Journal of Public Economics 72 (1999) 329–360

The cost-effectiveness of alternative instruments for environmental protection in a second-best setting

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Received 31 January 1998; received in revised form 30 August 1998; accepted 30 September 1998

Abstract

This paper employs analytical and numerical general equilibrium models to examine the significance of pre-existing factor taxes for the costs of pollution reduction under a wide range of environmental policy instruments. Pre-existing taxes imply significantly higher abatement costs for every instrument considered, but the cost-impact of such taxes differs sharply across instruments. Prior taxes can eliminate the cost-advantage of market-based instruments (emissions taxes and permits) over technology mandates or performance standards, particularly if the former policies fail to generate revenues and use the revenues to finance cuts in the prior distortionary taxes. The cost differences between instruments are highly sensitive to the extent of pollution abatement: for most policies, abatement costs converge to the same value as pollution abatement approaches 100 percent.

Keywords: Environmental instrument choice; General equilibrium efficiency analysis; Second-best regulation

JEL classification: D58; H21; L51

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PII: S0047-2727(98)00109-1

1. Introduction

Environmental policy makers often must choose among alternative instruments for reducing emissions of pollution. A number of considerations affect this choice, including administrative ease, the costs of monitoring and enforcement, the probability distribution of policy errors in the face of uncertainty, effects on the distribution of income, and political feasibility.[1]

In recent years another important consideration has emerged: namely, the implications of pre-existing distortionary taxes (such as income, payroll, and sales taxes) for the costs of pollution abatement under various instruments. The potential significance of pre-existing taxes to environmental policy has been suggested by recent analyses of environmentally motivated taxes in a second-best setting. This work has shown that pre-existing factor (income) taxes tend to raise the costs of environmental tax initiatives, even when the revenues from environmental taxes are used to finance cuts in factor taxes.[2] Two welfare effects underlie these results. By driving up the price of (polluting) goods relative to leisure, environmental taxes tend to compound the factor-market distortions created by pre-existing taxes, thereby producing a negative welfare impact termed the *tax-interaction effect*. At the same time, environmental taxes whose revenues are recycled through cuts in marginal tax rates reduce the distortions caused by the pre-existing taxes, which contributes to a positive welfare impact. Typically this *revenue-recycling effect* only partly offsets the tax-interaction effect, implying that the overall impact of pre-existing taxes is to raise costs.[3]

These welfare impacts are also relevant to the choice between environmental taxes and other environmental policy instruments, and a few recent studies have taken some first steps in analyzing instrument choice in this second-best context. Parry (1997) employed an analytical model to investigate the choice between emissions taxes and emissions permits (or quotas), while Goulder et al. (1997) and Parry et al. (1998) used analytical and numerical general equilibrium models to examine this choice in the context of sulfur dioxide and carbon dioxide abatement,

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[1] An extensive literature examines how these considerations might influence instrument choice. See, for example, Hahn (1986), Nichols (1984), Stavins (1991), and Weitzman (1974).

[2] See, for example, Bovenberg and de Mooij (1994), Parry (1995), Oates (1995), and Goulder (1995a), (1995b). A related insight is that the optimal environmental tax rate in a second-best setting with distortionary taxes is typically lower than in a first-best world. See, for example, Bovenberg and van der Ploeg (1994), Parry (1995), and Bovenberg and Goulder (1996).

[3] Of course, the new revenues could be devoted to purposes other than reducing marginal rates of existing taxes. They could be used, for example, to increase government spending, to finance lump-sum tax cuts, or to reduce the deficit. These different methods of recycling the revenues can have very different efficiency consequences. In the case where emissions tax revenues are returned as lump-sum tax cuts, the revenue-recycling effect does not materialize and there is no offset to the tax-interaction effect.

respectively.[4] These studies show that while both taxes and non-auctioned emissions permits (or quotas) produce the costly tax-interaction effect, only taxes can generate the offsetting revenue-recycling effect. As a result, pre-existing taxes disproportionately raise the costs of non-auctioned permits. These studies show that the cost disadvantage of non-auctioned permits can be very important to overall efficiency. Indeed, Parry et al. (1998) estimate that reducing carbon emissions through carbon permits will be efficiency-*reducing* unless the marginal benefits from carbon abatement exceed US$18 per ton. These papers demonstrate the significance of second-best considerations in connection with the control of two important pollutants – sulfur dioxide and carbon dioxide. However, the studies examine only two policy instruments.

A recent paper by Fullerton and Metcalf (1997) expands the domain of instrument choice to include a 'technology restriction' policy (specifically, a constraint on the ratio of emissions to labor input).[5] They use an analytical model to show that initial, incremental abatement involves zero cost under the technology restriction policy, but involves strictly positive costs under the pollution-permit policy. Their model is limited, however, in that it cannot examine the costs of non-incremental abatement.

The present paper investigates how the costs and overall efficiency impacts of different environmental policy instruments are affected by pre-existing taxes and the extent of pollution abatement. Using analytical and numerical general equilibrium models, we examine the cost-effectiveness of emissions taxes, emissions permits (or quotas), fuels taxes, performance standards, and mandated technologies.

Our paper extends the prior literature on instrument choice in several ways. First, it extends prior first-best cost-effectiveness analyses (that is, analyses ignoring prior distortionary taxes[6]) by deriving both analytical and numerical results in a unified, general equilibrium framework, and by considering both

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[4] Previously, Ballard and Medema (1993) examined instrument choice in a second-best setting, employing a numerical general equilibrium model to evaluate the efficiency impacts of pollution taxes and subsidies to pollution abatement. They demonstrated that pollution taxes are generally more efficient than abatement subsidies. The difference is due in part to the pollution tax's ability to raise revenue that can be used to reduce marginal rates of existing taxes. This fiscal issue is reflected in the *revenue-recycling effect* discussed below.

[5] They also consider a policy that subsidizes all goods in the economy other than a polluting good. In their model this is formally equivalent to a tax on the polluting good.

[6] Throughout this paper, the term 'first-best' refers to a setting with no prior distortionary taxes. Strictly speaking, the absence of distortionary taxes does not guarantee a first-best situation: environmental resources, in particular, may not be efficiently priced or regulated. However, we find it useful to apply this label to provide a contrast with the second-best setting involving prior distortionary taxes.

incremental and 'large' pollution reductions.[7] The attention to both small and large changes is important: we show that the relative costs of different instruments depend importantly on the extent of abatement. Indeed, while relative costs differ significantly at low levels of abatement, for all instruments except for the fuel tax, the costs per unit of abatement converge to the same value as the extent of abatement approaches 100 percent.

Second, the paper extends the emerging literature on instrument choice in a second-best setting. In contrast with the studies by Parry, by Goulder et al., and by Parry et al., we consider not only emissions taxes and permits but also technology mandates, performance standards, and fuel (input) taxes. And in contrast with Fullerton and Metcalf (1997), we consider abatement beyond the initial increment (as well as some additional instruments). We find that pre-existing taxes raise the costs of abatement under each of the instruments considered, relative to its costs in a 'first-best' (no-prior-tax) world. This extra cost is an increasing function of the magnitude of pre-existing tax rates, and for plausible tax rates and parameter values the cost increase is substantial (27 percent or more). Importantly, pre-existing taxes do not raise the costs of all instruments by the same proportion, and thus prior taxes influence the relative attractiveness of different instruments. The cost-impact of pre-existing taxes is particularly large for (non-auctioned) emissions permits; here the cost increase can be several hundred percent. This result has important policy implications. Economists have long argued that tradable emissions permits and emissions taxes are more cost-effective than performance standards, technology mandates, and other traditional forms of regulation (see, for example, Cropper and Oates, 1992). Our results suggest that tradable emissions permits will yield substantial cost savings over performance standards or technology mandates only if the permits are auctioned, with revenues used to cut other taxes.

Finally, the paper applies the numerical model to analyze nitrogen oxide (NO_x) emissions regulation under the 1990 Clean Air Act Amendments. These regulations have relied primarily on performance standards and are expected to yield emission reductions of approximately 20 percent in 2005 relative to the forecasted baseline emissions. Our results suggest that pre-existing taxes raise the cost of this reduction by about US$1.3 billion (1990 dollars) in the year 2005. At the same time, our results cast doubt on the possibility of achieving the same emissions reductions at lower cost under a policy of tradable emissions permits. In addition to investigating policy costs in the context of NO_x regulation, we consider alternative scenarios in order to bring out general principles applicable to other pollutants.

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[7] The most extensive prior analysis of a broad range of instruments appears to be Spulber (1985). That study involves only analytical methods and (accordingly) considers only small changes. Tietenberg (1985) and Lester et al. (1997) (chapt. 7) also consider a large set of instruments, but these studies do not offer a unified general-equilibrium analysis.

A caveat is in order. Our analysis abstracts from heterogeneity in firms' abatement cost schedules and associated information problems faced by regulators. These issues can be very important to the choice of policy instruments and should be considered along with the second-best issues investigated here.

