of noncarbon energy technologies in the future. The price increases will also have a dynamic effect on the distribution of resources devoted to research and development, innovation, and commercialization, with relatively more resources going to carbon-saving research and less elsewhere. What we do not know, and have only begun to conceptualize, is the effect of this altered resource distribution on social welfare. Will resources be diverted from medical research, nanotechnology, and telecommunications? And if they are, what social gains have we lost so that we can have carbon-free energy?

Another way to view this challenge is to consider the difficulty of evaluating different outcomes characterized by large changes in relative prices (e.g., carbon and other pollutants) over long periods of time. Existing approaches have tended to emphasize marginal changes in consumption and productions, with local preferences and technology estimable from recent data. As we consider nonmarginal changes over long periods of time, new tools for cost analysis will undoubtedly be needed.

[42] This topic is the subject of Chapter XXX of this handbook.

	1992	1993	1994
Pollution Abatement and Control	104.6	110.0	121.8
Pollution Abatement	100.5	105.8	117.6
Consumers	7.9	8.4	9.8
Business	65.9	69.0	76.6
Government	26.6	28.4	31.2
Regulation and Monitoring; Research and Development	4.2	4.2	4.2
Portion of Expenditures from PACE Survey			21%
Portion of Expenditures from Government Finance Survey/Census			22%

Source: Vogan (1996), Tables 2 and 10.

Table 1: Current Spending ($Billions) on Pollution Abatement and Control

	Inelastic demand	Elastic demand, no labor tax	Elastic demand, with labor tax
"Cost" to firm	$c \cdot x_0$	$c \cdot x_1$	$c \cdot x_1$
Partial equilibrium	$c \cdot x_0$	$c \cdot \frac{1}{2}(x_1 + x_0)$	$c \cdot \frac{1}{2}(x_1 + x_0)$
GDP loss[a]	$-c \cdot x_0$ $= l_0 - l_1$	$-c \cdot x_1 + (x_0 - x_1)\, p_0$ $= l_0 - l_1$	$-c \cdot x_1 + (x_0 - x_1)\, p_0$ $= l_0 - l_1$
True welfare loss (general equilibrium)	$c \cdot x_0$	$c \cdot \frac{1}{2}(x_1 + x_0)$	$c \cdot \frac{1}{2}(x_1 + x_0) + t \cdot (l_1 - l_0)$

[a]Negative numbers indicate gains.

Table 2: Different Cost Measures and Different Modeling Assumptions

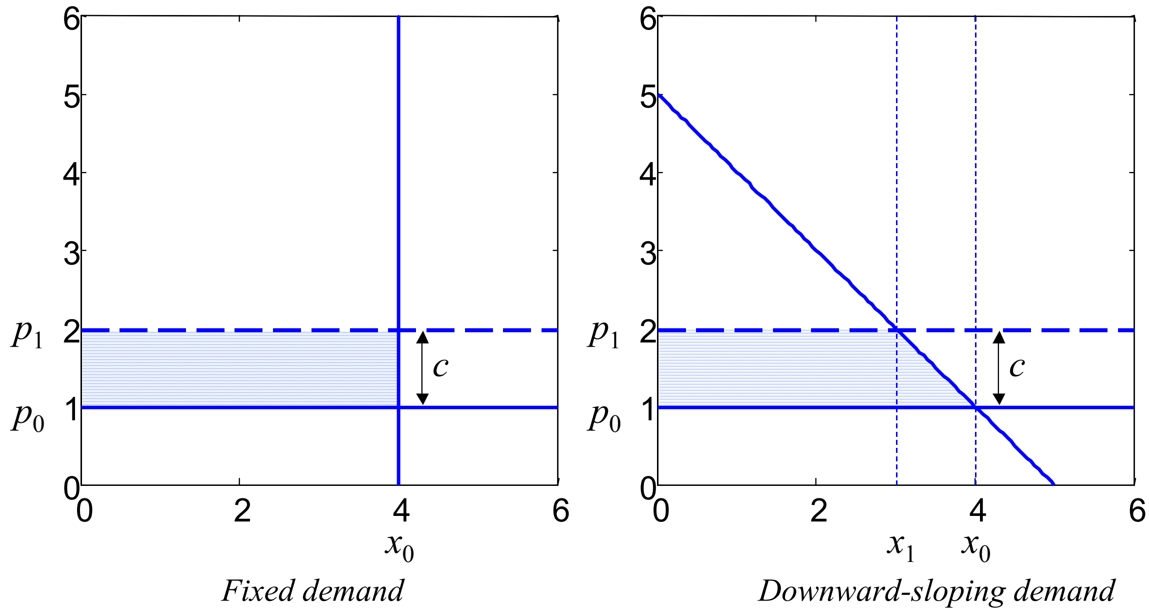

Figure 1: Effect of Environmental Controls on Consumer Surplus, No Labor Tax (shaded area shows deadweight loss)