The paper is organized as follows. Section 2 presents an analytical model that reveals the different efficiency impacts of the policy instruments. This model is extended in later sections to introduce more realism and gauge the empirical importance of these differences in efficiency. Section 3 presents the extended model, which is solved numerically. Section 4 provides results from simulations with the extended model. The final section offers conclusions.

2. The analytical model

This section uses an analytical model to compare the costs of environmental policy instruments in the presence of distortionary taxes. Detailed derivations for all of the results below are provided in a separate mathematical appendix available from the authors on request.

2.1. Model assumptions

We develop a static model in which a representative household enjoys utility from a polluting consumption good (X), a non-polluting consumption good (Y), and leisure, with leisure equal to the household time endowment ($\bar{L}$) less labor supply (L). Emissions (E) from producing X cause environmental damages that reduce consumer utility. The household utility function is given by:

$$U = u(X,Y,\bar{L} - L) - \phi(E), \tag{2.1}$$

where $u(.)$ is utility from non-environmental goods and is quasi-concave, and $\phi(.)$ is disutility from waste emissions and is weakly convex. The separability assumption in (2.1) implies that the demands for X and Y and the supply of labor do not vary with changes in E.[8] X and Y are produced by competitive firms using

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[8] Relaxing this assumption would complicate the tax-interaction effect discussed below. If, compared with consumption, leisure is a relatively strong (weak) substitute for environmental quality, then this effect is weakened (strengthened). There is little empirical evidence on the relative ease of substitution between leisure, overall consumption, and environmental quality. Under these circumstances it seems reasonable to assume separability, which implies that changes in environmental quality do not affect the relative attractiveness of consumption and leisure. For a discussion of the significance of the separability assumption, see Espinosa and Smith (1995).

labor, which is the only factor of production.[9] We assume that the marginal product of labor is constant in each industry and normalize output to imply marginal products (and a wage rate) of unity. This normalization implies that the unit cost of producing X or Y is unity.

Firms can reduce waste emissions per unit of output through 'end-of-pipe' treatment, which requires the purchase of abatement equipment or services. We assume that such equipment or services are produced directly from labor. Emissions per unit of X are $e_0 - a$, where e_0 is baseline emissions per unit (that is, emissions per unit in the absence of regulation) and a is the reduction in per-unit emissions from utilizing abatement equipment or services. Economy-wide emissions, E, are therefore equal to $(e_0 - a)X$. Thus, total emissions fall as a result of reduced production of X (the *output-substitution effect*) and increased abatement activity (the *abatement effect*).

The total cost (C) of abatement activity to the firm is given by:

$$C = c(a)X, \tag{2.2}$$

where $c(a)$ is a convex function representing the per-unit cost of abatement activity.

The government levies a proportional tax of t_{L} on labor earnings, regulates emissions, and provides a lump-sum transfer G to households. G denotes the nominal transfer; we assume that the government adjusts G so that real transfers are held constant for all policy changes. We also assume government budget balance; any revenue consequences of environmental policies are offset by adjusting t_{L}.

The household budget constraint is:

$$p_X X + Y = (1 - t_{\mathrm{L}})L + G, \tag{2.3}$$

where p_X is the demand price of X (equal to unity in the absence of regulation). Households choose X, Y and L to maximize utility (2.1) subject to the budget constraint (2.3), taking environmental damages as given. From the resulting first-order conditions and (2.3) we obtain the uncompensated demand and labor supply functions:

$$X(p_X, 1 - t_{\mathrm{L}}, G), \quad Y(p_X, 1 - t_{\mathrm{L}}, G), \quad L(p_X, 1 - t_{\mathrm{L}}, G). \tag{2.4}$$

Substituting these equations into (2.1) gives the indirect utility function:

$$V = \mathrm{v}(p_X, 1 - t_{\mathrm{L}}, G) - \phi(E). \tag{2.5}$$

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[9] In an extended model that considered other primary factors (such as capital), our quantitative results would vary to the extent that pre-existing taxes on these factors differed and the environmental policy imposed a different burden on these factors. This issue is examined in detail in Bovenberg and Goulder (1997).

Throughout this paper the term ‘costs’ refers to the gross efficiency costs of environmental policies, that is, the costs before netting out environment-related benefits from changes in $\phi(.)$.

We now define two concepts relating to the burden of the labor tax. These will be useful in expressing the tax-interaction effect. First, define:

$$M \equiv \frac{-t_L(\partial L/\partial t_L)}{L + t_L(\partial L/\partial t_L)}. \tag{2.6}$$

This is the (partial equilibrium) efficiency cost from raising an additional dollar of labor tax revenue in order to finance government consumption expenditure. It equals the *marginal cost of public funds* minus one.[10] The numerator of (2.6) is the efficiency loss from an incremental increase in t_L. This equals the wedge between the gross wage (the value marginal product of labor) and the net wage (the marginal social cost of labor in terms of foregone leisure), multiplied by the reduction in labor supply. The denominator is marginal labor tax revenue (from differentiating $t_L L$).

The corresponding expression when revenue is returned to households as lump-sum transfers is the partial equilibrium *marginal excess burden* of the labor tax, given by:

$$M' \equiv \frac{-t_L(\partial L^C/\partial t_L)}{L + t_L(\partial L/\partial t_L)} = \frac{\varepsilon^C t_L/(1-t_L)}{1-\varepsilon^U t_L/(1-t_L)}, \quad \varepsilon^C \equiv \frac{\partial L^C}{\partial(1-t_L)}\frac{1-t_L}{L},$$

$$\varepsilon^U \equiv \frac{\partial L}{\partial(1-t_L)}\frac{1-t_L}{L}, \tag{2.6$'$}$$

where the superscript C denotes a compensated derivative and ε^C and ε^U denote the compensated and uncompensated labor supply elasticities.[11] Below we provide and interpret key equations that decompose the efficiency cost of the various policies.

2.2. Emissions tax (with revenues returned through cuts in distortionary tax rates)

Consider a revenue-neutral tax of t_E imposed on emissions, with revenues from

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[10] It is a partial equilibrium concept because it does not take into account the indirect effect of the labor tax on emissions and emissions tax revenue. We assume that M is positive, in keeping with evidence that the uncompensated labor supply for the whole economy is positive (see below).

[11] In this case, there is an income effect from the increased government transfers to households, and thus expression (2.6′) depends on both the compensated and uncompensated derivatives of labor supply, whereas (2.6) depends only on the uncompensated derivative. Browning (1987) provides a comprehensive discussion of the formula in (2.6′). For a discussion of the relationship between the marginal excess burden and marginal cost of public funds, see Hakonsen (1998).

this tax used to finance cuts in the distortionary tax, t_L. The government budget constraint is:

$$t_E E + t_L L = G, \tag{2.7}$$

that is, revenues from the emissions tax and (reduced) labor tax exactly finance the given level of government spending. The profit per unit of X is:

$$p_X - \{1 + c(a) + t_E(e_0 - a)\}, \tag{2.8}$$

and in equilibrium profits are zero. The emissions tax raises the marginal cost (and thus the price) of X because it induces firms to incur abatement costs $c(a)$ and because it exacts a tax payment of $t_E(e_0 - a)$. Firms choose a, the emissions abatement per unit of X, to maximize profits. This gives the first-order condition:

$$t_E = c'(a). \tag{2.9}$$

Eq. (2.9) states that abatement activity occurs until the marginal abatement cost per unit of X equals the emissions tax rate. Eqs. (2.4) and (2.9) imply $E = E(t_E,t_L)$.

We now consider an incremental, revenue-neutral increase in t_E. The efficiency cost of this policy can be expressed as:

$$\begin{aligned}-\frac{1}{\lambda}\frac{d\nu}{dt_E} = {} & \underbrace{c'(a)\left(\frac{da}{dt_E}\right)X}_{dW^A} + \underbrace{\left(-\frac{dX}{dt_E}\right)t_E(e_0-a)}_{dW^O} - \underbrace{M\left\{E+t_E\frac{dE}{dt_E}\right\}}_{\partial w^R} \\ & + \underbrace{\left[(1+M)t_L\left(-\frac{\partial L}{\partial p_X}\right)+M'Xs_G\right]\frac{dp_X}{dt_E}}_{\partial W^I},\end{aligned} \tag{2.10}$$

where λ is the marginal utility of income and s_G is the share of government transfers in household income, given by $s_G = G/[G + (L - t_L)L]$. The first two terms on the right-hand side comprise the *primary cost* of this policy. The term labeled dW^A represents the cost from the *abatement effect*. This is the efficiency cost associated with firms' expenditure on end-of-pipe abatement activities. The term labeled dW^O represents the cost from the *output-substitution effect*. This is the efficiency cost associated with households substituting away from (the now higher-priced) X to other goods and leisure. This effect equals the reduction in consumption of X multiplied by the increase in marginal production cost of X caused by the emissions tax. (In the expanded model of Section 3, which includes intermediate inputs, a third channel for emissions reduction – the *input-substitution effect* – also applies.)