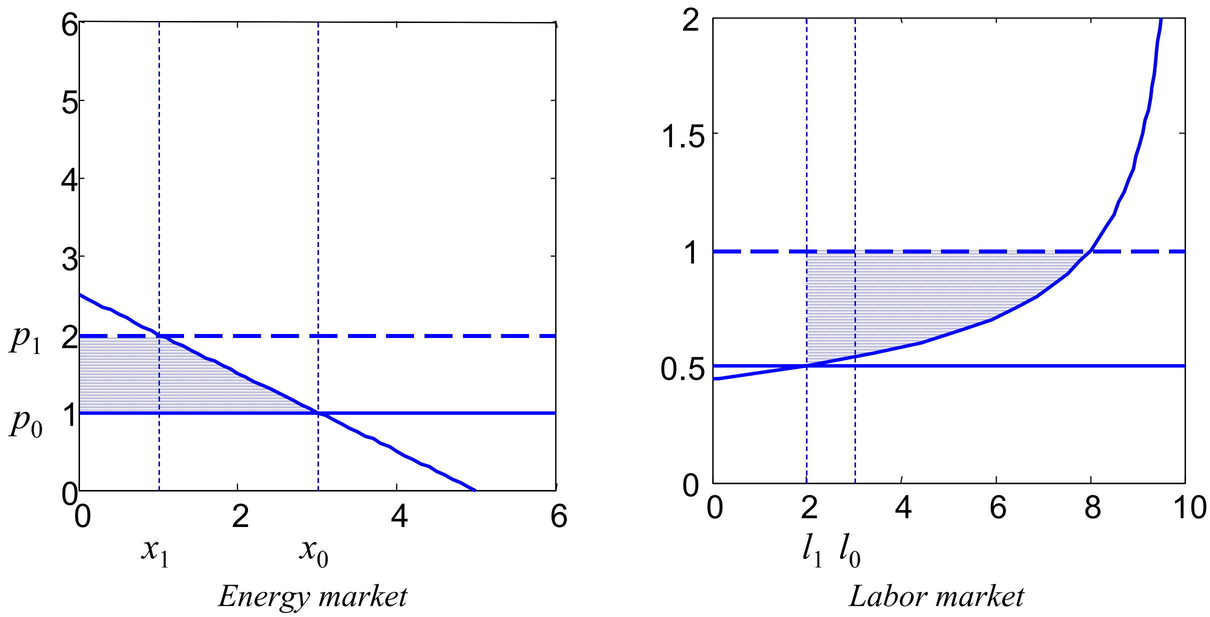

Figure 2: Effect of Environmental Regulation with Preexisting Labor Tax (shaded area shows deadweight loss)

References

Aasness, Jorgen, Torstein Bye, and Hans Terje Mysen. 1996. Welfare Effects of Emission Taxes in Norway. *Energy Economics* 18 (4):335-46.

Alavalapati, Janaki, Bill White, Pam Jagger, and Adam Wellstead. 1996. Effect of Land Use Restrictions on the Economy of Alberta: A Computable General Equilibrium Analysis. *Canadian Journal of Regional Science* 19 (3):349-65.

Arrow, K.J., W.R. Cline, K.-G. Maler, M. Munasinghe, R. Squitieri, and J.E. Stiglitz. 1996. Intertemporal Equity, Discounting and Economic Efficiency. In *Climate Change 1995 - Economic and Social Dimensions of Climate Change*, edited by J. P. Bruce, H. Lee and E. F. Haites. Cambridge, UK: Cambridge University Press.

Atkeson, Andrew, and Patrick J. Kehoe. 1999. Models of Energy Use: Putty-Putty versus Putty-Clay. *American Economic Review* 89 (4):1028-43.

Atkinson, Scott E., and Donald H. Lewis. 1974. A Cost-Effectiveness Analysis of Alternative Air Quality Control Strategies. *Journal of Environmental Economics and Management* 1 (3):237-250.

Ballard, Charles L., and Don Fullerton. 1992. Distortionary Taxes and the Provision of Public Goods. *Journal of Economic Perspectives* 6 (3):117-31.