Table 1 summarizes the extent that these effects are utilized under the emissions tax and other policies. The emissions tax exploits all of these effects 'fully' in that it induces the most cost-effective level of adjustment along each of these dimensions. As discussed below, under the other policies some of the

Table 1
Determinants of primary costs by policy

Instrument	Abatement effect	Input-substitution effect	Output-substitution effect
Emissions tax	Full	Full	Full
Emissions permits	Full	Full	Full
Performance standard	Full	Full	Partial
Technology mandate	Full	Partial	Partial
Fuels tax	None	Full	Full

dimensions are exploited either partially or not at all, which implies higher primary costs to achieve given emissions abatement targets.

In a first-best (no-prior-tax) setting, the relative cost-effectiveness of different policies can be explained fully in terms of the differences in primary costs. But in a second-best setting (with pre-existing distortionary taxes), two additional cost terms come into play. These are represented by ∂W^{R} and ∂W^{I} in (2.10). ∂W^{R} is the efficiency gain from using the additional emissions tax revenue to finance cuts in the distortionary labor tax. This is the (marginal) *revenue-recycling effect*. It equals the product of the marginal cost of public funds minus one and the marginal revenue from the emissions tax. ∂W^{I} is the efficiency loss from the *tax-interaction effect*. The emissions tax increases the price of X, implying an increase in the price of consumption and thus a reduction in the real wage. This reduces labor supply and produces a marginal efficiency loss of $t_{L}(-\partial L/\partial p_{X})(\mathrm{d}p_{X}/\mathrm{d}t_{E})$, which is the tax wedge between the gross and net wage multiplied by the reduction in labor supply. The reduction in labor supply also reduces tax revenues. The efficiency cost of replacing the lost revenue is M times the lost tax revenues. Finally, as the price of consumption rises, nominal government spending must increase in order to hold real spending constant. The cost of raising this extra revenue is M' times the change in nominal government spending, Xs_{G}. The combined efficiency loss from these three sources is the tax-interaction effect. As discussed below, the tax-interaction effect usually dominates the revenue-recycling effect: pre-existing distortionary taxes raise the costs of a given emissions tax, even when revenues are recycled through cuts in these prior taxes.

2.3. Emission permits

Now consider the impact of a set of (non-auctioned) tradable emission permits. In this model, with homogeneous producers, this policy is equivalent to one where the government chooses an overall acceptable level of emissions and allocates emissions permits so that each firm is entitled to a level of emissions proportional to its baseline emissions. The key difference between this policy and the emissions tax is that it does not raise revenues for the government, and that, consequently, the policy does not allow for a reduction in the distortionary tax, t_{L}. If emissions permits were auctioned by the regulator, their effects would be identical in this model to those of an emissions tax.

Under the non-auctioned permit policy, the government budget constraint is:

$$t_L L = G. \tag{2.7a}$$

The permit policy can be represented as a virtual tax on emissions, where the ‘revenues’ from this tax are rebated to firms in lump-sum fashion. Such revenues correspond to the rents generated by the permit policy.[12] We use t_E^v to denote the virtual tax. The cost of an incremental increase in the virtual tax can be decomposed as follows:

$$-\frac{1}{\lambda}\frac{dv}{dt_E} = \underbrace{c'(a)\left(\frac{da}{dt_E^v}\right)X}_{dW^A} + \underbrace{\left(-\frac{dX}{dt_E^v}\right)t_E^v(e_0-a)}_{dW^O} + \underbrace{\left[(1+M)t_L\left(-\frac{\partial L}{\partial p_X}\right)+M'Xs_G\right]\frac{dp_X}{dt_E^v}}_{\partial W^I}. \tag{2.10a}$$

A comparison of (2.10a) with (2.10) reveals that the permit involves a primary cost (dW^A plus dW^O) analogous to that under the emissions tax. In a first-best setting, where only the primary costs matter, the tax and permit policies have equivalent efficiency impacts (Table 1).

The permit policy also induces a similar tax-interaction effect (∂W^I) because, like an emissions tax, it drives up the price of consumption goods and reduces the real wage. The key difference from the emissions tax is the absence of the revenue-recycling effect, which implies that a given emissions reduction is more costly under the non-auctioned permit policy than under the emissions tax.[13]

2.4. Fuel tax

Next, consider a revenue-neutral tax of t_X per unit on the production of X. We refer to this as a fuel tax, implicitly regarding the pollution-related good X as a fuel. The general equilibrium cost of an incremental increase in the fuel tax is:

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[12] The limited supply of emissions permits implies reduced output, which gives rise to economic rents.

[13] Parry (1997) and Goulder et al. (1997) examine in detail the significance of the revenue-recycling effect to the relative costs of permits and taxes.

$$-\frac{1}{\lambda}\frac{\mathrm{dv}}{\mathrm{d}t_X}=\underbrace{t_X\left(-\frac{\mathrm{d}X}{\mathrm{d}p_X}\right)}_{\mathrm{d}W^{\mathrm{O}}}-\underbrace{M\left(X+t_X\frac{\mathrm{d}X}{\mathrm{d}t_X}\right)}_{\partial W^{\mathrm{R}}}$$

$$+\underbrace{(1+M)t_{\mathrm{L}}\left(-\frac{\partial L}{\partial p_X}\right)+M's_{\mathrm{G}}X}_{\partial W^{\mathrm{I}}}. \qquad (2.10b)$$

This expression differs from (2.10), which applies to the emissions tax, in that the abatement effect is missing. Under this policy, profits per unit of X are:

$$p_X-\{1+t_X+c(a)\}. \qquad (2.8a)$$

The first-order conditions for profit maximization imply $a=0$. Because this policy does not raise the price of emissions, it gives firms no incentive to engage in end-of-pipe abatement expenditure and the policy generates only the output-substitution effect. For this reason the fuel tax fails to generate the efficient emissions-output ratio. As demonstrated in the numerical simulations below, the primary cost of the fuel tax exceeds that of the emissions tax and permit, for a given level of emissions reduction. This reflects the fuel tax's inability to exploit the abatement effect. The fuel tax also generates the tax-interaction and revenue-recycling effects because it increases the price of good X and raises revenue. Whether the fuel tax is more or less costly than the emissions permit depends on whether the fuel tax's failure to exploit the abatement effect is as important as the permit's failure to exploit the revenue-recycling effect. The numerical results of Section 4 will show that these relative costs depend importantly on the level of abatement.

2.5. Command-and-control policies

The literature on environmental regulation distinguishes two main types of command-and-control (CAC) policies: technology mandates and performance standards. Technology mandates require firms to adopt a specific pollution abatement technology, such as a catalytic converter on new cars or desulfurization equipment on new coal-fired power plants. Performance standards require firms to achieve a ratio of emissions to a measure of input or output that does not exceed a given maximum; examples here are the standards for NO_x emissions from power plants, which limit the ratio of such emissions to energy input. In the present analytical model, spending on abatement is the only way to reduce emissions per unit of output. Thus, technology mandates and performance standards are equivalent here. For now we refer to these policies as the technology mandate, but it should be kept in mind that the results here hold for the performance standard as well. The extended model incorporates intermediate inputs and thus allows input substitution, permitting us to distinguish the two types of policies.

If the policy mandated by the government is the most cost-effective among the available technology alternatives, we will refer to the policy as the least-cost mandated technology. If instead the regulators compel firms to adopt a technology that is not the most cost-effective, we will label the policy as a higher-cost mandated technology. We employ the parameter θ to represent the proportion by which the cost of the higher-cost technology mandate exceeds that of the least-cost mandate. Specifically, we represent the cost of the abatement effect under technology mandate by $\theta c(a)$, where $\theta = 1$ under the least-cost policy and $\theta > 1$ under the higher-cost policy.[14]

The general equilibrium cost of the technology mandate is:

$$-\frac{1}{\lambda}\frac{\mathrm{d}v}{\mathrm{d}a} = \underbrace{\theta c'(a)X}_{\mathrm{d}W^{\mathrm{A}}} + \underbrace{\left[(1+M)t_{\mathrm{L}}\left(-\frac{\partial L}{\partial p_X}\right) + M' s_{\mathrm{G}} X\right]\frac{\mathrm{d}p_X}{\mathrm{d}a}}_{\partial W^{\mathrm{I}}}. \tag{2.10c}$$

The cost is higher than under the emissions tax for two main reasons. First, this policy involves a higher primary cost because it does not fully utilize the output-substitution effect, $\mathrm{d}W^{\mathrm{O}}$.[15] Under the technology mandate, the regulator specifies both the type of technology, indexed by θ, and the required level of end-of-pipe abatement per unit of output, indicated by a. Profits per unit of X are:

$$p_X - \{1 + \theta c(a)\}. \tag{2.8b}$$

Unlike the emissions tax, the technology mandate does not charge firms for their 'residual emissions' – the per-unit emissions that the firm continues to generate after the policy is introduced. In contrast, under the emissions tax, if a firm produces another unit of output, it incurs not only the cost of the additional inputs and additional abatement expense but also the tax on residual emissions. Thus, when $\theta = 1$ the price of output will be lower under the technology mandate than under the emissions tax, as is evident from comparing the unit cost of good X under the technology mandate and under the emissions tax (the bracketed terms in (2.8) and (2.8b)). Because the price of output is too low under the technology

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[14] This is an ad hoc assumption because θ is imposed exogenously rather than determined by the model. We include the higher-cost policy as a reminder that the compliance costs of command-and-control policies are likely to be significantly higher in a more general model that captures heterogeneity in the costs of emissions reduction among firms and imperfect information among regulators about firms' costs of abatement.