Ballard, Charles L., and Steven G. Medema. 1993. The marginal efficiency effects of taxes and subsidies in the presence of externalities: A computational general equilibrium approach. *Journal of Public Economics* 52:199-216.

Ballard, Charles L., John B. Shoven, and John Whalley. 1985. General Equilibrium Computations of the Marginal Welfare Costs of Taxes in the United States. *The American Economic Review* 75 (1 (March)):128-138.

Barbera, Anthony J., and Virginia D. McConnell. 1990. The Impact of Environmental Regulations on Industry Productivity: Direct and Indirect Effects. *Journal of Environmental Economics and Management* 18:50-65.

Bartik, Timothy J. 1988. The Effects of Environmental Regulation on Business Location in the United States. *Growth Change* 19 (3):22-44.

Bazerlon, Coleman, and Kent Smetters. 1999. Discounting Inside the Beltway. *Journal of Economic Perspectives* 13 (4):213-228.

Becker, Randy, and Vernon Henderson. 1997. Effects of Air Quality Regulation on Decision of Firms in Polluting Industries. Cambridge, MA: NBER.

Berger, James O. 1985. *Statistical Decision Theory and Bayesian Analysis*. 2nd ed. New York: Springer-Verlag.

Bergman, Lars. 1991. General Equilibrium Effects of Environmental Policy: A CGE-Modeling Approach. *Environmental and Resource Economics* 1 (1):43-61.

Boadway, R.W. 1975. Cost-benefit Rules in General Equilibrium. *Review of Economic Studies* 42:361-374.

Böhringer, Christoph, and Thomas F. Rutherford. 1997. Carbon Taxes with Exemptions in an Open Economy: A General Equilibrium Analysis of the German Tax Initiative. *Journal of Environmental Economics and Management* 32 (2):189-203.

Bovenberg, A. Lans , and Lawrence H. Goulder. 2000. Neutralizing the Adverse Industry Impacts of CO_2 Abatement Policies: What Does It Cost? In *Behavioral and Distributional Effects of Environmental Policies: Evidence and Controversies (forthcoming)*, edited by C. Carraro and G. Metcalf. Chicago: University of Chicago Press.

Bovenberg, A. Lans, and Ruud A. de Mooij. 1994. Environmental Levies and Distortionary Taxation. *The American Economic Review* 84 (4):1085-1089.

Bovenberg, A. Lans, and Lawrence H. Goulder. 1996. Optimal Environmental Taxation in the Presence of Other Taxes: General Equilibrium Analyses. *The American Economic Review* 86 (4):985-1000.

Bovenberg, A. Lans, and Ben J. Heijdra. 1998. Environmental Tax Policy and Intergenerational Distribution. *Journal of Public Economics* 67 (1):1-24.

Bovenberg, A. Lans, and Frederick van der Ploeg. 1996. Optimal Taxation, Public Goods and Environmental Policy with Involuntary Unemployment. *Journal of Public Economics* 62 (1-2):59-83.

Boyd, James, Alan J. Krupnick, and Janice Mazurek. 1998. Intel's XL Permit: A Framework for Evaluation. Washington: Resources for the Future.

Bruce, Neil, and Richard G. Harris. 1982. Cost-Benefit Criteria and the Compensation Principle in Evaluating Small Projects. *Journal of Political Economy* 90 (4):755-76.

Bull, Nicholas, Kevin A. Hassett, and Gilbert E. Metcalf. 1994. Who Pays Broad-Based Energy Taxes? Computing Lifetime and Regional Incidence. *Energy Journal* 15 (3):145-64.

Bureau of the Census, U.S. Department of Commerce. 1973-1997. *Pollution Abatement Costs and Expenditures*. Washington, D.C.: U.S. Department of Commerce.

———. 2000. *Pollution Abatement Costs and Expenditure Survey 1999 (draft)*. Washington, D.C.: U.S. Department of Commerce.

Caves, Douglas W., and Laurits R. Christensen. 1980. Global Properties of Flexible Functional Forms. *American Economic Review* 70 (3):422-432.

Chipman, John S., and James C. Moore. 1980. Compensating Variation, Consumer's Surplus, and Welfare. *American Economic Review* 70 (5):933-49.

Christainsen, Gregory B., and Robert H. Haveman. 1981. The Contribution of Environmental Regulations to the Slowdown in Productivity Growth. *Journal of Environmental Economics and Management* 8 (4):381-390.

Cline, William. 1992. *The Economics of Global Warming*. Washington: Institute of International Economics.