[15] As indicated by (2.10c), the output-substitution effect is absent for incremental changes in a. However, this effect is non-zero for larger changes.

mandate, it does not fully exploit the output-substitution effect, and this contributes to higher primary costs.

In addition, the technology mandate involves higher second-best costs than those under the emissions tax. For a small or moderate θ, the output price is lower under the technology mandate than under the emissions tax, so the tax-interaction effect is smaller. However, since the revenue-recycling effect is absent, the overall second-best costs are higher. Although both primary costs and second-best costs are higher, the ratio of these costs is the same as the corresponding ratio under the emissions tax, as discussed below. Thus, the higher costs of the technology mandate reflect both higher primary costs and higher second-best costs.

2.6. The cost impact of pre-existing taxes

We now explore how pre-existing taxes affect the overall costs of emissions reductions under each policy. Here we compare overall efficiency costs (including second-best effects) to the primary costs alone. While previous subsections observed only incremental changes, here we consider large changes. We assume demand, supply and marginal cost curves are linear in order to obtain second-order cost approximations. These restrictions are avoided in the numerical simulations presented in Section 4.

2.6.1. Emissions tax

The ratio of the general equilibrium cost of the emissions tax relative to the primary cost is:

$$\frac{\Delta W^{\mathrm{A}} + \Delta W^{\mathrm{O}} - \Delta W^{\mathrm{R}} + \Delta W^{\mathrm{I}}}{\Delta W^{\mathrm{A}} + \Delta W^{\mathrm{O}}} = 1 + M. \tag{2.11}$$

Since $M > 0$, this term will be greater than 1, indicating that the costs associated with the tax-interaction effect are only partly offset by the revenue-recycling effect. Thus, despite the fact that the revenues from the emissions tax are devoted to cuts in distortionary taxes, pre-existing taxes raise the policy's costs. This is the case because the policy raises revenues from a narrow-based (emissions) tax at the expense of revenues from a broad-based (labor) tax, and narrow-based taxes involve higher costs.[16]

Under this policy, the net efficiency impact from the combination of the

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[16] This is now a familiar result. Thus, given the initial conditions specified here, such reforms yield an environmental 'dividend' but not a double (i.e., second) dividend in the form of a reduction in the overall efficiency cost of the tax system. For more discussion, see the surveys by Oates (1995), Goulder (1995b), and Bovenberg and Goulder (1997) and the references therein. Formula (2.11) assumes that X and Y are equal substitutes for leisure. If X were instead a relatively weak substitute for leisure, the revenue-recycling effect could exceed the tax-interaction effect (Parry, 1995).

tax-interaction and revenue-recycling effects is proportional to primary cost. The magnitude of the tax-interaction effect depends on the change in output price, which in turn is determined by the primary cost and the charge on residual emissions. The size of the revenue-recycling effect depends on the charge on residual emissions. When the revenue-recycling effect is subtracted from the tax-interaction effect, the contributions from the charge on residual emissions cancel, and thus the net efficiency impact is proportional to primary cost.

Table 2 offers an intuitive summary of the contribution of the second-best effects to the overall costs of the emissions tax and other policies. Because the emissions tax applies to residual emissions, it leads to a 'large' impact on output prices, and the tax-interaction effect is large. The revenue-recycling effect is large as well. The overall second-best contribution is described as moderate, because the revenue-recycling effect offsets much of the tax-interaction effect.

2.6.2. Emissions permits

The general equilibrium cost of the permit policy expressed relative to the primary cost is:

$$\frac{\Delta W^{\mathrm{A}} + \Delta W^{\mathrm{O}} + \Delta W^{\mathrm{I}}}{\Delta W^{\mathrm{A}} + \Delta W^{\mathrm{O}}} = 1 + M + \frac{2M'[1 - (\Delta E/E_0)]}{\Delta E/E_0}. \tag{2.11a}$$

This expression exceeds $1+M$ in (2.11) so long as the proportionate emissions reduction, $\Delta E/E_0$, is less than unity. Thus, interactions with the tax system raise the cost of the permit proportionately more than they raise the costs of the emissions tax because non-auctioned permits do not generate the efficiency benefit from the revenue-recycling effect. In fact, using our benchmark assumptions of $t_{\mathrm{L}} = 0.4$, $\varepsilon^{\mathrm{C}} = 0.4$ and $\varepsilon^{\mathrm{U}} = 0.15$ (see below), Eq. (2.11a) implies that when the proportionate reduction in emissions is 10 percent, the general equilibrium costs of emissions permits are 6.5 times the primary costs! Thus, as indicated in Table 2,

Table 2
Second-best contributions to overall costs

Instrument	Impact on output price	Tax-interaction effect[a]	Revenue-recycling effect[a]	Proportional net second-best contribution[a]
Emissions tax[b]	Large	Large	Large	Moderate
(Non-auctioned) emissions permits	Large	Large	0	Potentially huge
Performance standard	Moderate	Moderate	0	Moderate
Technology mandate	Moderate	Moderate	0	Moderate
Fuels tax[b]	Large	Large	Large	Moderate

[a] Relative to primary cost.

[b] It is assumed that revenues from the tax are used to finance cuts in rates of pre-existing taxes. See text for discussion.

the contribution of prior taxes to the costs of emissions permits is potentially huge. However, in the limit as the proportionate emissions reduction approaches unity, the cost discrepancy between the emissions permit and emissions tax declines to zero. This happens because at 100 percent emissions reduction the emissions tax generates no revenue and hence no revenue-recycling effect.

2.6.3. Fuel tax

The ratio of general equilibrium costs to primary costs under the fuel tax is again $1 + M$. Just as under the emissions tax, there is a large tax-interaction effect and a large and partly offsetting revenue-recycling effect. Thus, the proportional net impact of the tax-interaction and revenue-recycling effects is the same as under the emissions tax, as summarized in Table 2. However, since primary costs are higher under the fuel tax (as discussed in Section 2.4 above), the overall costs (including second-best costs) are higher.

2.6.4. Technology mandate

The general equilibrium cost of the technology mandate expressed relative to the primary cost is:

$$\frac{\Delta W^{\mathrm{A}} + \Delta W^{\mathrm{I}}}{\Delta W^{\mathrm{A}}} = 1 + M. \tag{2.11b}$$

The proportional increase in cost under this policy is the same as that under the emissions tax, which is lower than that under the emissions permit. Like the emissions permit, the technology mandate does not generate the revenue-recycling effect. However, this is compensated for by the fact that the tax-interaction effect is relatively weaker than under the emissions tax and permit. Thus the overall second-best impact under this policy is moderate (Table 2). Note that θ does not enter (2.11b): although θ directly affects the primary costs of a CAC policy, it has no effect on the ratio of overall costs to primary costs.

2.6.5. Summary

This analysis has decomposed the overall efficiency costs of different instruments into primary costs (reflecting the abatement and output-substitution effects) and the second-best tax-interaction and revenue-recycling effects. In a first-best setting, the emissions tax and permit policies have identical primary costs. The fuel tax and technology mandate have higher primary costs because they do not fully exploit the potential channels for emissions reduction.

In a second-best setting, the cost-rankings change significantly. Second-best interactions substantially raise the cost of the permit policy relative to other policies because the permit policy does not generate the revenue-recycling effect to counteract the tax-interaction effect. The technology mandate does not produce the revenue-recycling effect either, but this is compensated for by the fact that it produces a relatively weak tax-interaction effect. At ‘low’ levels of emissions abatement, the marginal revenue-recycling effect is very large and becomes an especially important determinant of relative policy costs.

3. The numerical model

We now extend the previous model by incorporating intermediate inputs in production. This yields a new channel for emissions reduction: now emissions can be reduced not only through output-substitution and abatement effects but also by way of the *input-substitution effect* – altering the mix of intermediate inputs. Incorporating intermediate goods provides greater realism and allows the performance standard and technology mandate to have different economic impacts.

The numerical model distinguishes two intermediate goods: a polluting intermediate good (D) and a ‘clean’ intermediate good (N). As before, there are two final consumption goods: C_N represents final output from industries that use N relatively more intensively, and C_D is final output from industries that use D relatively more intensively. The extended model is solved numerically to obtain ‘exact’ welfare assessments, in contrast with the second-order approximations obtained above. This is potentially important for ‘large’ reductions in emissions. (A full model description is in our earlier working paper (Goulder et al., 1998) and in an appendix available from the authors on request.)

3.1. Model structure

3.1.1. Household behavior

As in the previous model, a representative household derives utility from consumption (of C_N and C_D) and from leisure, according to the constant-elasticity-of-substitution (CES) utility function:

$$U = U(l,C_D,C_N,E) = \left(\alpha_l l^{(\sigma_U-1)/\sigma_U} + \alpha_F C^{(\sigma_U-1)/\sigma_U}\right)^{\sigma_U/(\sigma_U-1)} - \phi(E), \qquad (3.1)$$

where C is composite consumption, defined by

$$C = \left(\alpha_{C_D} C_D^{(\sigma_C-1)/\sigma_C} + \alpha_{C_N} C_N^{(\sigma_C-1)/\sigma_C}\right)^{\sigma_C/(\sigma_C-1)}, \qquad (3.2)$$

and where $l \equiv \bar{L} - L$ represents leisure time, E is aggregate emissions, and the α’s are parameters.[17] σ_U and σ_C are the elasticities of substitution between goods and

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[17] Homothetic preferences over consumption goods, together with separability between consumption goods and leisure, imply that consumption goods are equal substitutes for leisure (Deaton, 1981).

leisure and between the two consumption goods, respectively. The household maximizes utility subject to the budget constraint:

$$p_{C_D}C_D + p_{C_N}C_N = p_L L(1 - t_L) + \pi(1 - t_R) + p_C G, \tag{3.3}$$

where t_L is the tax rate on labor income, t_R is the tax rate on rent income, L is labor time, π is policy-generated rent (applicable under the permit policy), G is (constant) real government spending in the form of transfers to households, and p_C is the composite price of consumption. Except in the sensitivity analysis in Section 5, the tax rates t_L and t_R are assumed to be the same. Taxes finance a fixed real level of government transfers to households.

3.1.2. Firm behavior

We use a CES form for production functions in all industries:

$$X_j = \left(\sum_i \alpha_{i,j} X_{i,j}^{(\sigma_j - 1)/\sigma_j}\right)^{\sigma_j/(\sigma_j - 1)},\ i = \{D,N,L\},\ j = \{D,N,C_D,C_N\}, \tag{3.4}$$

where X is output, the $\alpha_{i,j}$'s are share parameters, and the σ_j's are the elasticities of substitution between factors in production. Pollution emissions from industry j, E_j, follow

$$E_j = \beta_D X_{D,j}\left[1 - \alpha_E\left(\frac{A_j}{\beta_D X_{D,j}}\right)\right]^{\gamma}, \tag{3.5}$$

where A_j is real expenditure (in the form of additional labor input) by industry j on emissions abatement, α_E and γ are parameters describing the emissions abatement technology, and β_D represents emissions per unit of the polluting intermediate good. This emissions function is homogenous of degree one in abatement spending and in the amount of the polluting intermediate good used, which is consistent with the assumption of constant returns in production. The parameter γ determines the curvature of the abatement cost function. This function is assumed to be convex: $\gamma < 1$. Our central case value for γ is 0.5, implying that marginal abatement costs are linear.

Producers choose the profit-maximizing input quantities and abatement subject to any constraints imposed by pollution regulation, taking input and output prices as given. Profits equal the value of output minus expenditures on labor, intermediate inputs and abatement, less any tax charged per unit of residual emissions (τ_e) or per unit of output (τ_j). Thus, profit for industry j (π_j) is:

$$\pi_j = (p_j - \tau_j)X_j - \sum_i p_i X_{i,j} - \tau_e E_j - A_j, \tag{3.6}$$

where p_i and p_j are the prices of inputs and outputs, respectively. Note that because the production function and abatement function both exhibit constant returns to scale, profits will equal zero under all policies except the permit policy, under which profits equal permit rents.

3.1.3. Government policy

With this extended model we examine five policies: an emissions tax, a set of emissions permits, a fuel tax, a performance standard, and a technology mandate. These correspond to the policies examined earlier, except that by including intermediate inputs we now can distinguish the performance standard and technology mandate. As before we distinguish between a least-cost and higher-cost technology mandate.[18] The fuel tax is now levied on a (polluting) intermediate input rather than on a final good.

We assume that each policy is implemented in a way that avoids introducing additional efficiency losses attributable to the uneven treatment of industries. Therefore, in all cases policies are introduced so as to ensure that the private marginal cost of emissions reduction is equated across industries. Hence the emissions tax policy applies the same tax rate to all industries, the emissions permit is set such that the shadow price of the permit is constant across industries, and so forth.

The government's budget constraint is:

$$p_C G = t_L(T - l) + t_R \pi + \tau_e \sum_j E_j + \sum_j \tau_j X_j. \tag{3.7}$$

The tax rates t_L and t_R on labor and rent income are adjusted to hold government revenue constant.

3.1.4. Equilibrium conditions

In general equilibrium, supply must equal demand for all produced goods, government revenue must equal government transfer payments, and pollution emissions must equal a specified target. Because production and abatement functions are linearly homogeneous, the supply of each good is perfectly elastic at given factor prices and tax rates. Under these conditions the set of equilibrium conditions reduce to three equations: aggregate labor demand equals aggregate supply, government revenue equals expenditures, and aggregate pollution emissions equal the target level.[19]

3.2. Data and parameters

Table 3 summarizes our benchmark data set, which depicts the United States economy in 1990. Production data were obtained from the Commerce Department

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[18] In the presence of heterogeneity a similar distinction could be made for the performance standard.

[19] More details on the solution of the numerical model are provided in Goulder et al. (1998).

Table 3
Benchmark data for the numerical model. Input–output flows (in millions of 1990 dollars per year except as otherwise noted)

	D	N	C_D	C_N	Leisure time	Total output value
D	91,441.0	111,842.7	156,881.1	6264.3		366,429.1
N	88,073.5	4,741,097.5	464,159.9	2,670,485.6		7,963,816.5
L	186,914.7	3,110,876.3			1,832,106.1	5,129,897.1
Total output value	366,429.1	7,963,816.5	621,041.0	2,676,750.0		
Emissions (millions of tons)	23.0					

$\sigma_D = \sigma_N = 0.8$ (elasticities of substitution in intermediate goods production).
$\sigma_{C_D} = \sigma_{C_N} = 0.9$ (elasticities of substitution in final goods production).
$\sigma_C = 0.85$, $\sigma_U = 0.96$ (elasticities of substitution between final goods and between consumption and leisure, respectively).
$\alpha_E = 1.55 \times 10^{-4}$ (effectiveness of technological abatement).
$\gamma = 0.5$ (curvature of abatement cost function – implies linear marginal abatement costs).

Bureau of Economic Analysis. The pollution-related intermediate good comprises fossil fuels (oil, coal, and natural gas), while the clean intermediate good includes all other intermediates. The consumer good C_D is a composite of the consumer goods whose production involves intensive use of fossil fuels (utilities, motor vehicles, and gasoline), while the good C_N embraces all other final goods.

Elasticities of substitution in the production functions and the inner nest of the consumer utility function are taken from the disaggregated general equilibrium data set developed by Cruz and Goulder (1992). The α distribution parameters for production functions were calibrated based on assumed elasticities of substitution and the identifying restriction that firms adopt cost-minimizing mixes of inputs.

An important preference parameter is σ_U, the consumption-leisure substitution elasticity. We choose the value for this parameter, along with the labor time endowment, to imply uncompensated and compensated labor supply elasticities of 0.15 and 0.4, respectively. These are typical estimates from the literature and are meant to represent the effects of changes in the real wage on average hours worked and the labor force participation rate.[20] We assume a pre-existing tax rate on labor and rent income of 40 percent.[21] These parameters imply a marginal

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[20] These values are roughly consistent with a recent survey of opinion among labor economists by Fuchs et al. (1998), assuming a weight of 0.6 and 0.4 for the male and female elasticities reported in Table 2 of that survey.

[21] The effective tax rates on labor and profit income are roughly the same: labor is subject to personal income and payroll tax, while capital is subject to personal and corporate income tax. Total revenues from income taxes amount to about 36 percent of net national product. The marginal tax rate is higher because of various deductions. Other studies assume factor income rates of around 40 percent (Browning, 1987; Lucas, 1990).

excess burden of labor taxation in our model equal to 0.3, which is consistent with other studies (see, e.g., Browning, 1987).

Our central case values for pollution-related parameters are based on characteristics of NO_x emissions. The pollution content for the polluting intermediate good (the β_D parameter) is derived by dividing the actual 1990 emissions of NO_x, as given by Pechan et al. (1996), by the quantity of the polluting good. The parameter α_E, which expresses the effectiveness of abatement, was calibrated so that 2.2 million tons of emissions abatement (relative to initial emissions of 23.0 million tons) could be achieved at a marginal cost of US$500 per ton, a figure extrapolated from Pechan et al. (1996). Although our model is benchmarked based on characteristics of NO_x emissions, in Section 4.4 below we adopt different data assumptions to examine how alternative characteristics of pollution generation or abatement might affect policy costs.

4. Numerical results

To facilitate comparisons among policies we use the emissions tax costs as a reference point, comparing the costs of other policies to these costs. We emphasize the rankings of policy costs, as opposed to absolute costs, and examine how the rankings depend on parameters of the abatement-cost, production, and utility (consumer demand) functions. These parameters affect cost by determining the relative contributions of the abatement, and input-substitution, and output-substitution effects. We abstract from heterogeneity in firms' production technologies (and thus in abatement costs). Such heterogeneity can affect – perhaps dramatically – the relative costs of command-and-control policies by imposing additional informational burdens or by affecting the extent to which regulations equate marginal abatement costs across firms or industries.