Codsi, George, K. R. Pearson, and Peter J. Wilcoxen. 1992. General-Purpose Software for Intertemporal Economic Models. *Computer Science in Economics and Management* 5 (1):57-79.

Coggins, Jay S., and John R. Swinton. 1996. The Price of Pollution: A Dual Approach to Valuing SO2 Allowances. *Journal of Environmental Economics and Management* 30 (1):58-72.

Congressional Budget Office. 1985. Environmental Regulation and Economic Efficiency. Washington: Congress of the U.S., Congressional Budget Office.

Congressional Budget Office, Congress of the United States. 2000. Who Gains and Who Pays under Carbon-allowance Trading? The Distributional Effects of Alternative Policy Designs. Washington: CBO.

Conrad, Klaus, and Michael Schröder. 1991. The Control of CO2 Emissions and Its Economic Impact: An AGE Model for a German State. *Environmental and Resource Economics* 1 (3):289-312.

Council on Environmental Quality, Executive Office of the President. various years. *Environmental Quality*. Washington: GPO.

Crandall, Robert W. 1993. *Manufacturing on the Move*. Washington: Brookings Institution.

Davies, Terry. 2001. Reforming Permitting. Washington: Resources for the Future.

DeCanio, Stephen J. 1993. Barriers within Firms to Energy-Efficient Investments. *Energy Policy* 21.

Deck, Leland. 1997. Visibility at the Grand Canyon and the Navajo Generating Station. In *Economic Analyses at EPA*, edited by R. D. Morgenstern. Washington: Resources for the Future.

Diamond, Peter A., and James A. Mirrlees. 1971. Optimal Taxation and Public Production II: Tax Rules. *The American Economic Review* 61 (3):261-278.

Diewert, W. Erwin. 1983. Cost-Benefit Analysis and Project Evaluation: A Comparison of Alternative Approaches. *Journal of Public Economics* 22:265-302.

Dixit, Avinash K., and Robert S. Pindyck. 1994. *Investment Under Uncertainty*. Princeton: Princeton University Press.

Dowlatabadi, Hadi, Raymond J. Kopp, and F. Ted Tschang. 1995. Distributional and Environmental Consequences of Taxes on Energy: A Partial Equilibrium Model of U.S. Household Energy Demand. In *Advances in the Economics of Energy and Resources: Energy Use and Sustainable Economic Growth*, edited by J. Moroney. Greenwich: JAI Press.

Färe, Rolf. 1988. *Fundamentals of Production Theory*. Berlin: Springer-Verlag.

Färe, Rolf, Shawna Grosskopf, C. A. Knox Lovell, and Suthathip Yaisawarng. 1993. Derivation of Shadow Prices for Undesirable Outputs: A Distance Function Approach. *The Review of Economics and Statistics* 75 (2):374-385.

Freeman, A. Myrick, III. 1985. Methods for Assessing the Benefits of Environmental Programs. In *Handbook of Natural Resource and Energy Economics*, edited by A. V. Kneese and J. L. Sweeney. New York: North-Holland.

Gaskins, Darius W. Jr., and John P. Weyant. 1993. Model Comparisons of the Costs of Reducing CO2 Emissions. *The American Economic Review: AEA Papers and Proceedings* 83 (2):318-323.

Gollup, Frank M., and Mark J. Roberts. 1983. Environmental Regulations and Productivity Growth: The Case of Fossil-Fueled Electric Power Generation. *Journal of Political Economy* 91 (4):654-674.

Goulder, Lawrence H. 1995a. Effects of Carbon Taxes in an Economy with Prior Tax Distortions: An Intertemporal General Equilibrium Analysis. *Journal of Environmental Economics and Management* 29 (3 (Part 1)):271-297.

———. 1995b. Environmental Taxation and the Double Dividend: A Reader's Guide. *International Tax and Public Finance* 2:157-183.

Goulder, Lawrence H., Ian W. H. Parry, and Dallas Burtraw. 1997. Revenue-Raising versus Other Approaches to Environmental Protection: The Critical Significance of Preexisting Tax Distortions. *RAND Journal of Economics* 28 (4):708-31.

Goulder, Lawrence H., Ian W. H. Parry, Roberton C. III Williams, and Dallas Burtraw. 1998. The Cost-Effectiveness of Alternative Instruments for Environmental Protection in a Second-Best Setting: Resources for the Future.

Gray, Wayne. 1996. Does State Environmental Regulation Affect Plant Location. Worcester, MA: Clark University.

Gray, Wayne B. 1987. The Cost of Regulation: OSHA, EPA, and the Productivity Slowdown. *The American Economic Review* 77 (5 (December)):998-1006.

Gray, Wayne B., and Ronald J. Shadbegian. 1994. Pollution Abatement Costs, Regulation and Plant-level Productivity. Washington: Center for Economic Studies, U.S. Department of Commerce.

Greenstone, Michael. 1997. The Marginal Effects of Environmental Regulations on the Manufacturing Sector: Evidence from the 1970 and 1977 Clean Air Act Amendments. Princeton, NJ: Economics Department, Princeton University.

Gruenspecht, H. K. 1982. Differentiated Regulation: The Case of Auto Emission Standards. *American Economic Review* 72:328-331.

Gruenspecht, Howard, and Robert Stavins. 2002. A level field on pollution at power plants. *Boston Globe*, January 26.

Hahn, Robert, and Wilbur Steger. 1990. An Analysis of Jobs at Risk and Job Losses From the Proposed Clean Air Act Amendments. Pittsburgh, PA: CONSAD Research Corporation.

Harberger, Arnold C. 1971. Three Basic Postualtes for Applied Welfare Economics: An Interpretive Essay. *Journal of Economic Literature* 9:789-797.

Harrington, Winston, and Richard Morgenstern. 2000. On the Accuracy of Regulatory Cost Estimates. *Journal of Policy Analysis and Management* 19 (2):297-322.

Hartman, R.S., D. Wheeler, and M. Singh. 1997. The Cost of Air Pollution Abatement. *Applied Economics* 29:759-774.

Hausman, Jerry A. 1985. Taxes and Labor Supply. In *Handbook of Public Economics*, edited by A. J. Auerbach and M. S. Feldstein. Amsterdam: North-Holland.

Hazilla, Michael, and Raymond J. Kopp. 1990. Social Cost of Environmental Quality Regulations: A General Equilibrium Analysis. *Journal of Political Economy* 98 (4):853-873.

United States Senate. 2001. Environment and Public Works Committee. *Hearing to receive testimony on how S. 556 would affect the environment, the economy, and any improvements or amendments that should be made to the legislation.* 107th Congress. November 1.

Henderson, Vernon J. 1996. Effects of Air Quality Regulation. *American Economic Review* 86 (4):789-813.

Hicks, J.R. 1942. Consumers' Surplus and Index-Numbers. *Review of Economic Studies* 9 (2):126-137.

Hines, James R., Jr. 1999. Three Sides of Harberger Triangles. *Journal of Economic Perspectives* 13 (2):167-188.

Howarth, Richard B. 1998. An Overlapping Generations Model of Climate Economy Interactions. *Scandinavian Journal of Economics* 100 (3):575-591.

Jacoby, Henry, and Ian S. Wing. 1999. Adjustment Time, Capital Mobility and Policy Cost. *Energy Journal* Special Issue.

Jaeger, William K. 2000. Optimal Environmental Taxes. Eugene, OR: University of Oregon.

Jaffe, Adam B., Steven R. Peterson, Paul R. Portney, and Robert N. Stavins. 1995. Environmental Regulation and the Competitiveness of U.S. Manufacturing: What Does the Evidence Tell Us? *Journal of Economic* 33 (1 (March)):132-163.

Jorgenson, Dale W. 1986. Econometric Methods for Modeling Producer Behavior. In *Handbook of Econometrics*, edited by Z. Griliches and M. D. Intriligator. Amsterdam: North-Holland.

Jorgenson, Dale W., Lawrence J. Lau, and Thomas M. Stoker. 1980. Welfare Comparison under Exact Aggregation. *American Economic Review* 70 (2):268-72.

Jorgenson, Dale W., and Daniel T. Slesnick. 1983. Individual and Social Cost-of-Living Indexes. In *Price Level Measurement.*, edited by W. E. Diewert and C. Montmarquette. Ottawa: Statistics Canada.

———. 1985. General Equilibrium Analysis of Economic Policy. In *New Developments in Applied General Equilibrium Analysis.* New York and Melbourne: Cambridge University Press.

Jorgenson, Dale W., Daniel T. Slesnick, and Peter J. Wilcoxen. 1992. Carbon Taxes and Economic Welfare. In *Brookings Papers on Economic Activity*.

Jorgenson, Dale W., and Peter J. Wilcoxen. 1990. Environmental Regulation and U.S. Economic Growth. *Rand Journal of Economics* 21 (2):314-340.