4.1. First-best costs

We first examine costs in a first-best setting ($t_L = 0$), where only the primary costs apply. Fig. 1a shows costs under the different policy instruments, expressed as the ratio of the total costs of the policy in question to the total costs under the emissions tax.[22] The curve for the permit policy is constant at unity; that is, the costs of the emissions tax and permit policy are identical at all levels of emissions reduction. In the absence of distortionary taxes, the tax-interaction and revenue-recycling effects do not apply, and thus the source of differences in cost impacts – the revenue-recycling effect – is absent.

The first-best cost of the performance standard exceeds that of the emissions tax.

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[22] These are calculated by setting the labor tax in the numerical model equal to zero and returning any government revenues from the policies as lump sum transfers to households.

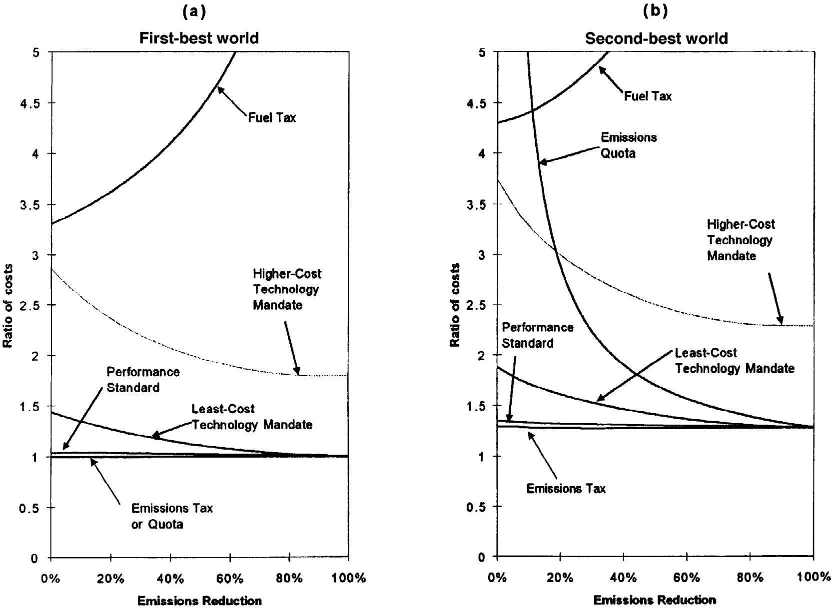

Fig. 1. Ratio of cost under policy alternative to cost under first-best emissions tax.

This is the case because the performance standard does not charge firms for residual emissions; hence the output-substitution effect is inefficiently weak. However, the cost discrepancy between the two instruments is very small because the output-substitution effect plays a relatively small role in reducing NO_x emissions.[23]

The cost of the least-cost technology mandate is higher than that of the performance standard. The technology mandate is the more restrictive of the two policies because firms have no incentive to reduce emissions by input-substitution. We assume that the abatement costs under the higher-cost technology mandate are twice those under the least cost mandate (i.e., $\theta = 2$). Under this assumption, the general equilibrium total costs of the higher-cost mandate in the first-best setting turn out to be roughly twice those of the least-cost mandate at all levels of emissions reduction.

Finally, the first-best cost of the fuel tax exceeds that of the emissions tax. Under the fuel tax, the abatement effect is absent. The bulk of NO_x emissions reductions under the emissions tax come from the abatement effect, and thus the fuel tax is much more costly than the emissions tax.

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[23] In the central case, a 50 percent reduction in emissions under an emissions tax achieves 79.0 percent of that reduction through the abatement effect, 18.5 percent through the input-substitution effect, and only 2.5 percent through the output-substitution effect.

Importantly – and perhaps surprisingly – the costs of the least-cost technology mandate and performance standard converge to those of the emissions tax and permit policy at 100 percent emissions reduction. This finding is robust to different parameter assumptions, as confirmed in the sensitivity analysis below.[24] In contrast with the emissions tax case, under the performance standard firms are not charged for their residual emissions, and thus the output-substitution effect is not fully utilized. This difference declines in importance at higher levels of emissions reduction. At 100 percent emissions reduction, there are no residual emissions, and both policies have the same effect on the price of the polluting good and the same output-substitution effect. Thus, their costs converge to the same value.[25] Like the performance standard, the technology mandate fails to fully utilize the output-substitution effect. Furthermore, the technology mandate fails to exploit the input-substitution effect, and thus has higher costs. However, the importance of this effect converges to zero as emissions reductions approach 100 percent. Thus the technology mandate's costs also converge to those of the other policies.

4.2. The significance of pre-existing taxes

Fig. 1b shows the general equilibrium costs of the policy instruments in a second-best setting with $t_L = 0.4$ initially. These costs are expressed relative to those of the emissions tax in a first-best world. By comparing results of Fig. 1b with those in Fig. 1a, one observes the impact of pre-existing taxes.

For the emissions tax, second-best costs exceed first-best costs by a constant factor of 1.27 – the tax-interaction and revenue-recycling effects contribute to a 27 percent increase in the costs of the emissions tax.[26] The second-best costs of the performance standard, technology mandate and fuel tax exceed those of the emissions tax by the same proportions as in Fig. 1a; pre-existing taxes raise the costs of all these policies by about 27 percent. The net impact of the tax-

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[24] The form of our abatement cost function (Eq. (3.2)) implies that firms can entirely eliminate emissions at finite cost. For many pollutants, for example lead and chlorofluorocarbons, this is a realistic assumption, since there are substitute inputs or alternative processes that yield the same economic services and involve nearly zero costs. But the assumption is not always apt. Williams (1998b) examines the scenario where marginal abatement costs approach infinity as emissions approach zero and finds that the convergence result still holds in some cases.

[25] Fig. 1a also shows that the relative inefficiency of the fuel tax is larger, the greater the extent of emissions reduction. This is not necessarily a general result, but arises because of our assumed functional forms. For details, see Goulder et al. (1998).

[26] The 27 percent increase is not entirely attributable to the net impact of the tax-interaction and revenue-recycling effects. We find that when one moves from a first-best to a second-best setting, primary costs rise. In our numerical simulations about half of the 27 percent increase reflects the increase in primary costs.

interaction and revenue-recycling effects is proportional to primary cost.[27]

The results of Fig. 1b for the impact of prior taxes are particularly striking in the case of the permit policy. Here, interactions with the tax system raise the overall costs by a larger amount, reflecting the fact that permits do not generate a revenue-recycling effect to counteract the tax-interaction effect. The differences in costs are especially great at modest levels of abatement. For emissions reductions below 24 percent, the cost of the permit policy is more than double that of the tax. However, under the emissions tax the marginal revenue-recycling effect declines with the level of emissions reductions as the tax base is eroded. At 100 percent abatement, no emissions tax revenues are raised and there is no revenue-recycling effect; thus the total costs of the emissions tax and permit are equal.[28]

Four points deserve emphasis. First, only the permit policy has positive marginal costs at initial, incremental abatement. An incremental amount of abatement increases the effective tax on labor through the tax-interaction effect. In a world with prior labor taxes, an incremental increase in the effective labor tax yields a first-order welfare cost – marginal costs are not incremental. In contrast, under the emissions tax marginal costs are zero at initial abatement, so the ratio of total costs of the permit and emissions-tax policies is infinite. For all the other instruments, at initial abatement the tax-interaction and revenue-recycling effects either are zero or they exactly offset each other, so that marginal costs are zero.[29]

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[27] For the two tax instruments, this proportionality occurs because the revenue-recycling effect exactly offsets the portion of the tax-interaction effect resulting from the charge on residual emissions, leaving a net second-best effect that is proportional to the first-best cost of the tax, as discussed in Section 2. The technology mandate and performance standard do not charge firms for residual emissions; hence under these policies the increase in price of final output only depends on the costs of abatement and input substitution – the first-best costs. Thus, in these cases, the tax-interaction effect itself is proportional to the first-best cost of the policy. Since these policies do not involve a revenue-recycling effect, the overall second-best effect is proportional as well.

[28] Note that the numerical model incorporates an indirect revenue-recycling effect from the taxation of permit rents. This equals 40 percent of the revenue-recycling effect under the emissions tax. Thus, the difference between the tax and permit is smaller than predicted by the analytical model, which ignores the taxation of permit rents.

[29] In the case of the performance standard and technology mandate, the tax-interaction and revenue-recycling effects are both zero at initial abatement. Thus the marginal cost curves emerge from the origin. A similar result was obtained by Fullerton and Metcalf (1997) under their technology-restriction policy, which resembles the performance standard considered here. The permit policy's marginal cost curve does not emerge from the origin because it has an efficiency loss from the tax-interaction effect and no offsetting revenue-recycling effect. There are other cases under which the net impact of the tax-interaction and revenue-recycling effects is strictly positive at incremental abatement. These include the case where the government introduces an emissions tax and returns the revenues in lump-sum fashion, and the case where the government policy increases the returns to a perfectly inelastic factor of production (Williams, 1998c, and, in the context of trade policy, Williams, 1998a). What is common to all these cases is that the government policy has effectuated a lump-sum transfer to the private sector, either by generating untaxed scarcity rents (the case emphasized by Fullerton and Metcalf), by providing explicit lump-sum transfers, or by generating additional rents to fixed factors. In a world with distortionary taxes, lump-sum transfers from the government to the private sector are costly in efficiency terms because the government ultimately must finance such transfers through distortionary taxes.