———. 1993. Reducing US carbon emissions: An econometric general equilibrium assessment. *Resource and Energy Economics* 15 (1 (March)):7-25.

Joshi, S., L. Lave, J.-S. Shih, and F. McMichael. 1997. Impact of Environmental Regulations on the U.S. Steel Industry. Pittsburgh: Carnegie Mellow University.

Just, Richard E., Darrell L. Hueth, and Andrew Schmitz. 1982. *Applied Welfare Economics and Public Policy*. Englewood Cliffs: Prentice-Hall.

Kahn, Mathew E. 1997. Particulate Pollution Trends in the United States. *Journal of Regional Science and Urban Economics* 27 (1):87-107.

Kaplow, Louis. 1996. The Optimal Supply of Public Goods and the Distortionary Cost of Taxation. *National Tax Journal* 49 (4):513-533.

Kokoski, Mary F., and V. Kerry Smith. 1987. A General Equilibrium Analysis of Partial-Equilibrium Welfare Measures: The Case of Climate Change. *The American Economic Review* 77 (3):331-341.

Kolstad, Charles Durgin. 1982. Economic and Regulatory Efficiency in Air Pollution Control. PhD, Economics, Stanford University, Stanford, CA.

Krupnick, Alan J. 1986. Costs of Alternative Policies for the Control of Nitrogen Dioxide in Baltimore. *Journal of Environmental Economics and Management* 13:189-197.

Lawson, Ann M. 1997. Benchmark Input-Output Accounts for the U.S. Economy 1992. *Survey of Current Business*:36-82.

Levinson, Arik. 1996. Environmental Regulations and Manufacturers' Location Choices: Evidence from the Census of Manufactures. *Journal of Public Economics* 62 (1-2):5-29.

Lind, Robert. 1982. A Primer on the Major Issues Relating to the Discount Rate for Evaluating National Energy Options. In *Discounting for Time and Risk in Energy Policy*, edited by R. Lind. Washington: Resources for the Future.

Lipton, David, and et al. 1982. Multiple Shooting in Rational Expectations Models [The Solution of Linear Difference Models under Rational Expectations]. *Econometrica* 50 (5):1329-33.

Lovins, Amory B. 1996. Negawatts. *Energy Policy* 24 (4):331-343.

Low, Patrick, and Alexander Yeats. 1992. Do 'Dirty' Industries Migrate? In *International Trade and the Environment*. Washington: The World Bank.

MacCracken, Chris, J.A. Edmonds, S. Kim, and R. Sands. 1999. Economics of the Kyoto Protocol. *Energy Journal* Special Issue.

Mas-Colell, Andreu, Michael D. Whinston, and Jerry R. Green. 1995. *Microeconomic Theory*. New York: Oxford University Press.

McClelland, John D., and John K. Horowitz. 1999. The Costs of Water Pollution Regulation in the Pulp and Paper Industry. *Land Economics* 75 (2 (May 1999)):220-232.

McConnell, Virginia D., and Robert M. Schwab. 1990. Impact of Environmental Regulation on Industry Location Decisions: The Motor Vehicle Industry. *Land Economics* 66 (1):67-81.

McGraw-Hill. various years. Annual McGraw-Hill Survey of Pollution Control Expenditures. New York: McGraw Hill Publishing.

McKibbin, Warwick J., M. Ross, R. Shackleton, and P. Wilcoxen. 1999. Emissions Trading, Capital Flows and the Kyoto Protocol. *Energy Journal* Special Issue.

Metcalf, Gilbert E. 1999. A Distributional Analysis of Green Tax Reforms. *National Tax Journal* 52 (4):655-681.

Morgenstern, Richard D., ed. 1997. *Economic Analyses at EPA*. Washington: Resources for the Future.

Morgenstern, Richard D., William A. Pizer, and Jhih-Shyang Shih. 1998. Jobs versus the Environment: An Industry-level Perspective. Washington, DC: Resources for the Future.

———. 1999. The Cost of Environmental Protection. Washington: Resources for the Future.

Morris, Glenn E., and Duncan M. Holthausen, Jr. 1994. The Economics of Household Solid Waste Generation and Disposal. *Journal of Environmental Economics and Management* 26 (3):215-34.

Murray, Brian C., and David N. Wear. 1998. Federal timber restrictions and interregional arbitrage in U.S. lumber. *Land Economics* 74 (1):76-91.

Nelson, R.A., T. Tietenberg, and M.R. Donihue. 1993. Differential Environmental Regulation: Effects on Electric Utility Capital Turnover and Emissions. *Review of Economics and Statistics* 77 (2):368-373.

Nestor, Deborah Vaughn, and Carl A. Jr. Pasurka. 1995. Alternative specifications for environmental control costs in a general equilibrium framework. *Economics Letters* 48 (3-4):273-280.

Newell, Richard, and William A. Pizer. 2003. Discounting the Distant Future: How Much Do Uncertain Rates Increase Valuations? *Journal of Environmental Economics and Management* Forthcoming.

Nordhaus, William, and Joseph Boyer. 1999. Requiem for Kyoto: An Economic Analysis. *Energy Journal* Special Issue.

Nordhaus, William D. 1991. The Cost of Slowing Climate Change: a Survey. *The Energy Journal* 12 (1):37-65.

Oates, Wallace E., and Robert M. Schwab. 1996. The Theory of Regulatory Federalism: The Case of Environmental Management. In *The Economics of Environmental Regulation.*, edited by W. E. Oates. Cheltenham, U.K.: Elgar.

OECD. 1999. *OECD Environmental Data Compendium*. Paris: OECD.

Olson, Mancur, Jr. 1965. *The Logic of Collective of Action: Public Goods and the Theory of Groups*. Cambridge: Harvard University Press.

Palmer, Karen, Wallace E. Oates, and Paul Portney. 1995. Tightening Environmental Stantards: The Benefit-Cost or the No-Cost Paradigm. *Journal of Economic Perspectives* 9 (4 (Fall 1995)):119-132.

Pearce, David W. 1991. The Role of Carbon Taxes in Adjusting to Global Warming. *Economic Journal* 101:938-948.

Peck, Stephen, and Thomas Teisberg. 1999. CO_2 Emission Control Agreements: Incentives for Regional Participation. *Energy Journal* Special Issue.

Perl, Lewis J., and Frederick C. Dunbar. 1982. Cost Effectiveness and Cost-Benefit Analysis of Air Quality Regulations. *The American Economic Review* 72 (2):208-213.

Pindyck, Robert S. 1995. Sunk Costs and Sunk Benefits in Environmental Policy: I. Basic Theory. Cambridge, MA: MIT.

Pittman, Russell W. 1981. Issue in Pollution Control: Interplant Cost Differences and Economies of Scale. *Land Economics* 57 (1):1-17.

Pizer, William A. 1999. Optimal Choice of Climate Change Policy in the Presence of Uncertainty. *Resource and Energy Economics* 21:25-287.

Porter, Michael E., and Claas van der Linde. 1995a. Green and Competitive: Ending the Stalemate. *Harvard Business Review* 73 (5 (Sept-Oct 1995)):121-134.

———. 1995b. Toward a New Conception of the Environment-Competitiveness Relationship. *Journal of Economic Perspectives* 9 (4):97-118.

Portney, Paul R. 1981. The Macroeconomic Impacts of Federal Environmental Regulation. In *Environmental Regulation and the U.S. Economy*, edited by H. M. Peskin, P. R. Portney and A. V. Kneese. Baltimore: Johns Hopkins University Press for Resources for the Future.

Ramsey, F.P. 1928. A Mathematical Theory of Saving. *The Economic Journal* 38 (152):543-559.

Repetto, Robert, and Duncan Austin. 1997. The Costs of Climate Protection: A Guide for the Perplexed. Washington: World Resources Institute.

Rose, A. 1983. Modeling the Macroeconomic Impact of Air Pollution Abatement. *Journal of Regulatory Science* 23 (4):441-459.

Rosewicz, Barbara. 1990. Americans Are Willing to Sacrifice to Reduce Pollution, They Say. *Wall Street Journal*, April 20, 1990, A1.

Rutherford, Thomas F. 1999. Applied General Equilibrium Modeling with MPSGE as a GAMS Subsystem: An Overview of the Modeling Framework and Syntax. *Computational Economics* 14 (1-2):1-46.

Rutledge, {Gary L., and Christine R. Vogan}. 1994. Pollution Abatement and Control Expenditures 1972-92. *Survey of Current Business* 74 (5):36-49.

Sabelhaus, John. 1996. Consumer Expenditure Survey: Family-Level Extracts, 1980:1 -1994:1. Washington: Congressional Budget Office.

Samuelson, Paul A. 1974. Complementarity---An Essay on the 40th Anniversary of the Hicks-Allen Revolution in Demand Theory. *Journal of Economic Literature* 12 (4):1255-1289.