Thus, for these other policies, the ratio of total costs is finite at initial, incremental abatement.

Second, the efficiency consequences of the permit policy depend importantly on the fact that permits are not auctioned, which means that the revenue-recycling effect is not exploited. If permits were auctioned and the revenues used to finance cuts in the marginal rates of pre-existing taxes, the efficiency impacts would be the same as that of the emissions tax.

Third, these results bear importantly on the evaluation of tradable permits systems. A key attraction of such systems is that they help achieve a more efficient allocation of abatement effort by promoting an equilibrium in which producers' marginal costs of abatement are equal. Typical estimates indicate that allowing for trades can reduce costs of compliance by 30 percent or more relative to the costs of a system with fixed emissions permits (that is, with no trades).[30] The results of Fig. 1b indicate that second-best considerations can have an equal or larger impact on costs. *The decision whether to grandfather or auction the permits can be as important to policy costs as the decision about whether to allow trades*.

Finally, the presence or absence of the revenue-recycling effect is also very important to the costs of emissions taxes. If revenues from an emissions tax were returned as lump-sum payments rather than used to reduce pre-existing tax rates, the revenue-recycling effect would not materialize and the costs of the emissions tax in our model would be the same as those of the (non-auctioned) emissions permit. Thus, Fig. 1b indicates that the cost of achieving a given abatement target can depend as much on how revenues are recycled as on the choice of whether to introduce an emissions tax or a fuels tax, particularly when modest levels of abatement are involved.

Regulations promulgated under the 1990 Clean Air Act Amendments are expected to yield about a 20 percent economy-wide reduction in NO_x emissions (relative to the projected baseline) in 2005.[31] These regulations rely mainly on performance standards. Our modeling suggests that pre-existing taxes raise the cost of these regulations in 2005 by about US$1.3 billion (1990 dollars). Fig. 1b suggests that achieving this overall reduction would be considerably more costly under (non-auctioned) tradable permits than under performance standards, unless the performance standards were imposed in an extremely inefficient way.[32]

[30] Tietenberg (1985) surveyed 11 studies, with costs under command and control estimated to be over six times larger on average as costs under the ideal least-cost approach. Numerous other studies have found significant cost savings from tradable pollution permit programs, although costs under these programs typically exceed the theoretical minimum by a substantial amount, in part because of flaws in program design (Hahn, 1989).

[31] This estimate is extrapolated from Pechan et al. (1996).

[32] Here we compare costs of achieving reductions in NO_x emissions by performance standards alone and by permits alone. There have been several recent proposals to superimpose regional permit-trading programs on existing regulations (which mainly involve performance standards). Calculating the costs of this mix of regulations requires attention to regional issues as well as considerably more industry disaggregation than that of the present model. In future work we hope to perform such cost calculations.

4.3. Alternative scenarios

We now examine the impacts of alternative parameterizations that change the relative importance of the output-substitution, input-substitution and abatement effects. The first alternative reduces α_E in (3.2) in order to quadruple the marginal abatement cost relative to the central case at given levels of abatement.[33] Fig. 2a shows the implications of this change for the second-best cost of each policy relative to the first-best emissions tax. Changing this parameter has virtually no effect on the curves for the emissions permit or performance standard, relative to the corresponding curves in Fig. 1b.[34] However, this change significantly raises the relative cost of the technology mandates, since technology mandates achieve a much greater proportion of emissions reduction through spending on abatement than does the emissions tax. In contrast, the absolute cost of the fuel tax remains unchanged in this scenario, since it does not utilize the abatement effect. Hence this policy's relative cost is now lower.

In Fig. 2b we quadruple the value of the substitution elasticity between consumer goods to increase the relative importance of the output-substitution effect. This has very little impact on the position of the curves relative to those in Fig. 1b because it is still the case that only a small fraction of the emissions

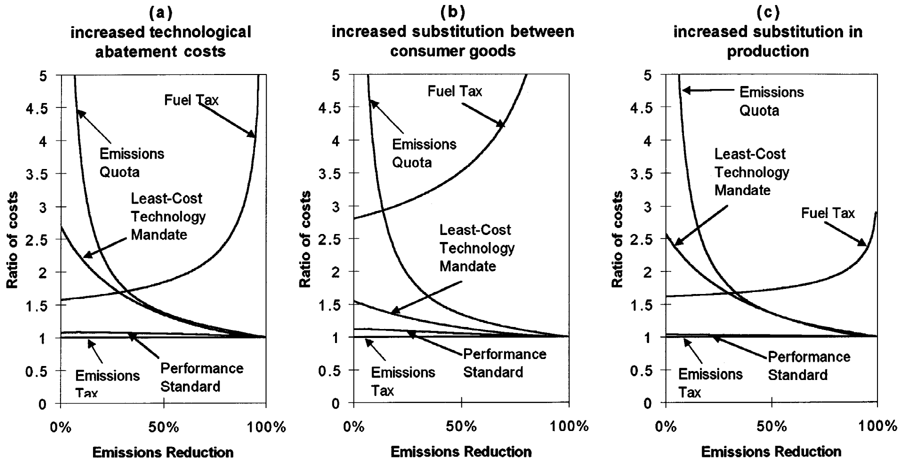

Fig. 2. Cost ratios under alternative parameter assumptions.

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[33] This variable differs widely among pollutants. For instance, the abatement effect is relatively less important in the context of reducing SO_2 emissions than for NO_x emissions. Some of the reduction in SO_2 emissions following the 1990 Clean Air Act Amendments has come from installing abatement technologies (scrubbers) but a more substantial part has come from substituting in favor of cleaner inputs (lower sulfur coal). In the case of CO_2, there are no commercially available abatement technologies.

[34] That is, the *ratio* of the cost under each policy to the cost under the emissions tax does not change. The *absolute* costs are of course greater in this alternative scenario.

reduction is due to the output substitution effect. Indeed, for all major pollutants, the bulk of emission reduction comes from reducing the emissions-output ratio, rather than by substituting away from pollution-related goods in consumption.

In Fig. 2c we quadruple the elasticity of substitution in production, which increases the importance of the input-substitution effect. This has virtually no impact on the relative cost of the emissions permit or performance standard. The technology mandate, however, derives relatively little of its emissions reductions from the input-substitution effect, so its cost relative to the emissions tax rises, even though its absolute cost falls slightly. In contrast, the relative cost of the fuel tax falls because it relies on the input substitution effect much more than the emissions tax does.

4.4. Efficiency impacts

We now consider the net efficiency impacts – environmental benefits less economic costs – of the different policies (under central values for parameters). Here we posit a range of values for the (constant) marginal benefits from reductions in NO_x emissions (or marginal damages from such emissions), and calculate the optimal level of abatement and net efficiency gain associated with each posited value.

Fig. 3a displays, in a first-best world, the ratio of the maximum efficiency gain

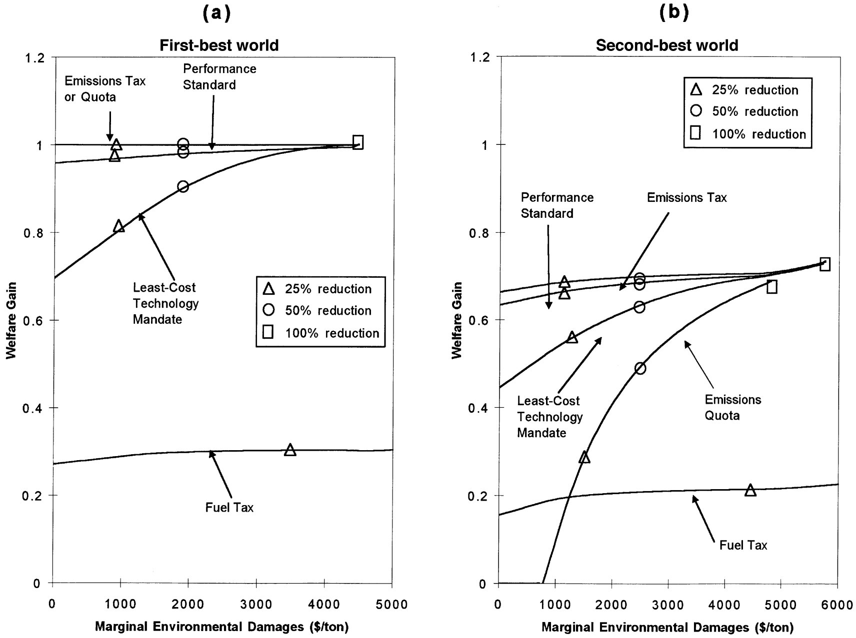

Fig. 3. Ratio of maximum welfare gain under given instrument to maximum gain under first-best tax.

under each policy to the efficiency gain under the optimal emissions tax. Fig. 3b offers complementary information: the ratio of the maximum gain under each policy in a second-best world to the maximum gain from an emissions tax in a first-best world. Thus, the differences in the ratios for Fig. 3a,b reveal the significance of pre-existing taxes. On each graph, the horizontal axis depicts the posited value for marginal pollution damages, while the triangles, circles, and rectangles respectively show, for each policy, the level of marginal damages that implies optimal emissions reductions of 25, 50, and 100 percent.