Schäfer, Dieter, and Carsten Stahmer. 1989. Input-Output Model for the Analysis of Environmental Protection Activities. *Economic Systems Research* 1 (2):203-28.

Schmalensee, Richard. 1994. The Costs of Environmental Protection. In *Balancing Economic Growth and Environmental Goals*, edited by M. B. Kotowski. Washington: American Council for Capital Formaation, Center for Policy Research.

Seskin, Eugene P., Robert J. Anderson, and Robert O. Reid. 1983. An Empirical Analysis of Economic Strategies for Controlling Air Pollution. *Journal of Environmental Economics and Management* 10 (2):112-124.

Stiglitz, J. E. 1979. Equilibrium in Product Markets with Imperfect Information. *American Economic Review* 69 (2):339-45.

Stiglitz, Joseph E., and P. Dasgupta. 1971. Differential Taxation, Public Goods and Economic Efficiency. *Review of Economic Studies* 38 (114):151-74.

Streitwieser, Mary L. 1995. Evaluation and Use of the Pollution Abatement Costs and Expenditures Survey Micro Data. Washington: Center for Economic Studies, U.S. Bureau of the Census.

Tietenberg, Tom. 1992. *Environmental and Natural Resource Economics*: HarperCollins.

———. 1998. Disclosure Strategies for Pollution Control. *Environmental and Resource Economics* 11 (3-4):587-602.

U.S. Bureau of the Census. 1997. Census of Governments, Volume 4, Number 5, Compendium of Government Finances. Washington: GPO.

U.S. Environmental Protection Agency. 1984. *Final Report: The Cost of Clean Air and Water: Report to Congress 1984*. Washington, D.C.: U.S. Environmental Protection Agency.

———. 1985a. Compilation of Air Pollutant Emissin Factors. Research Triangle Park, NC.

———. 1985b. Costs and Benefits of Reducing Lead in Gasoline: Final Regulatory Impact Analysis. Washington: Office of Policy Analysis.

———. 1990a. *Environmental Investments: The Cost of a Clean Environment*. Washington, D.C.: U.S. Environmental Protection Agency.

———. 1990b. OAQPS Control Cost Manual. Research Triangle Park, NC.

———. 1992. Regulatory Impact Analysis of Worker Protection Standard for Agricultural Pesticides. Washington: U.S. EPA. Office of Pesticide Programs.

———. 1999. *Final Report to Congress on Benefits and Costs of the Clean Air Act, 1970 to 1990*. Washington, D.C.: U.S. Environmental Protection Agency.

———. 2001. Analysis of Multi-Emissions Proposals for the U.S. Electricity Sector, Requested by Senators Smith, Voinovich, and Brownback. Washington: US EPA.

———. 2002. *New Source Review: Report to the President*. Washington, D.C.: U.S. Environmental Protection Agency.

U.S. Office of Technology Assessment. 1994. Industry, Technology and the Environment: Competitiveness, Challenges and Business Opportunities. Washington.

Vogan, Christine R. 1996. Pollution Abatement and Control Expenditures, 1972-94. *Survey of Current Business*, September.

Wald, A. 1950. *Statistical Decision Functions*. New York: Wiley.

Webster, C. J., and F. Wu. 1999a. Regulation, Land-Use Mix, and Urban Performance. Part 1: Theory. *Environment and Planning A* 31 (8):1433-42.

———. 1999b. Regulation, Land-Use Mix, and Urban Performance. Part 2: Simulation. *Environment and Planning A* 31 (9):1529-45.

Weitzman, Martin L. 1998. Why the Far-Distant Future Should Be Discounted at Its Lowest Possible Rate. *Journal of Environmental Economics and Management* 36 (3):201-208.

———. 2001. Gamma Discounting. *American Economic Review* 91 (1):260-271.

Weyant, John P., and Jennifer Hill. 1999. The Costs of the Kyoto Protocol: A Multi-Model Evaluation, Introduction and Overview. *Energy Journal* Special Issue.

Whalley, John. 1975. How Reliable is Partial Equilibrium? *Review of Economics and Statistics* 57:299-310.

Whalley, John, and Randall Wigle. 1991. Cutting CO2 Emissions: The Effects of Alternative Policy Approaches. *Energy Journal* 12 (1):109-24.

Wigley, T.M.L., R. Richels, and J.A. Edmonds. 1996. Economic and environmental choices in the stabilization of atmospheric CO_2 concentrations. *Nature* 379:240-243.