In the first-best case, the initial marginal cost for each policy is zero; hence there is scope for an efficiency gain so long as marginal pollution damages (marginal benefits from abatement) are positive. The potential efficiency gains are largest under the emissions tax and permit, as expected, since these policies have the lowest primary costs. Policies with higher primary costs produce lower potential efficiency gains.

The level of marginal damages necessary to justify a given level of emissions reduction also varies among the different policies. Marginal costs of emissions reductions are lowest for the emissions tax and permit; hence a given level of emissions reduction is justified at a lower level of marginal benefits. The other policies have higher marginal costs and thus require higher levels of marginal damages to justify a given level of emissions reduction. When marginal damages justify a 100 percent emissions reduction under the other policies, the efficient level of emissions reduction under the fuel tax is less than 50 percent.

In the second-best case shown in Fig. 3b, the same principles apply, but since the costs of all policies are higher, the potential overall efficiency gains are substantially lower. Under emissions permits, second-best elements have the most profound impact on marginal costs and, consequently, on potential efficiency gains. In fact, tradable permits cannot produce an efficiency gain unless marginal damages exceed a certain threshold value. This stems from the fact that, in the second-best case, the initial marginal cost of a permit policy is strictly positive, in contrast with the other policy instruments examined here.

As in the first-best situation, in a second-best world the efficient level of emissions reduction associated with a given value for marginal damages varies across policies. The sharpest differences are between the fuel tax and other policies. At values for marginal damages that justify nearly 100 percent emissions reduction for the other policies, the optimal emissions reduction under the fuel tax is only about 25 percent. This reflects the fact that marginal abatement costs rise most sharply under the fuel tax.

4.5. Further sensitivity analysis

Table 4 summarizes the sensitivity of the numerical results to a range of values for other important parameters. We vary the compensated and uncompensated labor supply elasticities, the initial labor tax rate, and the curvature parameter for

Table 4
Sensitivity analysis. Ratio of total costs under each policy to total costs under the emissions tax

	Emissions quota	Performance standard	Least-cost technology mandate	Fuel tax
1. Central case				
25% reduction	1.992	1.035	1.238	3.677
50% reduction	1.345	1.025	1.112	4.348
75% reduction	1.123	1.014	1.036	5.804
2a. Uncompensated labor supply elasticity = 0.0 $(\sigma_U = 0.6)$				
25% reduction	1.983	1.035	1.236	3.700
50% reduction	1.346	1.025	1.112	4.410
75% reduction	1.123	1.013	1.036	6.022
2b. Uncompensated labor supply elasticity = 0.3 $(\sigma_U = 2.4)$				
25% reduction	1.997	1.035	1.237	3.656
50% reduction	1.350	1.025	1.112	4.294
75% reduction	1.126	1.014	1.036	5.705
3a. Compensated labor supply elasticity = 0.2 $(\sigma_U = 2.4,\ T = 3.587\times10^{12})$				
25% reduction	1.499	1.035	1.237	3.672
50% reduction	1.174	1.025	1.111	4.315
75% reduction	1.062	1.014	1.036	5.645
3b. Compensated labor supply elasticity = 0.6 $(\sigma_U = 0.8,\ T = 7.795\times10^{12})$				
25% reduction	2.490	1.035	1.238	3.683
50% reduction	1.522	1.025	1.112	4.388
75% reduction	1.187	1.014	1.036	6.010
4a. Initial labor tax rate = 0.2				
25% reduction	1.369	1.035	1.236	3.692
50% reduction	1.130	1.025	1.112	4.363
75% reduction	1.046	1.013	1.035	5.803
4b. Initial labor tax rate = 0.6				
25% reduction	3.240	1.033	1.234	3.652
50% reduction	1.845	1.024	1.114	4.682
75% reduction	1.332	1.013	1.039	[a]
5a. Abatement cost curvature parameter $\gamma = 0.6667$				
25% reduction	1.776	1.022	1.122	3.876
50% reduction	1.256	1.011	1.041	6.457
75% reduction	1.087	1.005	1.012	10.832
5b. Abatement cost curvature parameter $\gamma = 0.3333$				
25% reduction	2.268	1.057	1.559	3.641
50% reduction	1.475	1.063	1.430	2.710
75% reduction	1.202	1.054	1.199	2.496

[a] In this case, for emissions reductions over 60% government revenues are insufficient to cover expenditures for any value of the labor tax rate, and thus no equilibrium exists.

the abatement cost function. The table displays the ratio of the total cost under each policy to the total cost under the emissions tax in a second-best setting, under a range of parameter values. These cost ratios are shown for emissions reductions of 25, 50, and 75 percent.

In the second and third rows we vary labor supply elasticities and pre-existing tax rates. Higher elasticities imply a greater degree of substitution between consumption and leisure, and this strengthens the tax-interaction and revenue-recycling effects. The costs of emissions permits are more sensitive to these elasticities than the costs of the emissions tax and fuel tax, since emissions permits do not generate the (offsetting) revenue-recycling effect. The costs of the performance standard and technology mandate are also less sensitive to these elasticities than emissions permits, since the tax-interaction effect is less important for these policies. For similar reasons, the relative costs of emissions permits are more sensitive to higher rates of pre-existing taxes, which strengthen the tax-interaction and revenue-recycling effects.

The final set of rows considers cases where firms' marginal cost curves are concave (row 5a) or convex (row 5b) in abatement expenditure, as opposed to linear (see (3.2)).[35] When these curves become concave (convex), the importance of the abatement effect is increased (decreased). Consequently, the relative costs of the performance standard and technology mandate increase (decrease) while the relative costs of the fuel tax are reduced (increased). This occurs because the performance standard and technology mandate rely relatively heavily on the abatement effect, while the fuel tax does not utilize this effect.

5. Conclusions

This paper has employed analytical and numerical general equilibrium models to compare, in first- and second-best settings, the cost-effectiveness of a range of environmental policy instruments. We find that, in both first- and second-best settings, the relative costs of the different instruments depend crucially on the level of emissions reduction. Indeed, for all of the instruments except the fuel tax, the costs of abatement converge as the level of abatement approaches 100 percent.

Pre-existing taxes significantly raise the costs of all environmental policies relative to their costs in a first-best world. The cost increase is proportionally larger for (non-auctioned) emissions permits than for the other policy instruments. Earlier work on instrument choice has emphasized the superiority of tradable emissions permits relative to command-and-control policies like performance standards and mandated technologies. Our results indicate that pre-existing taxes can undo the cost advantage of tradable permits over these other instruments.

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[35] For more details on this simulation, see Goulder et al. (1998).

Indeed, numerical results suggest that achieving the NO_x emissions reductions required under the 1990 Clean Air Act Amendments would be more costly under tradable permits than under performance standards unless the permits were auctioned.

Some limitations in the present study deserve attention. First, our analysis does not incorporate heterogeneity among polluting firms within a given industry. The significance of heterogeneity extends beyond the issue, discussed above, of the attractiveness of allowing trades in emissions rights. Heterogeneity augments the information burdens faced by regulators and consequently implies that mandated technologies will tend to be less efficient than suggested here, because regulators will have a difficult time discerning what technology is most appropriate. In addition, heterogeneity implies that many forms of regulation will involve serious costs of standard-setting, monitoring, and enforcement. To the extent that it is easier, for example, to monitor fuels or the use of mandated equipment than it is to monitor emissions, the fuel tax and mandated technology would enjoy an advantage over other policies. Thus, heterogeneous production and the associated information problems produce additional cost considerations that could importantly affect the relative attractiveness of the policies we have considered.[36]

Second, our analysis is static. Clearly the various policy instruments will differ in terms of their impacts for investments in research and development and the associated potential to reduce future costs of abatement. These dynamic effects may have significant welfare consequences that are not captured by our analysis.[37]

Finally, this study concentrates solely on efficiency issues. A comprehensive evaluation of alternative policy instruments must also take account of distributional impacts. Indeed, some would maintain that political feasibility is influenced far more by distributional concerns than by efficiency calculations. Future work that integrates efficiency and distributional issues in a second-best context may provide additional useful and highly practical policy insights.

Acknowledgements

The authors gratefully acknowledge helpful comments from Jesse David, Carolyn Fischer, Don Fullerton, Koshy Mathai, Robert Stavins, and Michael Toman. We also thank the National Science Foundation (grants SBR-9310362 and SBR-9613458) and Environmental Protection Agency (grant R825313-01) for financial support.

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[36] See, for example, Tietenberg (1985).

[37] For a theoretical investigation of the dynamic implications of instrument choice, see Milliman and Prince (1989) and Fischer et al. (1998). For a recent empirical examination, see Newell et al. (1998).

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